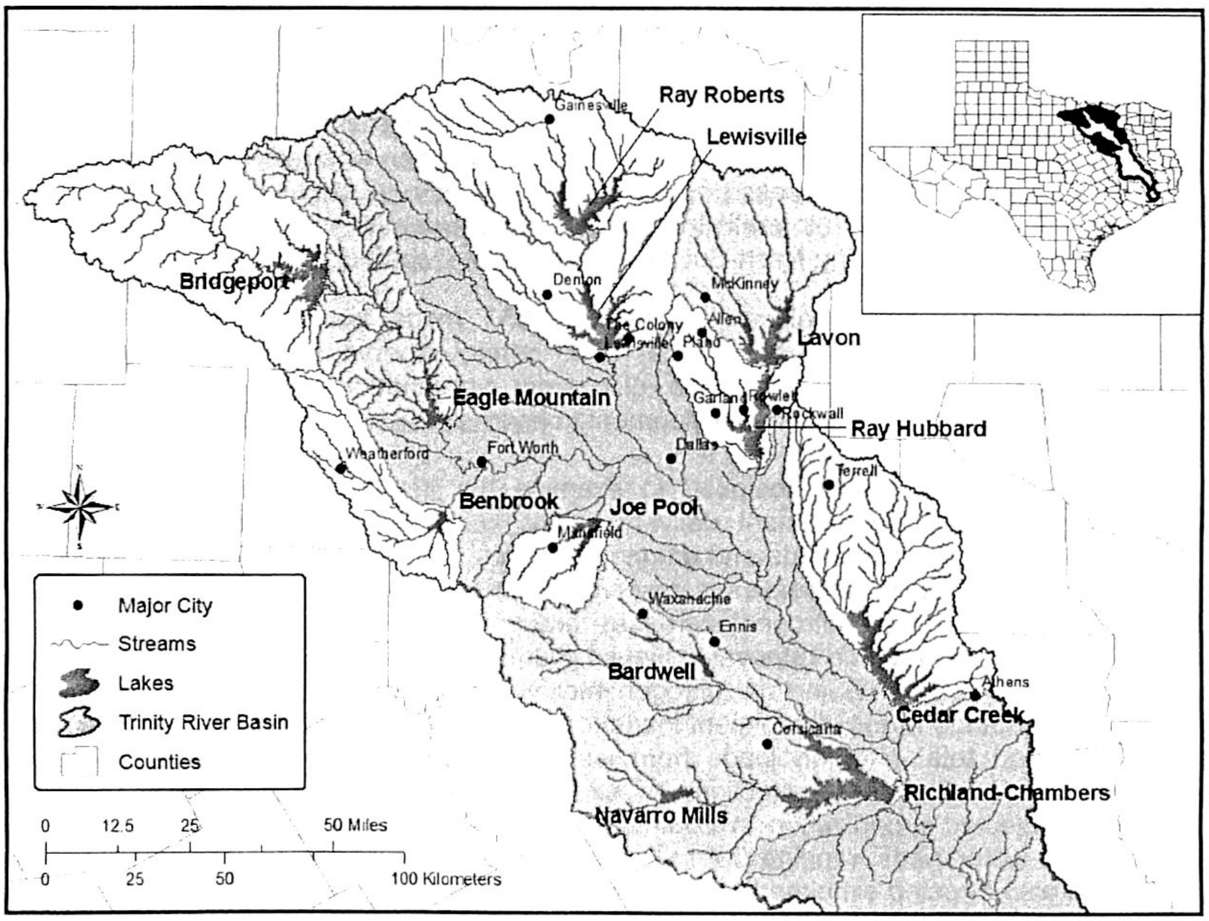
A report by the Tarrant Regional Water District, Spatial Sciences Laboratory, Texas AgriLife Urban Solutions Center, Texas AgriLife Blackland Research and Extension Center, and the USDA Agricultural Research Service Grassland, Soil and Water Research Laboratory

Trinity River Basin Environmental Restoration Initiative 2010

By Xiuying Wang, Michael White, Taesoo Lee, Pushpa Tuppad, Raghavan Srinivasan, Allan Jones and Balaji Narasimhan
Edited by Kristina Twigg



TWDB

Funded by the Texas Water Development Board
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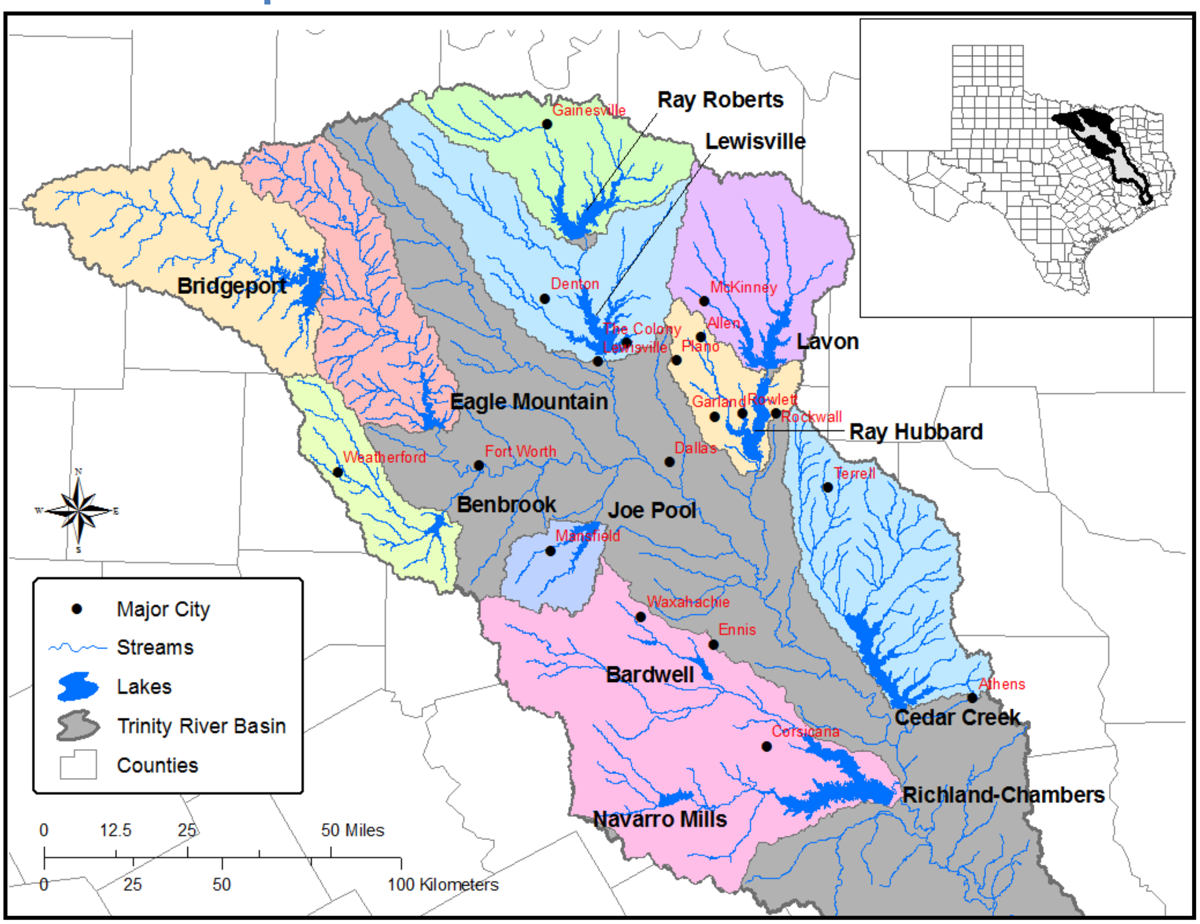

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Executive Summary

The Trinity River Basin begins near the Red River, northwest of Fort Worth. It extends southeast, through the hearts of Fort Worth and Dallas, to the Gulf of Mexico just east of Houston. It is the most populated river basin in Texas and supports most primary water needs for both the Dallas-Fort Worth and Houston areas. The Upper Trinity Basin, which includes the Dallas-Fort Worth Metroplex, supplies water to about one-fourth of Texas' population.

The anticipated rapid growth of North Central Texas over the next 20 to 50 years will almost certainly increase regional demands for high quality drinking water. This has increased concerns that sediment and nutrient loads received by drinking water reservoirs are and will continue to seriously reduce reservoir volumes and water quality.

The objectives of this project are to use the Soil and Water Assessment Tool (SWAT) model to (1) assess current rates and sources of sediment and nutrient loading of twelve major water supply reservoirs in ten watersheds of the Upper Trinity River Basin, (2) to predict the effects of anticipated future urbanization and (3) to consider possible conservation practices that could decrease sediment and nutrient loading of these reservoirs.

The Upper Trinity River Basin study area was delineated into ten watersheds, encompassing 12 major reservoirs. To make sure modeled sediment and nutrient values closely matched observed results, we calibrated and validated the SWAT model for monthly and annual flows at USGS gauging stations located generally within or at the outlet of each watershed. We also calibrated and validated SWAT for water quality parameters, including sediment, organic and mineral nitrogen, organic and mineral phosphorus, total nitrogen and total phosphorus, where these data were available. To assess how well observed and predicted values matched and to determine that SWAT was able to realistically predict water quantity and quality, we used model performance statistics, including Nash-Sutcliffe efficiency and coefficient of determination (R^2).

For each individual basin, a calibrated baseline SWAT model was then used for scenario analyses that included: removal of all small flood control reservoirs (ponds), removal of all wastewater treatment plant point-source discharges, elimination of livestock grazing on rangelands, and expansion of urban development to levels predicted for 2030.

Each simulated scenario was designed to examine the impacts of possible conservation practices on sediment and nutrient delivery to the reservoirs. The impacts of simulated scenarios varied within watersheds and among scenarios. For example, removal of small flood control structures produced increases in reservoir sediment loading ranging from 4% to 48%; total phosphorus and total nitrogen increases ranged from 4% to 10% based on model-predicted values. Point-source removal achieved reductions ranging from 0.3% to 24% in total phosphorus and 1% to 56% in total nitrogen received by the reservoirs. Finally, eliminating grazing on rangelands reduced sediment loads from 0.3% to 37%, total phosphorus loads from 0% to 11%, and total nitrogen loads from less than 1% to 19%. Furthermore, we used population and development projections to examine the impacts of urbanization on each watershed. Projected urbanization in 2030 had large effects on simulated total phosphorus loads in some watersheds, ranging from a reduction of 1% to an increase of 111%. Projected urbanization also affected simulated total nitrogen loads, from a reduction of 3% to an increase of 24%. Likewise, SWAT predicted changes in sediment loads due to urbanization, from a reduction of 10% to an increase of 32%. See table i.1 for a summary of these results.

Table i.1 Various scenarios modeled in this report and their impact on reservoir loading with sediment and nutrients. (Positive numbers indicate percent increases in sediment, nitrogen or phosphorus loading while negative numbers indicate percent decreases.)

Scenarios	Sediment loading	Nitrogen loading	Phosphorus Loading
Removal of small flood control structures	4 to 48	4 to 10	6 to10
Point-source removal	Not Applicable	-1 to -56	-0.3 to -24
Eliminating grazing	-0.3 to -37	-0.3 to -19	-0.02 to -11
Urbanization	-10 to 32	-3 to 24	-1 to 111

Differences in current land use and land cover, future urban changes, rangeland management, and the total basin area draining into ponds in each watershed affected the generation of constituent pollutants and reservoir loading. The reservoirs most sensitive to 2030 urbanization levels, as simulated by the SWAT model, were Benbrook, Joe Pool and Bridgeport. Lewisville, Ray Roberts, Cedar Creek and Lavon were most sensitive to pond removal while Lewisville, Benbrook, and Bridgeport responded most to elimination of grazing on rangelands. In urban areas and others not dominated by agriculture, point sources could be major contributors to water quality problems. Those basins most sensitive to elimination of nutrient point sources were Ray Hubbard, Lavon and Lewisville.

Results also highlight which subbasins and land uses within each watershed contribute more sediment and nutrients and should be targeted for the implementation of conservation practices.

However, future studies are needed to (1) fine-tune estimates on the impacts of numerous possible conservation practices that could be implemented to reduce reservoir sediment and nutrient loadings, (2) estimate the cost-effectiveness of these practices, (3) use this information to implement a watershed protection plan for each watershed, (4) provide this information to cities, counties, regional water and wastewater authorities, real estate developers and the public to improve stormwater ordinances and management, and (5) extend this kind of analysis to watersheds in other parts of Texas.

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Chapter 1: Introduction

The Trinity River Basin begins near the Red River, northwest of Fort Worth. It extends southeast, through the hearts of Fort Worth and Dallas, to the Gulf of Mexico just east of Houston. It is the most populated river basin in Texas and supports most of the primary water needs of both the Dallas-Fort Worth and Houston areas. The Upper Trinity Basin, which includes the Dallas-Fort Worth Metroplex, is home to over one-fourth of Texas' population.

Region C of the Texas water planning program is composed mainly of the Upper Trinity River Basin but also contains small parts of the Red, Brazos, Sulphur and Sabine river basins. The Region is heavily urbanized and is growing rapidly. In 2010, the population was approximately 6.3 million, a 25% increase from 2000. The population of Region C is projected to grow to 9.1 million in 2030 and 13.0 million in 2060. Dallas and Tarrant counties currently have 65% of the region's population with 81% located in cities with populations greater than 20,000. Water use in Region C has increased significantly in recent years, primarily in response to increasing population and municipal demand. Regional water use in 2006 was about 1.4 million acre-feet, about 90 percent of which was used for municipal supplies. Annual dry year demands are estimated to be 2.4 million acre-feet in 2030 and 3.3 million acre-feet in 2060. While normal year demands are expected to be 10 to 15 percent lower than these dry-year demands due to less outdoor irrigation.

Over 90 percent of the water use in Region C is supplied by surface water, with most of this coming from major reservoirs both within and outside the boundaries of Region C. In addition to surface water, groundwater can be an important source of supply in rural areas. Aquifers in the region include the Trinity (the largest groundwater source), Woodbine, Carrizo-Wilcox, Nacatoch and Queen City.

Cities and towns provide most of the retail water service in Region C. The three largest wholesale water providers are Dallas Water Utilities, Tarrant Regional Water District and North Texas Municipal Water District, which together provide about 85 percent of the water used in the Region. Wholesale water providers in Region C are pursuing several possible sources of future water supplies:

- Several new water supply reservoirs were proposed in the 2006 State Water Plan. However, legal, regulatory and economic issues make timely construction of these reservoirs uncertain.
- Because of aquifer characteristics, it is unlikely that groundwater will be able to supply a substantial portion of future demand.
- Over half of the water used for municipal supply in Region C is discharged as treated effluent from wastewater treatment plants. Wastewater reclamation and reuse is increasing rapidly in Region C, and reuse of treated wastewater will be a significant source of future water supplies for the region.
- Conservation is expected to be an important response to future water supply limitations.

In recent years, sediment surveys of Texas reservoirs as well as monitoring and modeling of sediment and nutrient loading have raised concerns that sediment loads are reducing the capacity of reservoirs while nutrient loading is reducing water quality in North Central Texas. Agricultural land use, urbanization and lack of management practices for controlling stormwater runoff, sediment yield and nutrient loading are suspected causes of the sediment and nutrient problems.

This study used the Soil and Water Assessment Tool (SWAT) to produce calibrated models of water, sediment and nutrients currently entering the twelve major water supply reservoirs of the Trinity River Basin in North Central Texas (Freestone County and above). These reservoirs drain a total area of about 7,289,609 acres in seven, eight-digit watersheds. In addition to this baseline estimate, four scenarios including: future urbanization, agricultural land management, reduction of point-source nutrients and removal of small, previously constructed flood control reservoirs (ponds) were simulated. The objective of these simulations was to gauge both the current rate of sediment and nutrient loading and possible impacts of future land use and management alternatives on watershed hydrology and sediment and nutrient transport to water supply reservoirs in North Central Texas. It should be noted that the varying degrees and access to data also determine the amount of work/detail done per watershed.

An Introduction to the Soil and Water Assessment Tool

SWAT is a basin-scale, distributed hydrologic and water quality model. Distributed hydrologic models allow basins to be subdivided into many smaller subbasins, helping the user incorporate spatial detail. The SWAT software system is designed to help scientists and decision makers manage soil and water resources at the watershed and river basin scales. A team of USDA-Agricultural Research Service, USDA-Natural Resources Conservation Service and Texas A&M University System engineers and scientists developed the model over the last 25 years. Over the last decade, the U.S. Environmental Protection Agency and a large number of engineers and scientists in the United States and around the world have become users and have contributed substantial resources to the model, its databases and interface development.

The SWAT system has been used successfully in many projects worldwide, which are documented in over 500 peer-reviewed scientific publications. Over 500 scientists and engineers have been trained in the use of the system, and more than 30 universities use the system in academic courses. Over the last decade, international and regional meetings of SWAT users and developers were held in the United States, Netherlands, Italy, Germany, China, Korea, Thailand, Chile, Portugal and Spain. SWAT training courses have been held in all of these and many other countries.

The SWAT system includes:

- Theoretical documentation describing the scientific basis for the tool,
- User guides to help train engineers and scientists in the use of the model,
- User interfaces to help users run the model and visualize its outputs,
- Critical input data to run the model in areas across the world for a wide variety of applications,
- The SWAT model itself.

SWAT system software, manuals and databases are part of the public domain and can be downloaded at <http://www.brc.tamus.edu/swat>. Information about SWAT meetings and training courses as well as an online database of relevant peer-reviewed literature can be found at the same Website.

The SWAT system is a multi-functional tool that can be used to answer a wide variety of questions about the structure, function and management of watersheds both large and small.

Hydrology. SWAT uses daily weather data to simulate surface and subsurface hydrology of a watershed. In addition, it is often used to estimate the impacts of soil and water conservation practices, urbanization, deforestation, reforestation, brush control and construction of ponds and reservoirs on streamflow, aquifer recharge and the frequency of droughts and floods.

Water Quality. The SWAT system is in use throughout the U.S. and in many countries worldwide to understand sources of water pollution and techniques for improving water quality in specific streams and reservoirs. Both point and nonpoint sources of pollution are considered.

Water quality indicators simulated include sediment, nutrients (nitrogen and phosphorus), chlorophyll, pesticides and bacteria. Among the most important uses are analyses required by the U.S. Clean Water Act (Total Maximum Daily Load assessments and Watershed Protection Plan development). The model has also been widely used to evaluate methods of reducing water pollution from dairies, feed lots and row crop agriculture.

Land Use. The SWAT system has often been used to estimate the effects of changing land use on the amount and quality of water in our streams and reservoirs. Specific land use changes that can be simulated include urbanization, increasing intensity of urban land uses, changing forest cover, brush control, conversion of forest and rangelands to agriculture (or the reverse), and construction of reservoirs.

Soil and Water Conservation. Federal and state agencies as well as local and regional water authorities promote good soil and water conservation practices to minimize soil erosion, river and reservoir sedimentation and water pollution. The SWAT system allows users to estimate the impacts of a wide variety of soil and water conservation practices, including small and large flood control reservoirs, urban stormwater management practices and agricultural soil conservation practices, which include crop rotations, filter strips, grassed waterways, cover crops, tillage practices, inorganic and organic fertilizers and soil amendments.

Climate Change. The SWAT system is not a global climate model and is not used to estimate the magnitude of climate change expected in the future. Instead, it is used to estimate the impacts of past or future climate change on the hydrology, water quality and agricultural production of watersheds. It is a very useful tool for estimating the effects of management practices or infrastructure (like ponds, reservoirs or irrigation systems) designed to adapt and mitigate the negative impacts of climate change.

SWAT is a continuous simulation model that operates on a daily time step and can be used for long-term simulations with generated or observed weather data. The SWAT model is also continually updated. New model versions are issued every few years and include new features and functionality. The version used in this study, SWAT 2005, is widely used both in the United States and internationally. Finally, SWAT 2005 is distributed with the full Formula Translator (FORTRAN) source code, allowing anyone to make modifications to the model. The newest version, SWAT 2009, was released after this study was nearing completion.

SWAT was created to overcome limitations of several previously developed models such as CREAMS, GLEAMS, EPIC, ROTO and SWRRB (table 1). By combining components of each of these models and developing new subroutines and functionality, SWAT allows the user to simultaneously simulate weather, hydrology, erosion-sedimentation, crop production, nutrient and pesticide transformations and movement, and bacterial contamination in small or large watersheds. For over a decade, SWAT has been used in numerous small and large-scale projects in the United States and internationally. For example, the HUMUS project (Hydrologic Unit Model for the United States; Srinivasan et al., 1998) used SWAT to model 350 USGS six-digit watersheds in 18 major river basins throughout the United States.

Table 1.1 SWAT arose from a combination of two applications: ROTO and the SWRRB, with CREAMS, GLEAMS and EPIC contributing to the development of SWRRB.

SWAT Development		
ROTO	Routing Outputs to Outlets	Arnold et al. (1995)
SWRRB	Simulator for Water Resources In Rural Basins	Williams et al. 1985)
CREAMS	Chemicals, Runoff and Erosion from Agricultural Management Systems	Knisel (1980)
GLEAMS	Groundwater Loading Effects on Agricultural Management Systems	Leonard et al. (1987)
EPIC	Erosion-Productivity and Impact Calculator	Williams (1990)

Chapter 2: Methodology

Project Objectives

Watershed management through modeling and water conservation education are two of the most cost-effective practices available for ensuring a safe and reliable public water supply. Therefore, the watershed modeling objectives of this project were to use the Soil and Water Assessment Tool (SWAT) model to:

- estimate the current rate of sedimentation and nutrient loading of ten major drinking water reservoirs in the Upper Trinity River Basin of North Central Texas and

- estimate the effects of expected urbanization and conservation practices including the effects of small flood control reservoirs, the reduction of point-source discharges and the reduction of livestock grazing rates on sediment and nutrient delivery to all ten reservoirs.

Project Design

We delineated the Trinity River Basin study area, from Freestone County and above, into ten watersheds, encompassing 12 major reservoirs (table 2.1 and figure 2.1). Using the general model input data described in the following section, SWAT simulated the production and transport of sediment and nutrients (nitrogen and phosphorous) from overland areas with varying soils and land uses. Overland areas refer to parts of the basin that are not part of the river channel or stream bank. Pollutants from land surfaces within these overland areas are transported into basin rivers and streams by runoff or irrigation and eventually end up in a reservoir. Therefore, SWAT calculated the sediment and nutrients resulting from overland areas as well as the amount actually received by each reservoir on an annual and monthly basis.

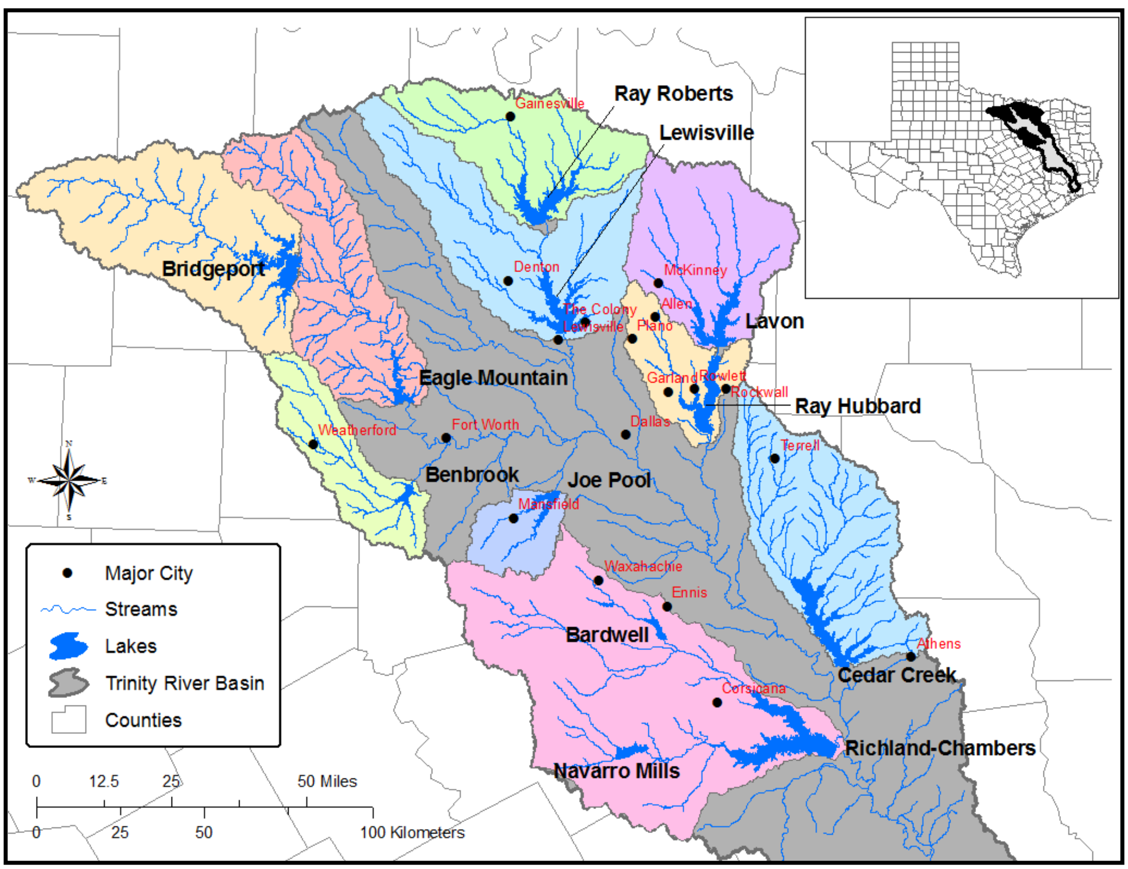


Figure 2.1 Locations of major reservoirs within the Trinity River Basin delineated for SWAT simulations.

Table 2.1 Major reservoirs within the Trinity River Basin study area.

Reservoir		Watershed area (acres)	Year completed
Lavon		492,481	1953
Lake Ray Hubbard		224,125	1969 (1996)
Ray Roberts		442,319	1986 (1990)
Lewisville		622,706	1954 (1979)
Joe Pool		143,321	1986 (1989)
Bridgeport		704,003	1931
Benbrook		271,816	1951
Eagle Mountain		551,045	1932
Cedar Creek		642,474	1966
Richland Chambers	Bardwell	108,726	1965
	Navarro Mills	200,402	1963
	Richland Chambers	1,274,322	1987

General Model Input Data

Although the following input data are mostly consistent between SWAT simulations, there are some variations due to applicability to particular basins and modeler preference, as four different modelers completed the chapters throughout this report. It should also be noted that varying degrees and access to data determined the amount of work done and detail included per watershed. The section below summarizes input data used to model the Upper Trinity River Basin study area.

Topography. A 30-meter (98-foot) Digital Elevation Model (DEM) from the National Elevation Dataset (NED) defined the topography (USGS, 1987) (table 2.2). Using the DEM's topographical information, we defined the stream network and its characteristics, such as channel slope, length and width. The resulting stream network was then used to define the layout, number of subbasins and their parameters, such as slope and slope length.

Soils. The soils database describes the surface and upper subsurface of a watershed. SWAT requires GIS soil data to define soil characteristics. SWAT uses this information to define each soil horizon (e.g., thickness, depth, texture, water holding capacity, etc.) and to determine daily runoff, erosion and an overall water budget for the soil profile. In most cases, SWAT utilized the Soil Survey Geographic (SSURGO) Database bundled with the ArcSWAT interface for this purpose. The USDA-NRCS Soil Data Mart, available online at http://soildatamart.nrcs.usda.gov, provided SSURGO GIS data (table 2.2). SSURGO is the most detailed soil database available. It describes each soil mapping unit as a single soil series. This 1:24,000-scale soils database is available as printed county soil surveys for over 90% of Texas counties. However, not all mapped counties are available in GIS format (vector or high resolution cell data). Therefore, in several cases, we used State Soil Geographic (STATSGO) data instead.

Land use. Land cover is perhaps the most important GIS data used in the model. The land cover theme affects the simulated amount and distribution of pasture, row crop and forest in the basin. These land covers differ radically in terms of erosion and nutrient losses. For example, forested areas contribute little to nutrient loading while pastures and row crops are thought to be primary sources of phosphorus. The Multi-Resolution Land Characteristics (MRLC) Consortium developed the National Land Cover Dataset (NLCD) from 1992 Landsat 5 Thematic Mapper (TM) satellite data (USGS, 2002). NLCD 2001 is an enhanced dataset from 1992 comprised of three elements: land cover, impervious surface and canopy density. NLCD data has a resolution of 30 meters (98 feet) and represents the first new land cover information since the 1970's. We used this dataset to provide land use inputs for SWAT. This data is available at www.mrlc.gov (table 2.2).

Weather. The National Climatic Data Center's National Weather Service stations located in and around each watershed provided daily precipitation and temperature (minimum and maximum) (table 2.2). This data is collected by professional NWS personnel and cooperators of the NWS (i.e., federal and state agencies), but the majority of the data are collected by private, unpaid volunteers. For this reason, these data are plagued with missing days, months or even years. Because the stations' periods of record are inconsistent, the number of active stations changed with time. To fix this problem, SWAT generates simulated weather when it detects missing data at a station. Thus, we replaced questionable or missing records with data from surrounding stations to provide a continuous daily record of rainfall and maximum and minimum temperature. Using its built-in weather data simulator, the SWAT model simulated other weather parameters too including wind speed, solar radiation and relative humidity. SWAT's weather generator (Nicks, 1974; Sharpley and Williams, 1990) uses monthly weather statistics from long-term, historical weather data. For example, mean daily wind speed is generated using average monthly wind speed and a random number between 0 and 1 in a modified exponential equation. Using a method developed for the EPIC model (Sharpley and Williams, 1990), daily average relative humidity values are calculated from a triangular distribution using average monthly relative humidity data.

Reservoirs. To model reservoirs, SWAT utilized available daily reservoir outflow data and reservoir characteristics from the National Inventory of Dams (table 2.2).

Table 2.2 Model input data type, scale and source for the Trinity River Basin.

Type	Scale	Source
Topography/DEM	1:24,000 (30-m resolution)	USGS National Elevation Dataset (NED) http://ned.usgs.gov
Land use/Land cover	1:24,000	USGS NLCD 2001 MRLC consortium www.mrlc.gov
Soils	1:24,000	SSURGO, USDA-NRCS Soil Data Mart http://soildatamart.nrcs.usda.gov
Dams[‡]		U.S. Army Corps of Engineers, National Inventory of Dams
Weather (precipitation and air temperature)		NOAA, NCDC http://www.ncdc.noaa.gov/oa/climate/stationlocator.html

[‡]The 12 major reservoirs were simulated as reservoirs, and the reaming small flood control structures were simulated as ponds.

Point Sources. Point sources were from the USGS Water Resource database, or EPA's Permit Compliance System identified actively operating point sources with measured discharge data. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1. Note that wastewater treatment plant (WWTP) discharge rates are permitted discharge rates, not the amount actually discharged. However, while data were available, the actual wastewater effluent concentrations and flows were used. See Cedar Creek and Eagle Mountain (table A-1). Permit limits, while available, were used for calculating nutrient loads for these point sources; otherwise nutrient loads were estimated using concentrations derived from a comprehensive survey of municipal wastewater dischargers in the Virginia portion of the Chesapeake Bay Basin (Wiedeman and Cosgrove, 1998), which are comparable to local available data. In SWAT, only one point source input file is required for each subbasin. Therefore, in subbasins with more than one discharger, total point-source output was combined into one SWAT input.

Subbasin Delineation. SWAT defined the subbasin layout using the DEM, a stream burn-in theme and a table of additional outlets. The stream burn-in theme helps SWAT define stream locations correctly in flat topography and consists of a database of digitized streams. Also, model predictions are only available at subbasin outlets, so additional outlets were added at points of interest, such as USGS stream gauges or water quality sampling sites. Finally, we used different stream threshold areas (the minimum contributing overland area required to define a single stream) to delineate subbasins within each watershed.

HRU Distribution. Hydrologic Response Units are the main building blocks of the SWAT model, dividing subbasins further into areas with similar slope, soil and land use data. A threshold, determined by the researcher, defines the minimum percent of land cover in a subbasin that will become an HRU. In most studies, though some differed, we used a minimum subbasin land use threshold of 3% and a minimum soil class threshold of 2%. These thresholds determined the number of HRUs in each of the subbasins under study.

Management. Telephone interviews with county conservation districts and Natural Resource Conservation Service personnel provided most of the management operations used in the model. These included various land use practices such as grazing rates, fertilization, tillage and more. Data gathered from interviews were representative of typical management operations in the area, although individual producers may utilize significantly different procedures.

Rangeland. Rangeland was considered unmanaged with a relatively light stocking rate of 1 animal unit per 20 acres all year long. In addition, SWAT did not simulate any fertilizer applications on rangeland.

Pasture. Pastureland contained Bermuda grass, fertilized and managed for hay production. SWAT used auto-fertilization with a nitrogen stress threshold value of 0.75. Also, SWAT default values dictated that a maximum of 178 pounds of nitrogen per acre could be applied at one time with up to 268 pounds of nitrogen per acre in any one year. Finally, the model simulated two hay cuttings during the modeled time period.

Cultivated Land. In the model, wheat graze-out with fall tillage comprised the cultivated areas. Management operations dictated field tillage in early September and nitrogen fertilizer applications of 50 pounds per acre before planting wheat on September 15. Also, grazing animals, represented by 1/3 animal unit per acre, grazed until the wheat was entirely consumed.

Forest Land. SWAT simulated forest as unmanaged and mixed, with primarily deciduous trees.

Urban Land. Within the urban land use category, a fraction of the total area was covered with impervious surfaces. SWAT used its default urban database to simulate different types of urban areas. For pervious surfaces (lawns), the model represented typical turf management, maintaining 1,339 pounds of biomass per acre with excess biomass converted to residue on a daily basis. This simulated mowing without the removal of grass clippings. Also, fertilization with 48 pounds of nitrogen per acre and 12 pounds of phosphorus (P_2O_5) per acre occurred early in the growing season.

Small Flood Control Reservoirs. Small flood control structures (PL-566 structures), referred to interchangeably throughout this report as ponds, affect hydrology by impounding water and trapping nutrients. Water in ponds is then subject to evaporation and seepage while nutrients and sediments settle to the pond floor. Most of the ponds throughout the 12 modeled basins were built between 1955 and 1990, therefore some are near the end of their design lives. In SWAT, the National Inventory of Dams defined pond characteristics and dictated the total number of these structures to be included in most basins. However in the Bridgeport and Benbrook basins, the National Land Cover Dataset (NLCD) defined pond characteristics because these basins included many small yet numerous ponds not included in the National Inventory of Dams. Using NLCD data, the drainage area of each pond was estimated by assuming a fixed surface to drainage area ratio of 25:1, and ponds were assumed to be 2 meters (6.6 feet) in depth and initially 75% full of water. SWAT allows for only one pond per subwatershed, so pond information for each subbasin, such as the surface area and storage at principal spillway, represents the sum of all ponds in the subbasin.

Channel Dimensions and Streambank Properties. Sediment is generated non-uniformly across the basin due to varying slopes, soil types and land uses. However, no data were available with which to gauge channel dimensions and streambank properties critical to sediment transport and streambank erosion. Therefore, the relative uncertainty in predictions regarding sediment loading of each reservoir is high due to the lack of data with which to calibrate channel degradation and deposition of sediments.

Model Calibration and Validation

After developing the model using the above inputs, we calibrated and validated the model, where data were available, to match streamflow and water quality conditions in each watershed. In each study, SWAT was calibrated and validated for flow using USGS gauging stations with available data located within or at the outlet of each watershed. When measured data were available, we also calibrated and validated SWAT outputs for water quality parameters, such as sediment, organic and mineral nitrogen, organic and mineral phosphorus, total nitrogen and total phosphorus. To assess how well SWAT predicted measured streamflow, sediment yields and nutrient loads, we used various statistical tools, including Nash-Sutcliffe efficiency and R^2 (see the model performance evaluation section below). While USGS gauging stations provided model data for calibration in most cases, we used other sources in some of the study basins with limited USGS data.

Calibration is the process by which a model is adjusted to make its predictions agree with observed data. Model calibration generally reduces uncertainty. Complex models often have many parameters, each with a range of values that may be equally valid. Careful selection of a single value within the appropriate range may improve model predictions. Furthermore, calibration requires observed data, which may not be available. In the absence of observed data, calibration is not an option. However, portions of a model may be calibrated while others may not. In each chapter, a list of calibrated parameters is given in table format (for example, table 3.2). If a parameter is not listed, default values were used. The parameters calibrated varied between basins due to basin characteristics and modeler preferences.

Validation is similar to calibration except adjustable model parameters are not modified. Validation improves the reliability of the model predictions by testing the model with observed data not used during calibration. The goal of validation is to determine whether the conceptual simulation model is an accurate representation of the system under study (Kleijnen, 1995). Validation is useful because models can be calibrated so that they seem to perform well during the calibration period by mimicking measured data, but the model may not properly represent fundamental processes important to the system. Validation challenges the model to replicate similar performance in a separate system or time. There is some confusion in the literature about the definition of validation and what it means to validate a model (Rykiel, 1996). For our purpose, model validation is the process during which model predictions are evaluated against measured data not used in calibration or model development. The purpose of validation is to provide an independent assessment of model performance.

Model Performance Evaluation

To evaluate how well the model represented actual conditions within the watershed, its performance was evaluated using qualitative and quantitative measures involving both graphical comparisons and statistical analyses. Mean, standard deviation, coefficient of determination (R^2) and Nash-Sutcliffe modeling efficiency (NSE) (Nash and Sutcliffe, 1970) are common statistical methods used to evaluate model predicted flow during calibration and validation. NSE describes the proportion of variance between the observed values and those accounted for by the model. It is calculated as:

$$NS_e = 1 - \left[\sum_{i=1}^{n} (P_i - O_i)^2 / \sum_{i=1}^{n} (O_i - \overline{O})^2 \right] \quad (1)$$

where P_i and O_i are predicted and observed values at each comparison point i; $\overline{O}$ is the arithmetic mean of the observed values, and n is the number of observations during the simulated period. Possible NSE values range from -∞ to 1.0 (1 inclusive). A value of 1 means that modeled results match perfectly with recorded data. There is no official performance rating for common watershed modeling statistics. However, most of the following chapters use statistics recommended by Moriasi et al. (2007) because they provide standardized model evaluation guidelines that support watershed modeling in CEAP-WAS. An NSE value greater than 0.75 can be considered very good; between 0.65 and 0.75 can be considered good while a value between 0.5 and 0.65 is considered only satisfactory (Moriasi et al., 2007).

Modeled Scenarios

After model development, calibration and validation, we used the baseline SWAT model to simulate several conservation practices and urbanization scenarios (described below) to evaluate their probable effects on sediment and nutrient loads received by the region's water supply reservoirs.

Ponds. Small flood control reservoirs were constructed in North Central Texas in the decades following World War II. These ponds trap sediment and nutrients, preventing their delivery to the reservoir. However, because many of these structures are reaching the end of their design life and some have been surrounded by urban development, many need to be renovated or removed. To simulate the effects of possible future removal of these structures on sediment and nutrient loading of water supply reservoirs, we modified the baseline model, which simulated all ponds, by removing these structures from the watersheds. We refer to this scenario as the removal of small flood control structures. Future studies could evaluate the effects of optimizing pond size and placement to minimize reservoir sediment loading.

Range Utilization. Rangeland is the major agricultural land use in North Central Texas. Overgrazing of rangelands can cause increased sediment and nutrient loading of streams. To simulate the possible effects of future agricultural programs designed to eliminate overgrazing, the baseline model was modified to remove all grazing activity.

Point Source Elimination. Wastewater treatment is expected to continue to improve in future decades, possibly greatly reducing point source nutrient loads discharged into North Central Texas streams. To simulate the maximum impacts of improved wastewater management on reservoir nutrient loading, all point-source discharges were eliminated from each baseline model. This scenario did not consider that the bioavailability of nutrients in point-source effluent may be much greater than that of other sources, particularly eroded materials. Therefore, it may underestimate the impact of point source reductions on reservoir water quality.

Urbanization. A great portion of the Trinity River Basin is expected to urbanize (figures 2.2 and 2.3). To accurately predict future expansion, the North Central Texas Council of Governments (NCTCOG) provided population projections by county for 2000 and 2030. Increases in urban land area in 2030 were taken uniformly from all non-urban categories in each subbasin to preserve the total subbasin area, and the percentages of land in all urban categories (low, medium and high density) remained the same. In other words, no attempt was made to reallocate the current distribution among these urban categories. The baseline model was then executed with the new land use data to simulate the effects of increased urbanization on sediment and nutrient loading of water supply reservoirs. These results were summarized at the subbasin level for mapping and at the entire basin for reporting.

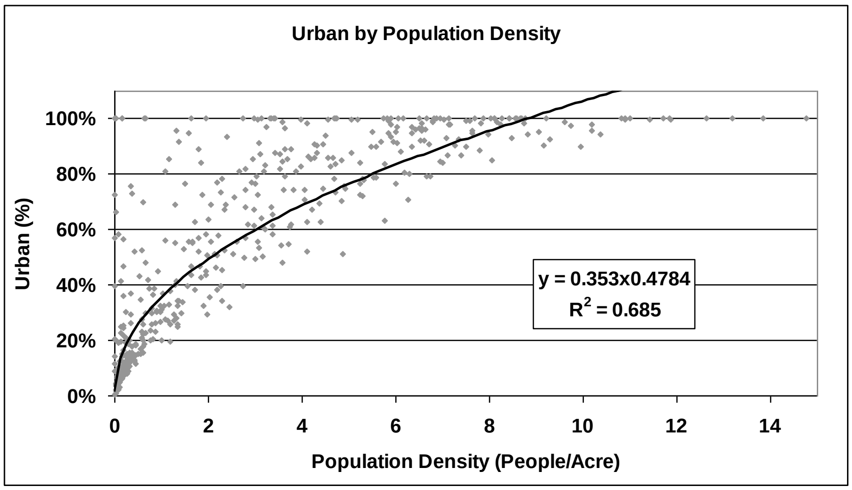


Figure 2.2 Relationship between the fraction of counties that is urban and population density, as derived from North Central Texas Council of Governments population data for 2000 as well as 2001 National Land Cover Data.

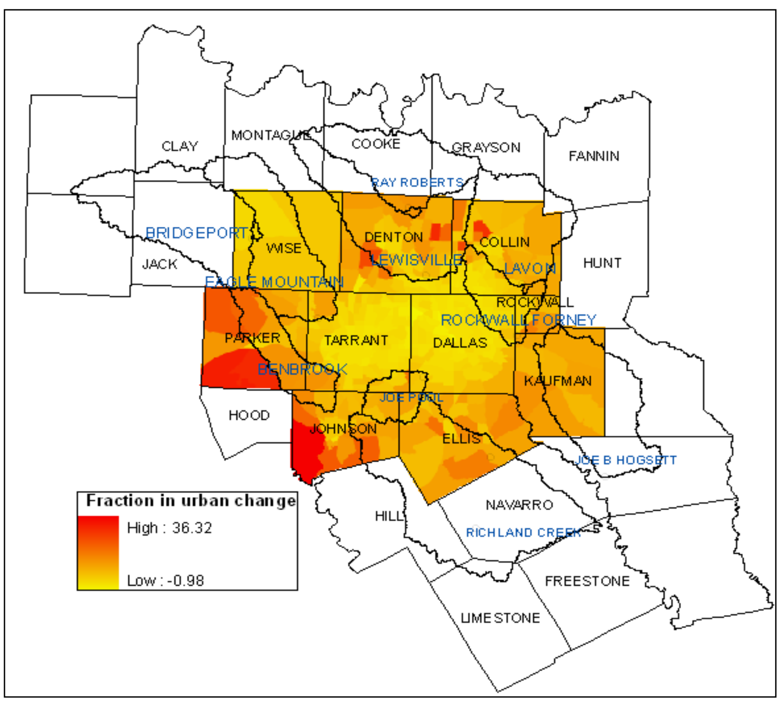


Figure 2.3 Fraction of urban change based on population projections for 2030. Urban changes in areas outside of the available projected area are estimated based on average urban change in each specific watershed (see maps in individual chapters).

In the following chapters, the results of each study are presented following the general methodology outlined above. However, keep in mind several model limitations.

Model Limitations

Hydrologic models are often used to provide decision makers with guidance in managing water quality issues. The uncertainty associated with model predictions is often underestimated or ignored completely. In this project, we made no attempt to quantify the uncertainty, and the vast majority of modeling efforts do not. It is not possible to calculate the true uncertainty associated with our predictions given the available data and current generation of models.

Limitations may be the result of data used in the model, inadequacies in the model or using the model to simulate situations for which it was not designed. Hydrologic models will always have limitations because the science behind the model is not perfect or complete. A model, by definition, is a simplification of the real world. Thus, understanding the limitations helps assure that accurate inferences are drawn from model predictions.

As such, there are several limitations that should be noted. The greatest limitation of these simulations is the lack of data with which to calibrate the model. Given this restriction, relative comparisons are the best use of these results (i.e., percent changes) as opposed to absolute model predictions (for instance, tons of sediment per year). The primary limitation of the data is uncertainty due to management at the field scale. Management practices differ between fields. Therefore, the exact regime of each is unknown. A single set of reasonable management operations was applied uniformly, but poor management may dramatically elevate sediment and nutrient loads from a particular area. The model does not account for these specific effects.

Scenarios involving greater departure from calibration conditions result in greater uncertainty. Although calibration assures the user that the results reflect the range of conditions encountered in the watershed, it does not assure that the model will be accurate in predicting the changes that result from drastic conversions in land use or management scenarios.

References

See the appendix.

Chapter 3: Joe Pool Basin

Introduction

The Joe Pool Basin encompasses an area of 143,321 acres in Texas' Trinity River Basin, covering parts of Dallas, Tarrant, Johnson and Ellis counties (figure 3.1). Joe Pool Basin contains one reservoir, Joe Pool Lake. The watershed modeling objective of this project was to use the Soil and Water Assessment Tool (SWAT) model to evaluate the effects of urbanization and other land use changes on sediment and nutrient delivery to Joe Pool Lake within the Trinity River Basin.

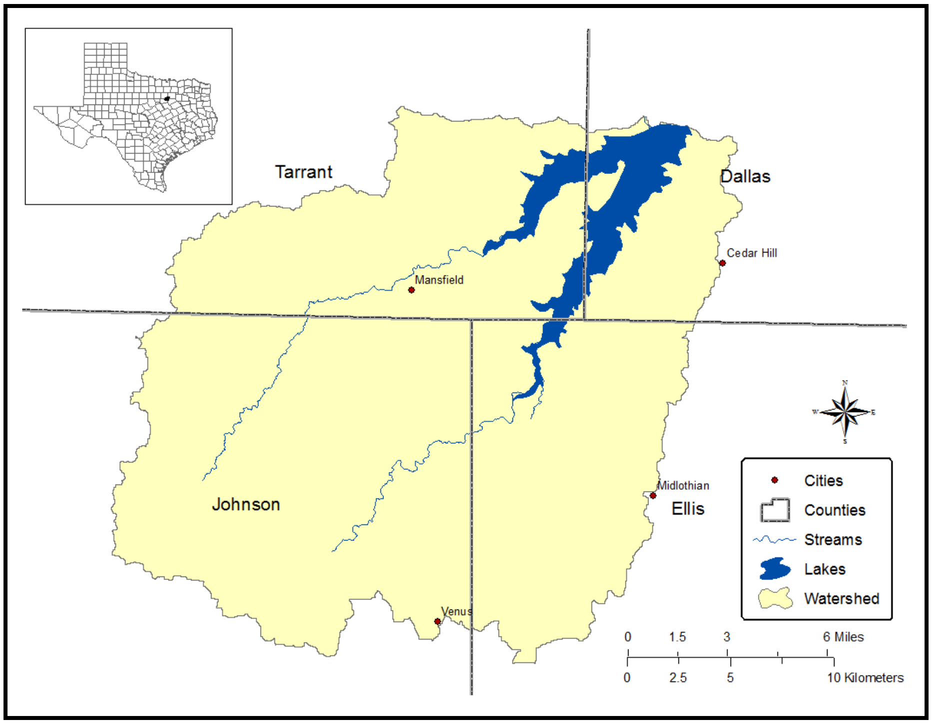


Figure 3.1 Location of Joe Pool Basin.

Model Input Data Tables and Figures

Topography

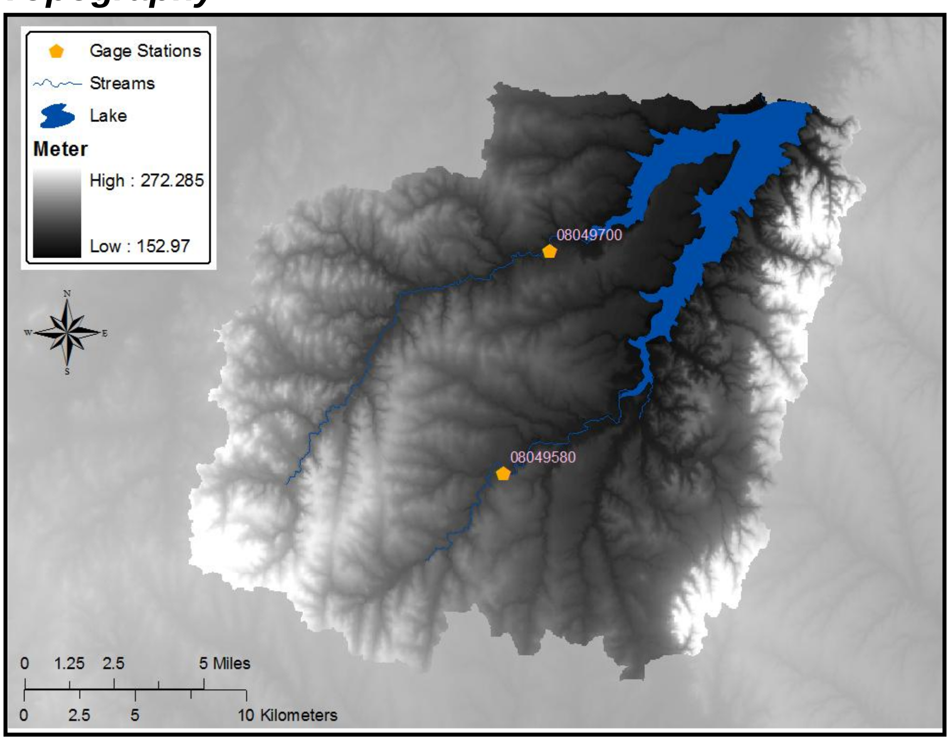


Figure 3.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of Joe Pool Basin.

Soils

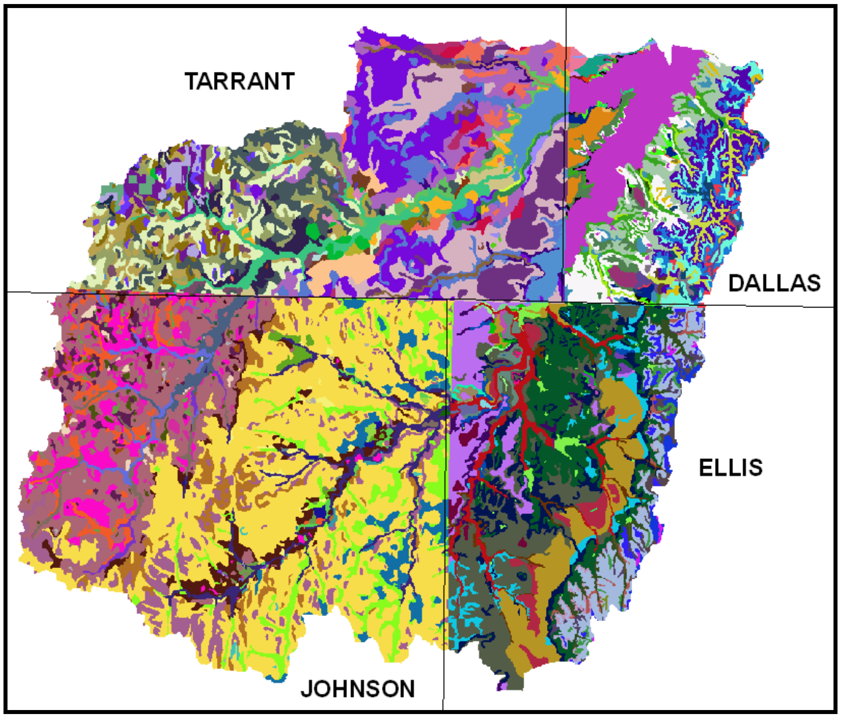


Figure 3.3 Soil Survey Geographic (SSURGO) data was used to define soil attributes in SWAT.

Land Use

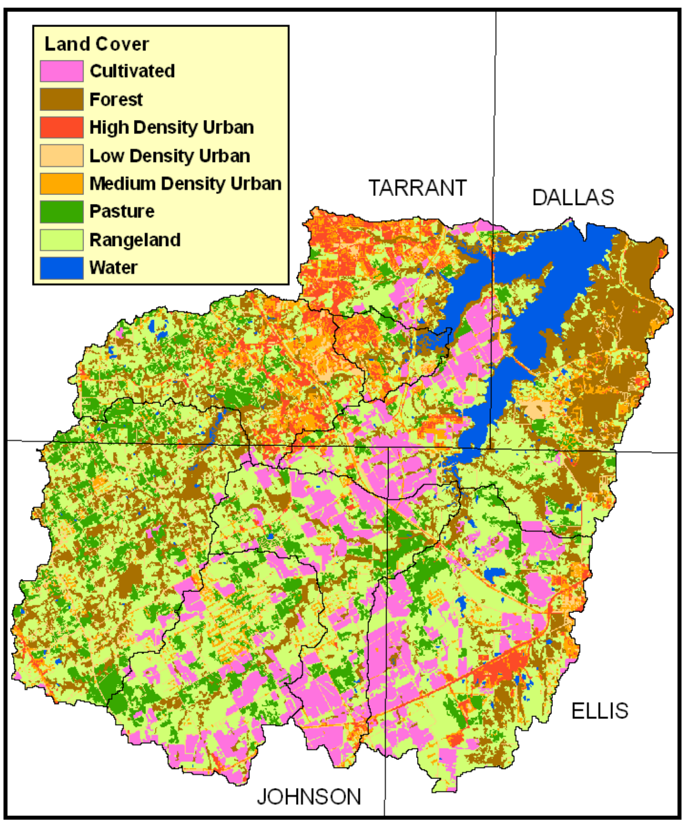


Figure 3.4 National Land Cover Data (2001) defined land use within Joe Pool Basin.

Weather

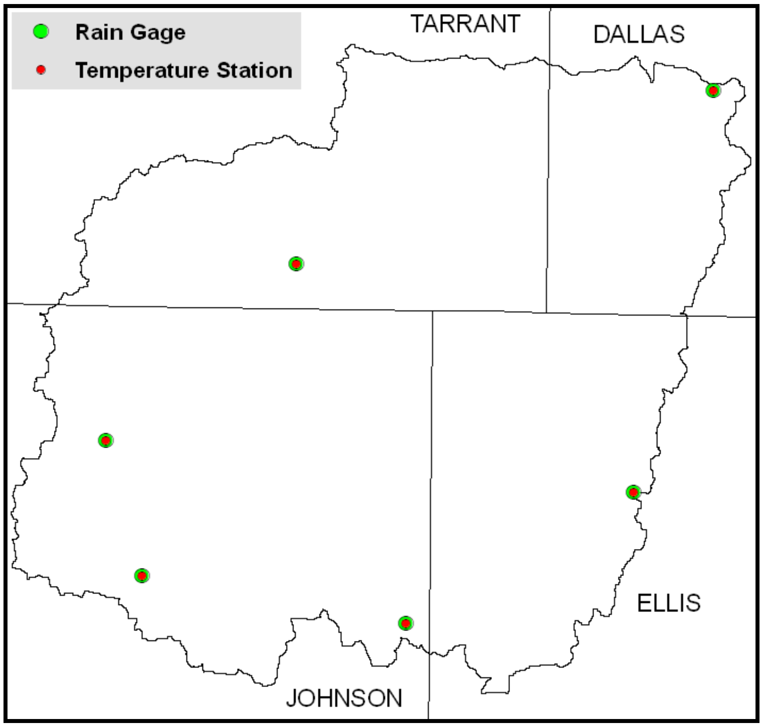


Figure 3.5 Six precipitation and temperature stations, located in and around the watershed, provided daily precipitation and temperature (minimum and maximum) data.

Point Sources

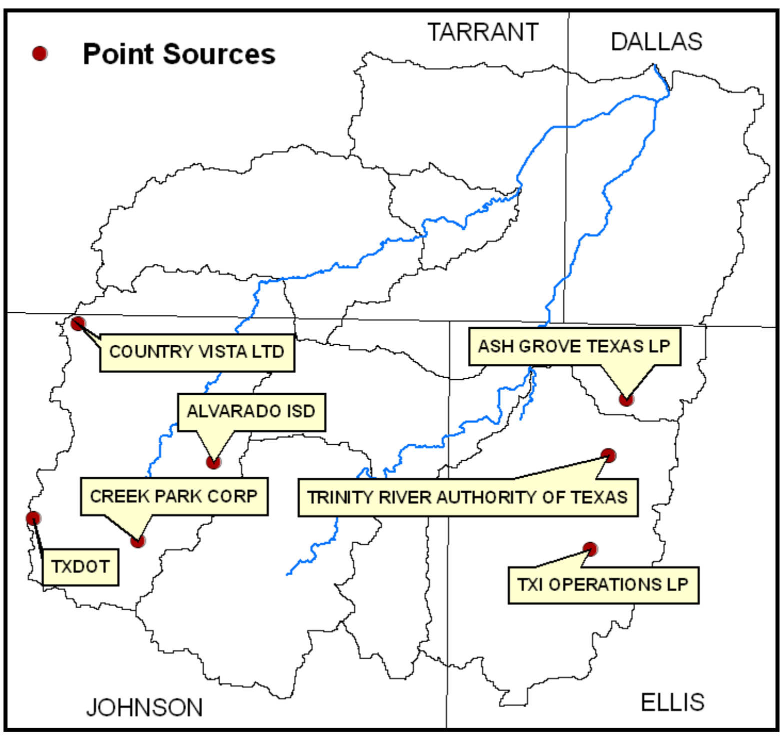


Figure 3.6 The USGS Water Resource Database identified seven point sources operating in the Joe Pool Basin with available discharge data. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

Reservoir

Table 3.1 The National Inventory of Dams provided reservoir characteristics for Joe Pool Lake.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Joe Pool	1	7,470	17.7	19,534	46.3	Measured

Subbasin Delineation

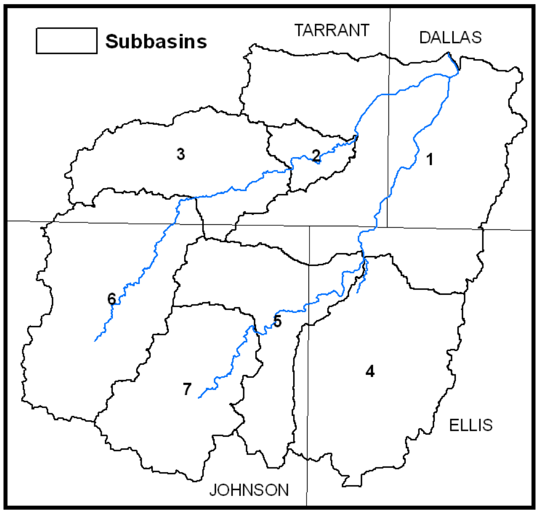


Figure 3.7 For this study, we used a stream threshold value of 7,018 acres to delineate subbasins, resulting in seven subbasins.

HRU Distribution

We used the thresholds outlined in Chapter 2 for determining HRUs, splitting the seven Joe Pool subbasins into 461 HRUs.

Ponds

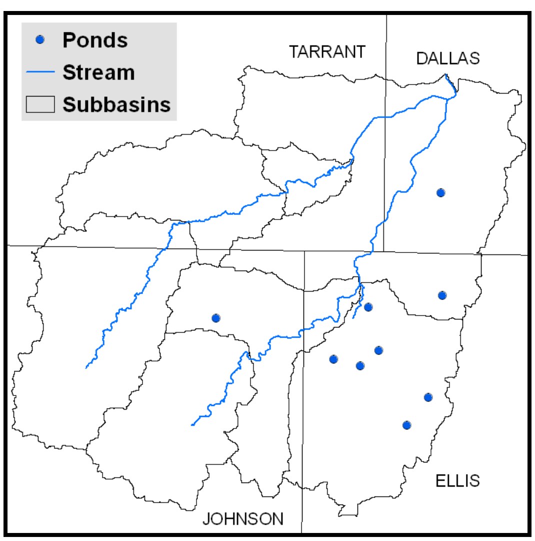


Figure 3.8 The U.S. Army Corps of Engineers' National Inventory of Dams provided the location of ponds in the Joe Pool Basin. According to the SWAT model, approximately 12% of the Joe Pool Basin area drains into ponds.

Model Calibration and Validation

Model simulation began in 1960. To determine the appropriate baseflow proportion, streamflow records from the USGS gauging station (08049700) at Walnut Creek near Mansfield (figure 3.9) were analyzed using the Baseflow Filter Program (Arnold and Allen, 1999; Arnold et al., 1995). We calibrated annual and monthly streamflow for the period lasting from 1965 to 1986 using flow records from USGS gauging station 08049700.

The model was validated for flow using the same gauging station (08049700) from 1987 through 2007. Validation was also conducted for flow using the USGS gauging station at Mountain Creek near Venus (08049580) from 2002 through 2007.

Data available from USGS gauging station 08049700 also allowed for both water quality calibration and validation. Calibrated parameters are listed in table 3.2.

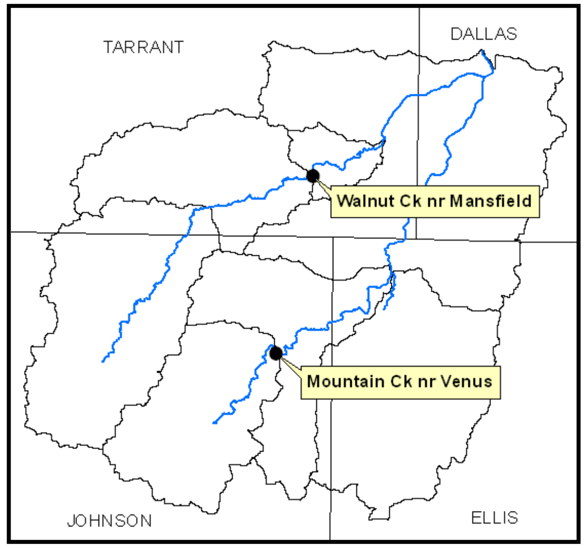


Figure 3.9 USGS gauges used for calibration and validation in the Joe Pool Basin.

Table 3.2 Model parameters and ranges used for calibration and final calibrated values. The model used default values for uncalibrated parameters, some of which are used in other chapters within this report.

Component	Parameter (file)	Description	Range	Calibrated value
Flow	CN2 (.mgt)	Initial NRCS runoff curve number for moisture condition II	-5–5	Default + 5.0
	ESCO (.bsn)	Soil evaporation compensation factor	0.01–1.0	0.95
	EPCO (.bsn)	Plant uptake compensation factor	0.01–1.0	1.0
	GW_REVAP (.gw)	Groundwater revap coefficient	0.02–0.2	0.08
	GWQMN (.gw)	Threshold depth of water in the shallow aquifer required for return flow to occur	0.0–300.0	20
	REVAPMN (.gw)	Threshold depth of water in the shallow aquifer for "revap" to occur [mm]	1.0–15.0	1.0
Sediment	SPCON (.bsn)	Linear parameter for estimating maximum amount of sediment that can be reentrained during channel sediment routing	0.0001–0.01	0.0015
	CH_COV (.rte)	Channel cover factor	0.0–1.0	0.2
	CH_EROD (.rte)	Channel erodibility factor	0.0 –1.0	0.02
Nutrients	RHOQ (.wwq)	Algal respiration rate at 20ºC (day^{-1})	0.05–0.50	0.33
Mineral nitrogen	SDNCO (.bsn)	Denitrification threshold water content (fraction of field capacity water content above which denitrification takes place)		1.1
	NPERCO (.bsn)	Nitrate percolation coefficient	0.01–1.0	0.01
Nitrogen in reach	AI1 (.wwq)	Fraction of algal biomass that is nitrogen	0.07–0.09	0.08
	RS4 (.swq)	Rate coefficient for organic N settling in the reach at 20ºC (day^{-1})	0.001–2.5	2.5
	BC3 (.swq)	Rate constant for hydrolysis of organic N to NH_4 in the reach at 20ºC (day^{-1})	0.2–1.0	1.0
Mineral phosphorus	PPERCO (.bsn)	Phosphorus percolation coefficient	10.0–17.5	17.5
	PHOSKD (.bsn)	Phosphorus soil partitioning coefficient	100–175	100

Phosphorus in reach	AI2 (.wwq)	Fraction of algal biomass that is phosphorus	0.01–0.02	0.02
	BC4 (.swq)	Rate constant for mineralization of organic P to dissolved P in the reach at 20ºC (day^{-1})	0.01–0.70	0.7
	RS5 (.swq)	Organic phosphorus settling rate in the reach at 20ºC (day^{-1})	0.001–2.5	2.5

Results and Discussion

Model Calibration and Validation

Flow

Calibration of streamflow (1965–1986) at the USGS gauging station at Walnut Creek near Mansfield (figure 3.10) resulted in annual R^2 and NSE values of 0.76 and 0.73, respectively, while a monthly comparison produced R^2 and NSE values of 0.74 and 0.73, respectively (table 3.3). Validation of annual and monthly streamflow for this gauging station (1987–2007) and for the gauging station at Mountain Creek near Venus (2002–2007) resulted in R^2 and NSE values ranging from 0.70 to 0.93 (table 3.3). Simulated flow for both the calibration and validation periods showed trends similar to the corresponding observed data (figures 3.10-3.13).

Table 3.3 Annual and monthly streamflow (m^3/s) calibration and validation results at USGS gauging stations.

Gauge station ID (site name)		Period	Measured			Simulated			Yearly		Monthly	
			Mean	Std_y	Std_m	Mean	Std_y	Std_m	R^2	NSE	R^2	NSE
08049700 (Walnut Creek near Mansfield)	Calibration	1965–1986	0.47	0.33	1.07	0.48	0.27	0.85	0.76	0.75	0.73	0.72
	Validation	1987–2007	0.89	0.58	1.57	0.81	0.55	1.49	0.74	0.71	0.74	0.73
08049580 (Mountain Creek near Venus)	Validation	2002–2007	0.34	0.22	0.50	0.35	0.30	0.50	0.90	0.72	0.72	0.70

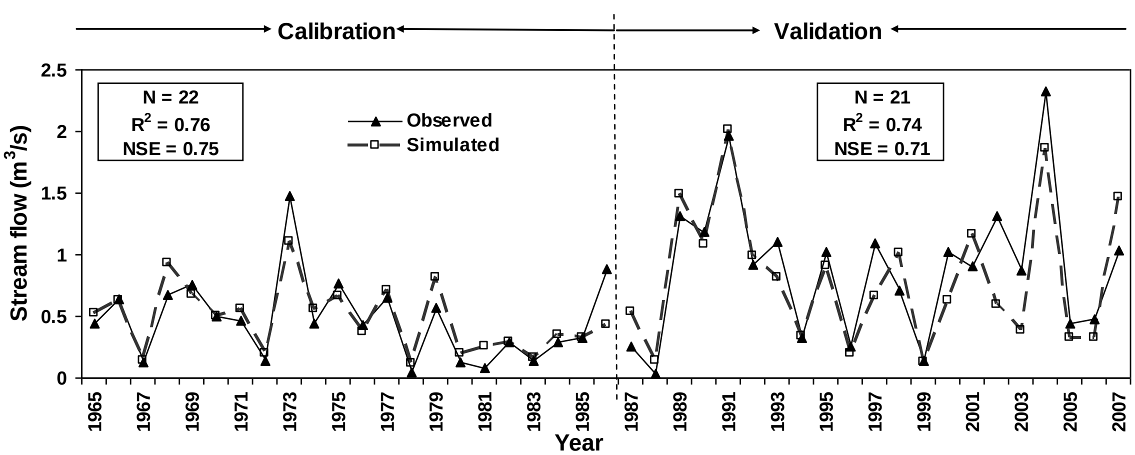


Figure 3.10
Measured and simulated yearly streamflow at USGS gauge 08049700 (Walnut Creek near Mansfield) for calibration (1965–1986) and validation periods (1987–2007).

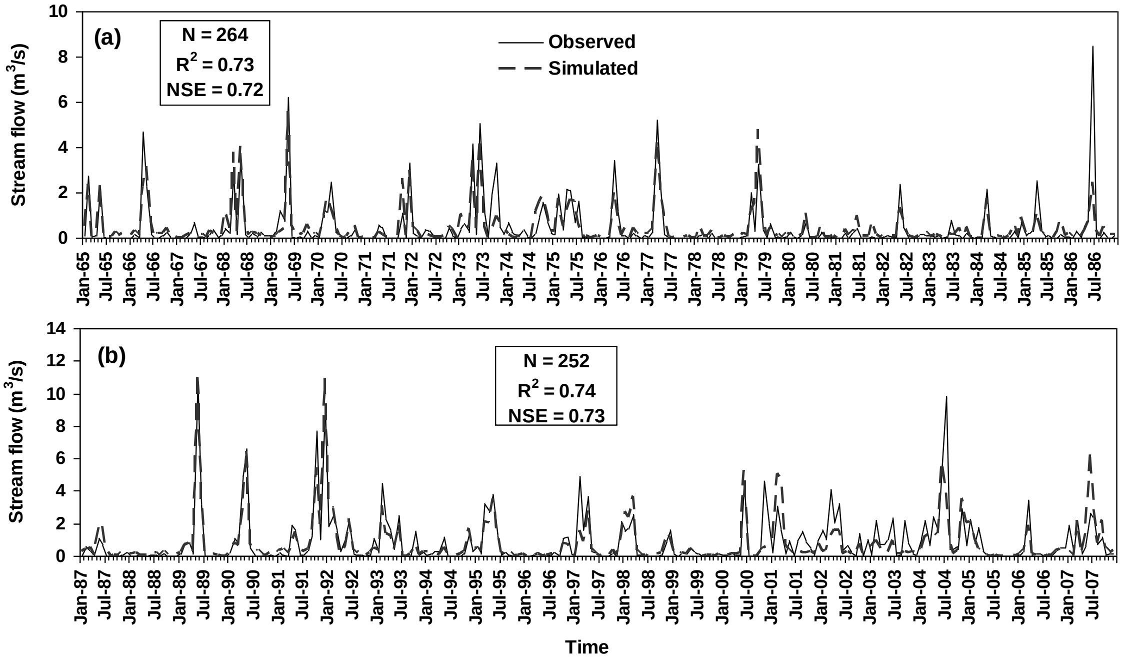


Figure 3.11
Measured and simulated monthly streamflow at USGS gauge 08049700 (Walnut Creek near Mansfield) for the (a) calibration period (1965–1986) and (b) validation period (1987–2007).

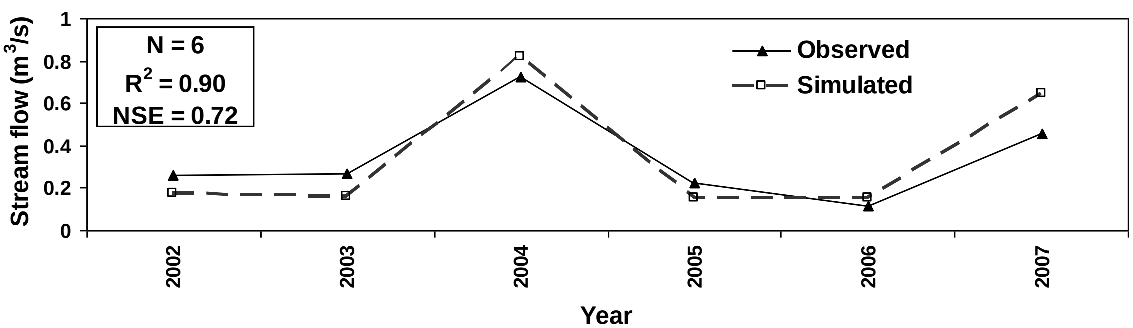


Figure 3.12
Validation of yearly streamflow at USGS gauge 08049580 (Mountain Creek near Venus).

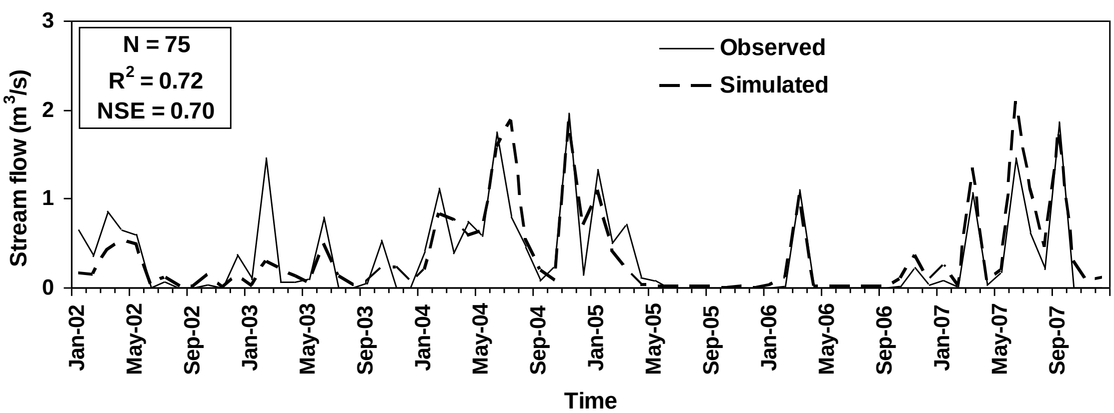


Figure 3.13
Validation of monthly streamflow at USGS gauge 08049580 (Mountain Creek near Venus) for 1/2002–12/2007.

Water Quality

Using USGS gauging station 08049700, we calibrated and validated average daily loads for days with available data. During the calibration period, simulated total phosphorus, nitrate-nitrite (NO_x), organic nitrogen and total nitrogen loads compared well with their corresponding measured values, and percent error was within ±10% (table 3.4 and figure 3.14). For the validation period, the average daily load of nitrate-nitrite was under-predicted.

Table 3.4 Water quality calibration and validation for days with available observations at USGS gauging station 08049700. (Average daily nutrient loads were calculated as the daily nutrient concentration multiplied by the corresponding daily flow volume.)

	Total P		NO_3-NO_2		Organic N		Total N	
	Observed	Simulated	Observed	Simulated	Observed	Simulated	Observed	Simulated
Calibration								
Mean in kg/day (lb/day)	6.96 (15.3)	7.34 (16.2)	17.38 (38.3)	17.88 (39.4)	29.45 (64.9)	28.29 (62.4)	94.00 (207.3)	85.03 (187.5)
% error		5.4		2.9		-4.0		-9.5
Validation								
Mean in kg/day (lb/day)	13.76 (30.3)	15.16 (33.4)	25.52 (56.2)	19.50 (43.0)	45.83 (101.1)	50.01 (110.3)	106.66 (235.2)	109.99 (242.5)
% error		10.2		-23.6		9.1		3.0

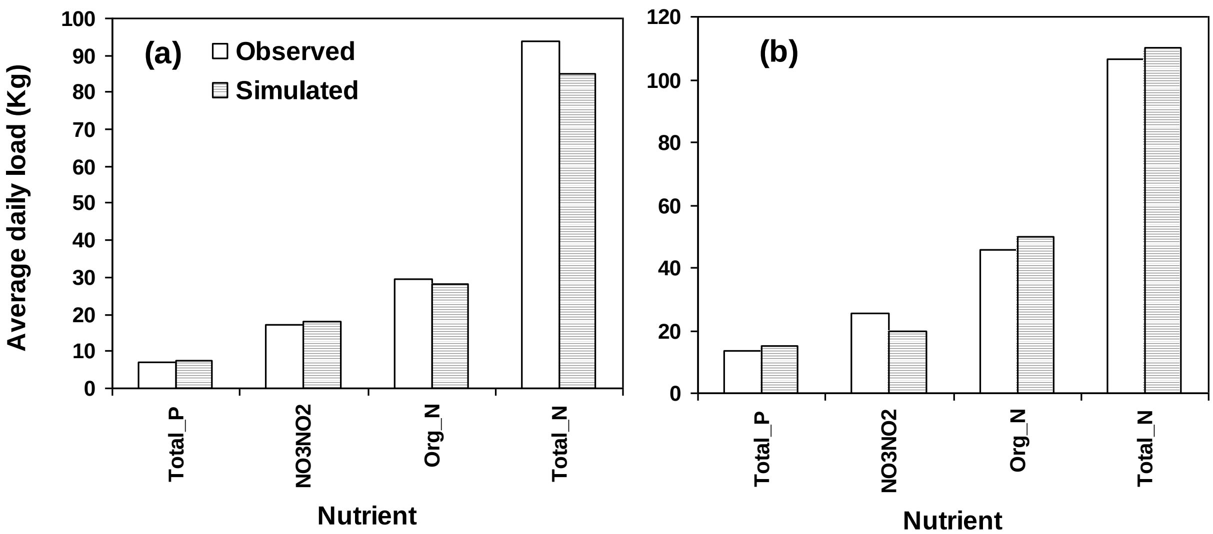


Figure 3.14 (a) Calibration and (b) validation of simulated average daily nutrient loads for days with available observations at USGS gauge 08049700 (Walnut Creek near Mansfield). Average daily nutrient loads were calculated as the daily nutrient concentration multiplied by the corresponding daily flow volume. These observed average daily nutrient loads were compared with corresponding average daily simulated values for both the calibration and validation periods.

Model Predictions

During the simulation period (1968–2007), the model predicted that Joe Pool Basin generated an average of 32,233 metric tons (35,531 tons) of sediment per year from overland flow under current management scenarios. Joe Pool Reservoir began operating in 1986. From this starting period until 2007, the model-predicted that the average annual sediment load generated from overland flow was 39,389 metric tons (43,419 tons) per year. While the modeled average annual sediment load delivered to Joe Pool Reservoir was about 42,368 metric tons (46,703 tons) per year, suggesting some sediment is generated by the river channels. Sediment is produced non-uniformly throughout the basin, as shown in figure 3.15.

During the simulation time period (1986–2007), the model predicted that overland flow generated phosphorus and nitrogen loads of 59,488 kilograms (131,149 pounds) per year and 129,778 kilograms (286,116 pounds) per year, respectively, under current management scenarios. Total phosphorus and nitrogen losses from overland flow are mapped in figures 3.16 and 3.17, respectively. Point sources generated 812 kilograms (1,790 pounds) of phosphorus per year and 9,863 kilograms (21,744 pounds) of nitrogen per year. The model predicted that on average 31,339 kilograms (69,091 pounds) of total phosphorus and 129,885 kilograms (286,347 pounds) of total nitrogen reached Joe Pool Reservoir annually from 1986–2007.

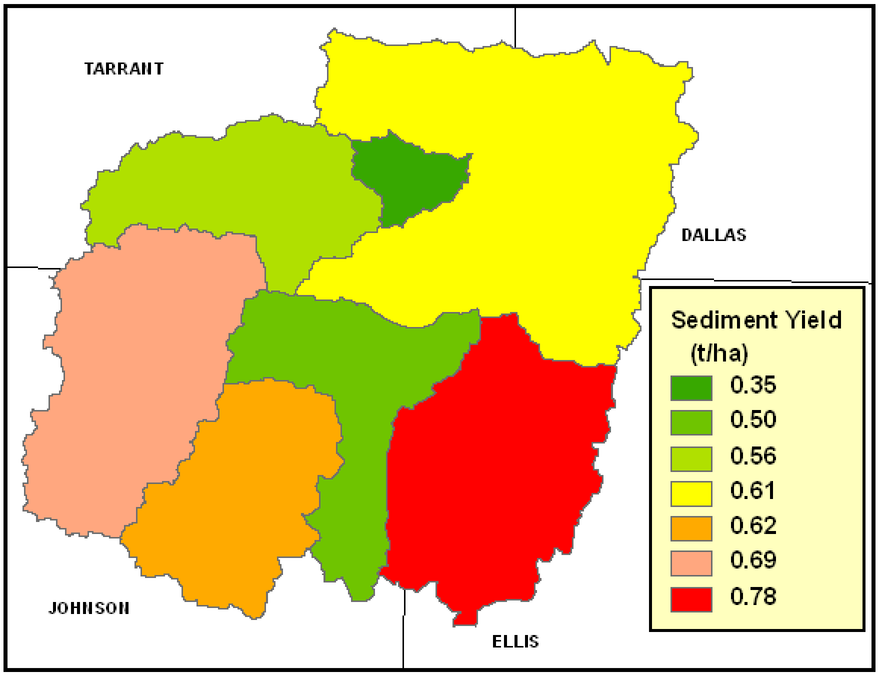


Figure 3.15 Sediment losses from Joe Pool Basin as predicted by the SWAT model.

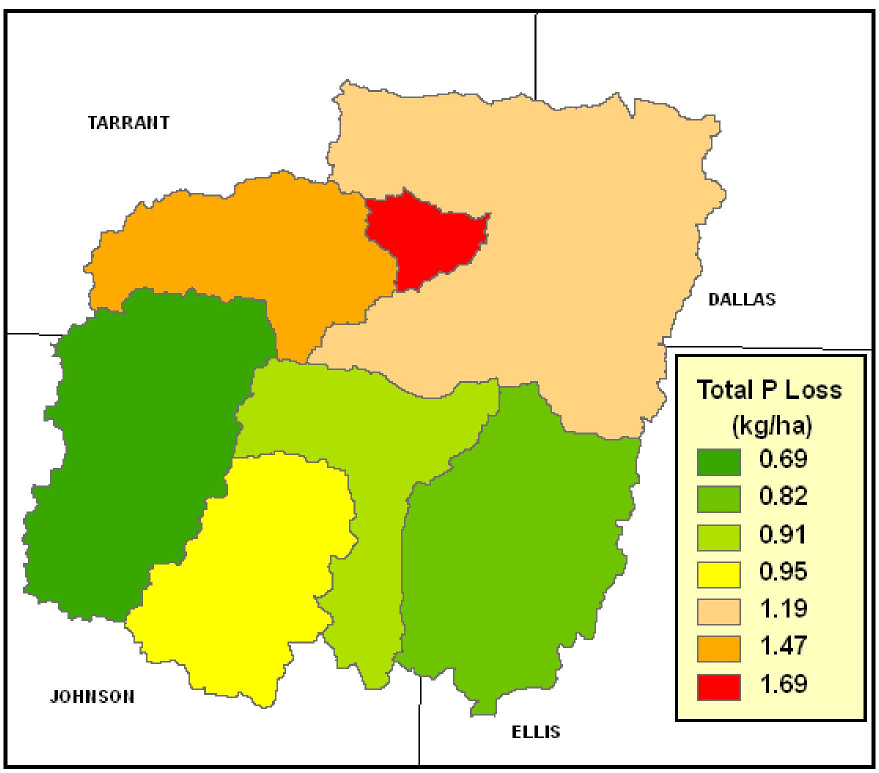


Figure 3.16 Phosphorus losses from Joe Pool Basin as predicted by the SWAT model.

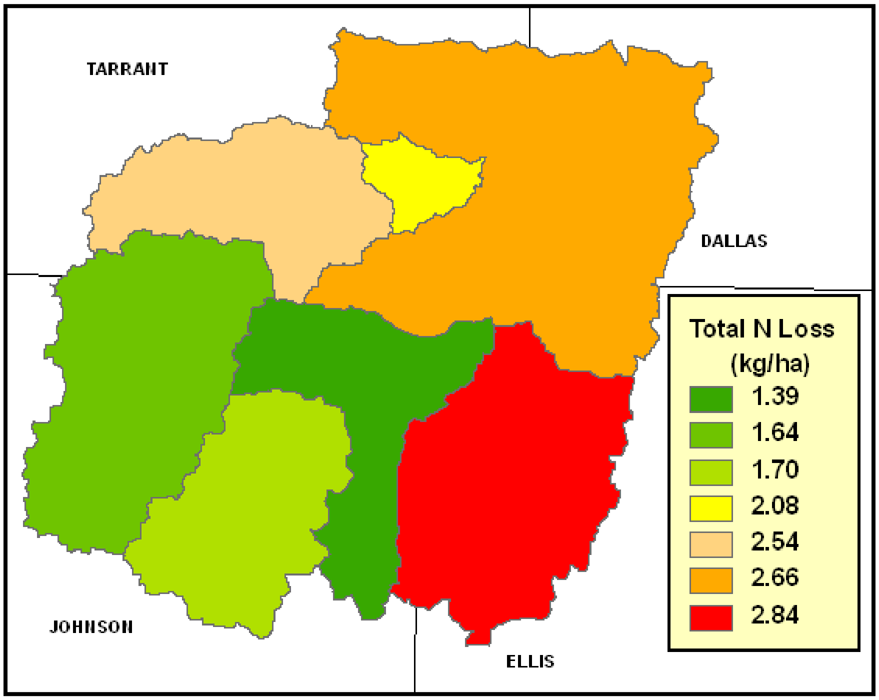


Figure 3.17 Total nitrogen losses from the Joe Pool Basin as predicted by the SWAT model.

Scenarios

Conservation Practices

SWAT simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted results for each scenario are given in table 3.5.

Ponds

Pond removal caused increases in sediment and nutrient loading (figure 3.18). By removing ponds, SWAT determined that the nine simulated ponds, modeled based on the National Inventory of Dams' PL-566 structures (see Chapter 2, General Model Input Data), reduce Joe Pool Reservoir's received sediment load by 10% (table 3.5).

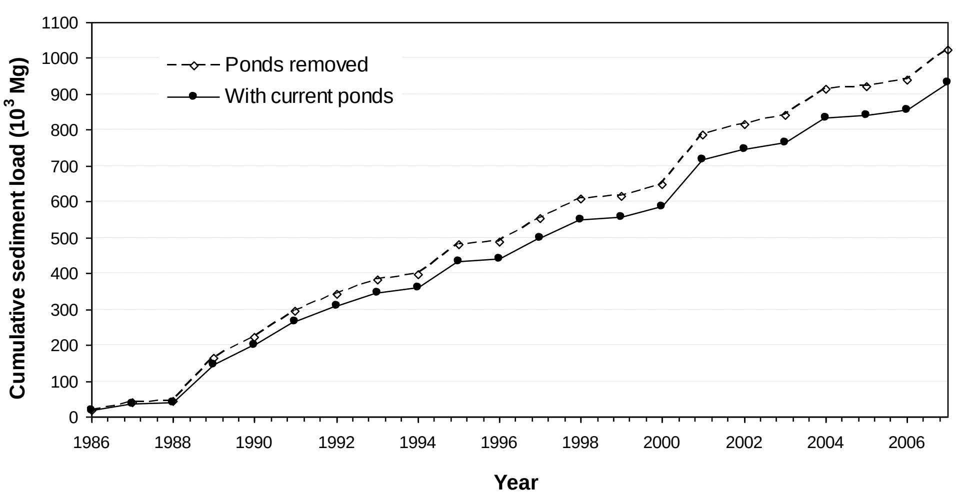


Figure 3.18 Cumulative sediment loads received by the Joe Pool Reservoir as predicted by the SWAT model.

Range Utilization

Removing grazing activity and all rangeland management resulted in an 8% reduction in the sediment load received by the Joe Pool Reservoir (table 3.5).

Point Source Load Elimination

The basin contains seven point sources. Eliminating these WWTPs reduced nutrient loading to Joe Pool Reservoir. Total phosphorus loads decreased by 1% and total nitrogen loads by 8%.

Urbanization

Overall, urban land use within the Joe Pool Basin is projected to increase from 24,711 acres (17% of the total watershed area) in 2000 to 43,243 acres (30% of the total watershed area) in 2030 based on North Central Texas Council of Governments' population projections and a relationship developed between population density and the fraction of urban area within each district (see Chapter 2, Modeled Scenarios). The increase in urban area was summarized at the subbasin level as fraction of urban change (figure 3.19). The average fraction of urban change in this watershed is 1.75 (24,711 acres of urban area in 2000 * 1.75 = 43,243 acres of urban area in 2030). This urban expansion caused total phosphorus and total nitrogen loads from within the basin to increase by 37% and 16%, respectively.

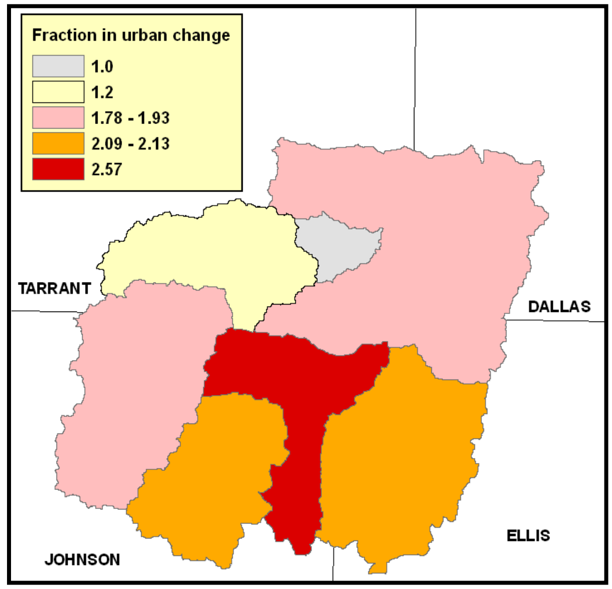


Figure 3.19 Projected changes in urban area from 2000 to 2030 (urban area in 2030 = urban area in 2000 * fraction of urban change).

Table 3.5 Changes in sediment and nutrient loads received by the Joe Pool Reservoir under different scenarios as derived from SWAT model simulations (1986–2007).

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline (with ponds)	42,368 metric tons/yr (46,702 tons/yr)	31,339 kg/yr (69,090 lb/yr)	129,885 kg/yr (286,347 lb/yr)
No Ponds	9.9%	5.8%	4.4%
No Range Grazing	-7.7%	-2.8%	-2.0%
Urban *	-3.6%	37.4%	16.4%
No Point Sources	-4.4%	-1.0%	-7.3%

* load changes from overland only

Conclusions

We used the SWAT model to simulate the effects of urbanization and other land use changes on hydrologic and water quality processes in the Joe Pool Basin. Using flow data available in the watershed, SWAT calibration and validation resulted in a modeled representation of the watershed that was within acceptable standards. For example, NSE values based on monthly flow comparisons ranged from 0.70 to 0.73 and from 0.71 to 0.75 for yearly flow comparisons. The model was also calibrated and validated for nutrient loads based on available average daily data. Simulated total phosphorus, organic nitrogen and total nitrogen loads compared well with corresponding measured values, with a percent error of about ±10%. Nitrate-nitrite was under-predicted during the validation period with a percent error of -24%. With recognized uncertainty in measured data, Harmel et al. (2006) indicate that model results within 10 to 31% of measured values are within the average uncertainty range of water quality data measured with a typical quality assurance/quality control effort.

During the simulation period ranging from 1986–2007, the model predicted that the total sediment reaching Joe Pool Reservoir was about 42,368 metric tons (46,702 tons) per year with channel banks contributing about 7% of the sediment load. The model predicted that 31,339 kilograms (69,090 pounds) of total phosphorus and 129,885 kilograms (286,347 pounds) of total nitrogen reached Joe Pool Reservoir annually.

In the scenario analyses, the complete removal of ponds, point sources and rangeland management demonstrated their overall contribution to the water quality of the Joe Pool Reservoir. Based on model predicted values, the removal of ponds would increase the sediment load received by Joe Pool Reservoir by 10% while total phosphorus and total nitrogen would increase by approximately 6% and 4%, respectively. Removing point sources achieved reductions of approximately 1% in total phosphorus and 7% in total nitrogen. Finally, removing grazing from rangelands reduced the sediment load by approximately 8%, total phosphorus by 3% and total nitrogen by 2%. SWAT also predicted the effects of urban expansion in 2030. In the urbanization projections, urban areas are expected to increase from 17% in 2000 to 30% in 2030. Associated with this expansion, the model predicts increases of approximately 37% in total phosphorus and 16% in total nitrogen from overland areas.

References

See the appendix.

Chapter 4: Lavon Basin

Introduction

Lake Lavon was completed in 1954. It provides flood control and municipal water supplies to Collin, Dallas and Rockwall counties. It also offers some of the best fishing, camping and boating in Texas. The Lavon Basin, which includes Lake Lavon, is found within USGS Hydrologic Unit Code 12030106 and encompasses an area of 477,655 acres within the larger Trinity River Basin in Texas. The Lavon Basin covers parts of Grayson, Fannin, Collin and Hunt counties (figure 4.1) and originates in Grayson County.

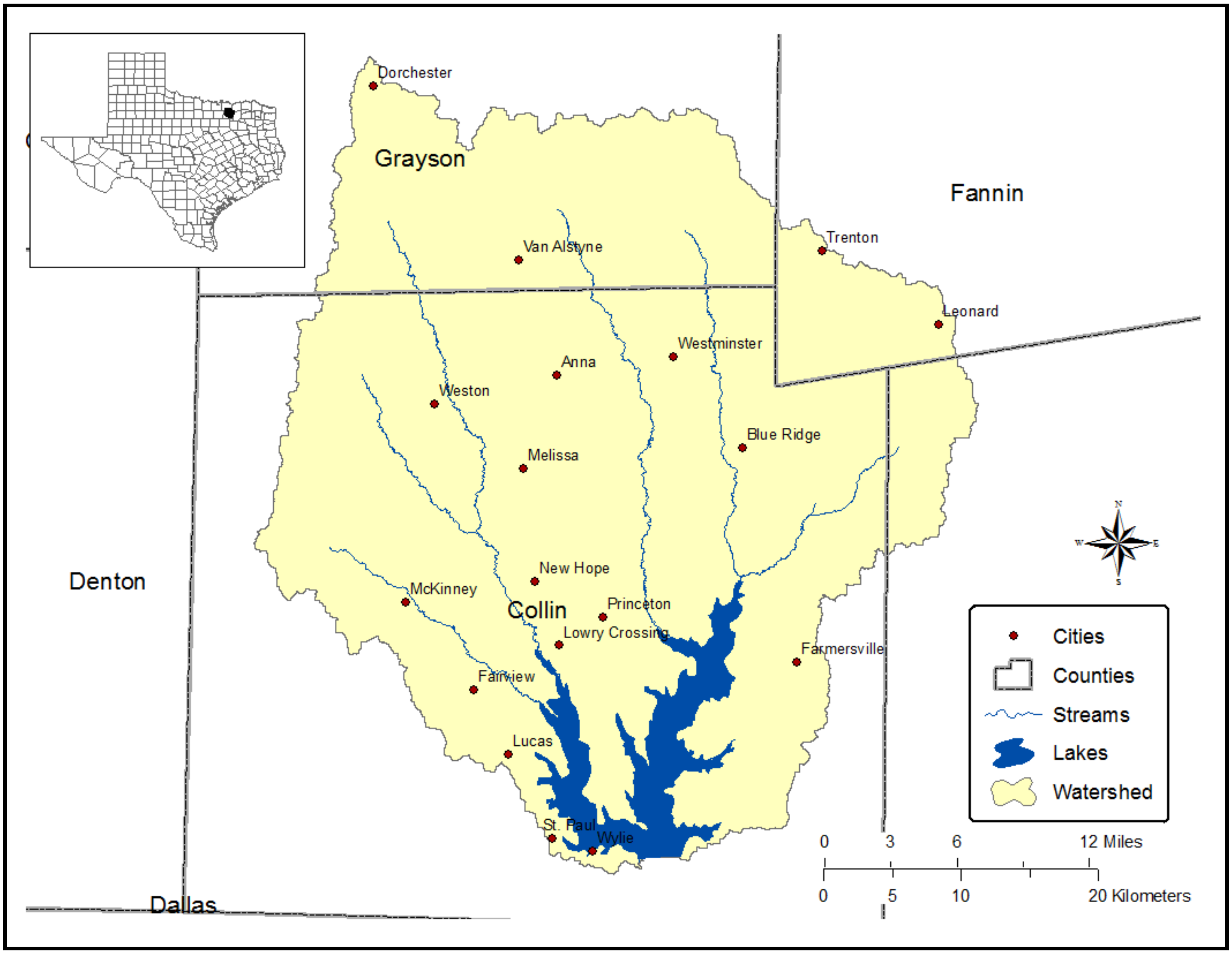


Figure 4.1 Location of Lavon Basin.

Model Input Data Tables and Figures

Topography

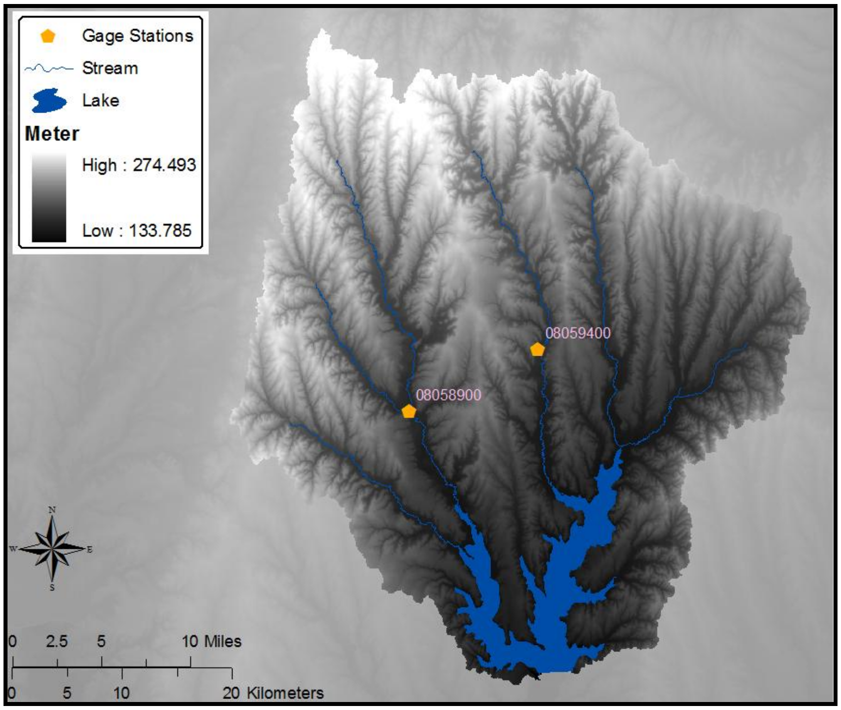


Figure 4.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of Lavon Basin.

Soils

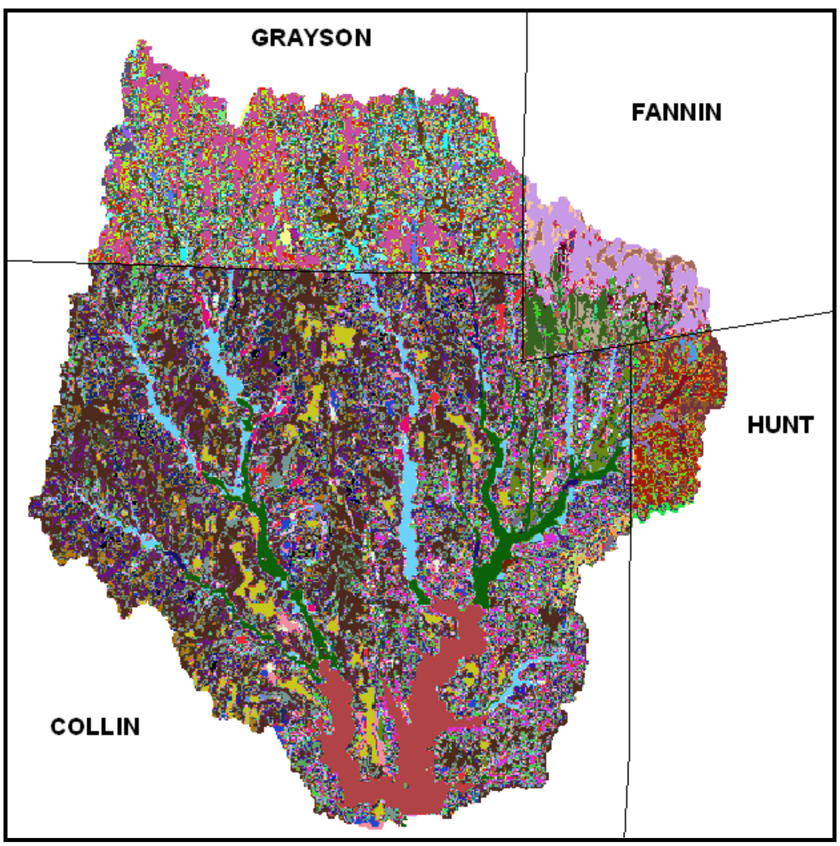


Figure 4.3 Soil Survey Geographic (SSURGO) data was used to define soil attributes in the SWAT model.

Land use

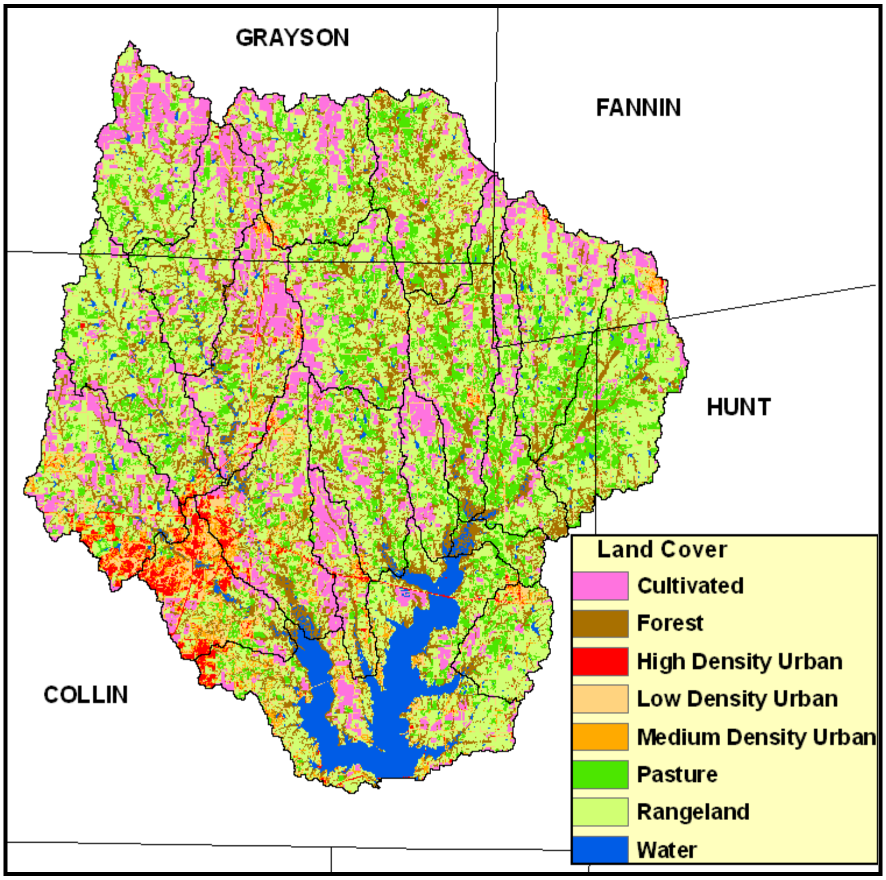


Figure 4.4 National Land Cover Data (2001) defined the land cover of the Lavon Basin.

Weather

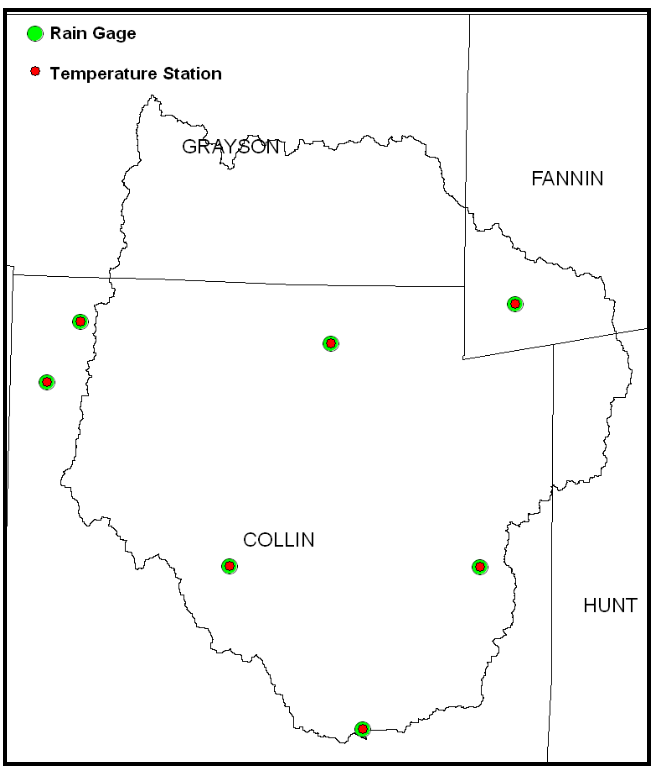


Figure 4.5 Seven precipitation and temperature stations, located in and around Lavon Basin, provided daily rainfall and temperature values (maximum and minimum) for the SWAT simulation model.

Point Sources

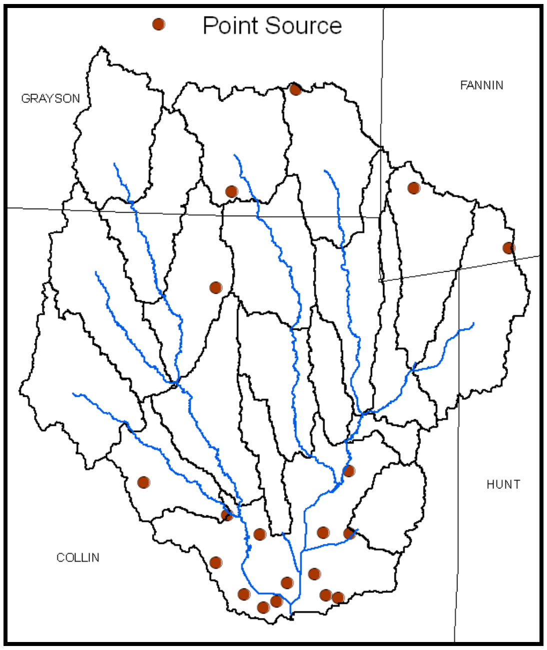


Figure 4.6 This figure shows the 19 actively operating wastewater treatment plants with measured discharge data that fall within the Lavon Basin. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

Reservoir

Table 4.1 The Lavon Basin contains one large reservoir, Lake Lavon, which was constructed prior to the simulation period. SWAT utilized available daily reservoir outflow data and reservoir characteristics from the National Inventory of Dams.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Lake Lavon	20	21,399	45.65	34,765	74.16	Measured

Subbasin Delineation

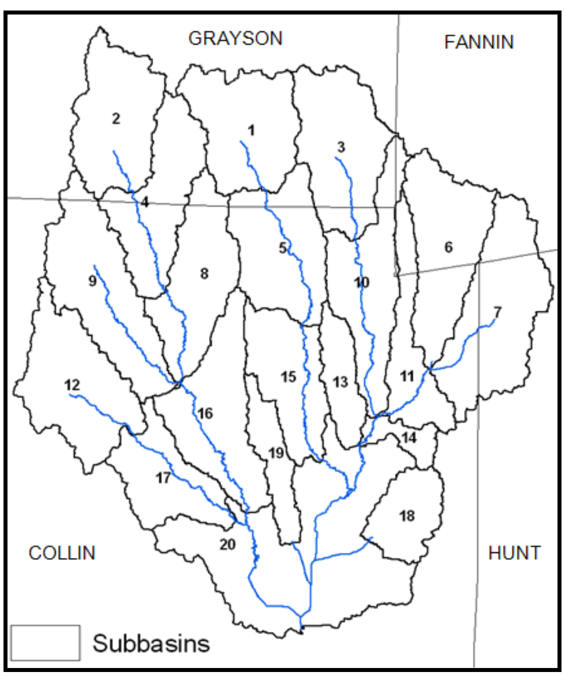


Figure 4.7 The Lavon SWAT model used a stream threshold value of 9,884 acres to delineate subbasins, resulting in 20 subbasins.

Ponds

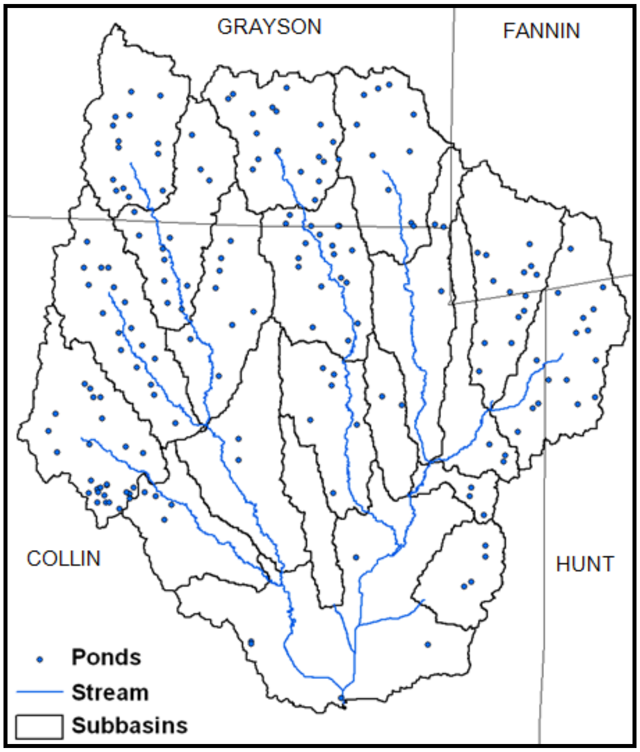


Figure 4.8 The National Inventory of Dams, created by the U.S. Army Corps of Engineers, provided SWAT with the location of ponds in the Lavon Basin. Approximately 34% of the basin area drains into ponds.

HRU Distribution

Each of the 20 subbasins were further split into HRUs based on researcher-defined thresholds, resulting in 1,164 HRUs.

Model Calibration and Validation

Model simulation began in 1960. To determine the appropriate baseflow proportion, streamflow records from the USGS gauging station at the East Fork Trinity River in McKinney (08058900, outlet of subbasin No. 8 in Lavon Basin) were analyzed using the Baseflow Filter Program (Arnold and Allen, 1999; Arnold et al., 1995). During calibration, we carefully matched the proportions of surface flow and baseflow. To calibrate SWAT for annual and monthly streamflow, we used flow records from the USGS gauging station at McKinney for the period ranging from 1976 through 1985. In addition, the McKinney gauging station also provided data used in the validation of flow from 1986 through 1995. We also validated flow at the USGS gauging station at Sister Grove Creek near Blue Ridge (08059400, outlet of subbasin No. 5 in Lavon Basin) for the period lasting from 1976 through 1995. The location of gauges is shown in figure 4.9.

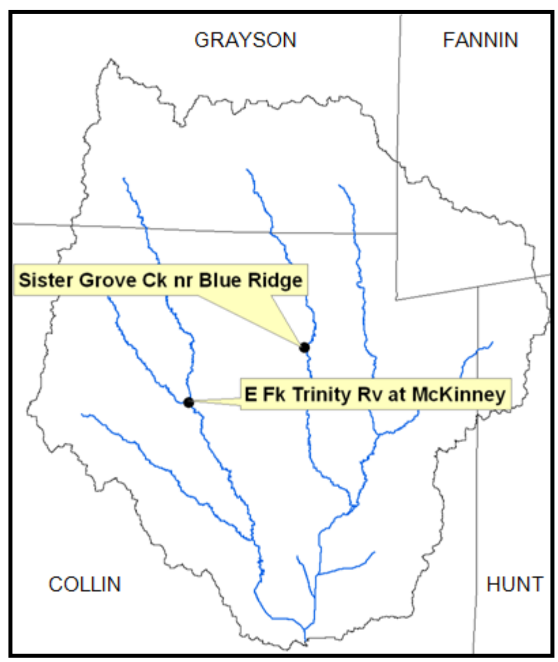


Figure 4.9 USGS gauges used for calibration and validation in the Lavon Basin.

The available monitoring station in McKinney provided data for calibration of the following water quality parameters: sediment, organic nitrogen, nitrate-nitrite, total nitrogen, mineral phosphorous and total phosphorous. However, due to limited data, all available water quality data were used for water quality calibration. So, to validate the model, a downstream monitoring station (USGS gauge 08061750) at the East Fork Trinity River near Forney provided additional data (see Chapter 7: Ray Hubbard Basin). Calibrated parameters are listed in table 4.2.

Table 4.2 Model parameters and ranges used for calibration and final calibrated values.

Component	Parameter (file)	Description	Range	Calibrated value
Flow	CN2 (.mgt)	Initial NRCS runoff curve number for moisture condition II	-5–5	Default + 5.0
	ESCO (.bsn)	Soil evaporation compensation factor	0.01–1.0	0.9
	EPCO (.bsn)	Plant uptake compensation factor	0.01–1.0	0.95
	GW_REVAP (.gw)	Groundwater revap coefficient	0.02–0.2	0.09
	GWQMN (.gw)	Threshold depth of water in the shallow aquifer required for return flow to occur	0.0–300.0	20
	REVAPMN (.gw)	Threshold depth of water in the shallow aquifer for "revap" to occur [mm]	1.0–15.0	1.0
Sediment	SPCON (.bsn)	Linear parameter for estimating the maximum amount of sediment that can be re-entrained during channel sediment routing	0.0001–0.01	0.0015
	CH_COV (.rte)	Channel cover factor	0.0–1.0	0.6
	CH_EROD (.rte)	Channel erodibility factor	0.0–1.0	0.7
Nutrients	RHOQ (.wwq)	Algal respiration rate at 20ºC (day^{-1})	0.05–0.50	0.1
Mineral nitrogen	SDNCO (.bsn)	Denitrification threshold water content (fraction of field capacity water content above which denitrification takes place)		1.1
	NPERCO (.bsn)	Nitrate percolation coefficient		1.0
Nitrogen in reach	AI1 (.wwq)	Fraction of algal biomass that is nitrogen	0.07–0.09	0.07
	RS4 (.swq)	Rate coefficient for organic N settling in the reach at 20ºC (day^{-1})		0.1
	BC3 (.swq)	Rate constant for hydrolysis of organic N to NH_4 in the reach at 20ºC (day^{-1})		0.02
Mineral phosphorus	PPERCO (.bsn)	Phosphorus percolation coefficient	10.0–17.5	10
	PHOSKD (.bsn)	Phosphorus soil partitioning coefficient	100–175	105
Phosphorus in reach	AI2 (.wwq)	Fraction of algal biomass that is phosphorus	0.01–0.02	0.02
	BC4 (.swq)	Rate constant for mineralization of organic P to dissolved P in the reach at 20ºC (day^{-1})		0.01
	RS5 (.swq)	Organic phosphorus settling rate in the reach at 20ºC (day^{-1})		0.2

Results and Discussion

Model Calibration and Validation

Flow

Calibration of annual streamflow (1976–1985) using data from the USGS gauging station at McKinney (figure 4.9) resulted in R^2 and NSE values of 0.91 and 0.90, respectively. While a monthly comparison during the calibration period resulted in R^2 and NSE values of 0.88 and 0.87, respectively (table 4.3). Validation of annual and monthly streamflow at the McKinney gauging station (1986–1995) and the Sister Grove Creek near Blue Ridge gauging station (1976–1995) resulted in R^2 and NSE values ranging from 0.70 to 0.83, respectively (table 4.3). Simulated flow for both the calibration and validation periods corresponded well with observed data (figures 4.10–4.13). Flow from Sister Grove Creek, another tributary that feeds into Lake Lavon, was not calibrated. However, NSE validation results were above 0.75, indicating that the calibrated parameters of the McKinney tributary fit well with the drainage area of Sister Grove Creek. This suggests that the hydrology of these two Lake Lavon tributaries is similar and supports the potential use of SWAT for making predictions about watersheds with similar characteristics using the same calibrated values.

Table 4.3 Annual and monthly streamflow (m^3/s) calibration and validation results at USGS gauging stations.

Gauge station ID (site name)		Period	Measured			Simulated			Yearly		Monthly	
			Mean	Std_y	Std_m	Mean	Std_y	Std_m	R^2	NSE	R^2	NSE
08058900 (East Fork Trinity River at McKinney)	Calibration	1976–1985	**2.48**	2.33	5.88	**2.67**	2.00	5.04	**0.91**	**0.90**	**0.88**	**0.87**
	Validation	1986–1995	**4.39**	1.83	6.99	**3.81**	1.60	5.26	**0.81**	**0.70**	**0.83**	**0.80**
08059400 (Sister Grove Creek near Blue Ridge)	Validation	1976–1995	**1.83**	1.07	2.96	**1.59**	1.02	2.65	**0.81**	**0.75**	**0.77**	**0.76**

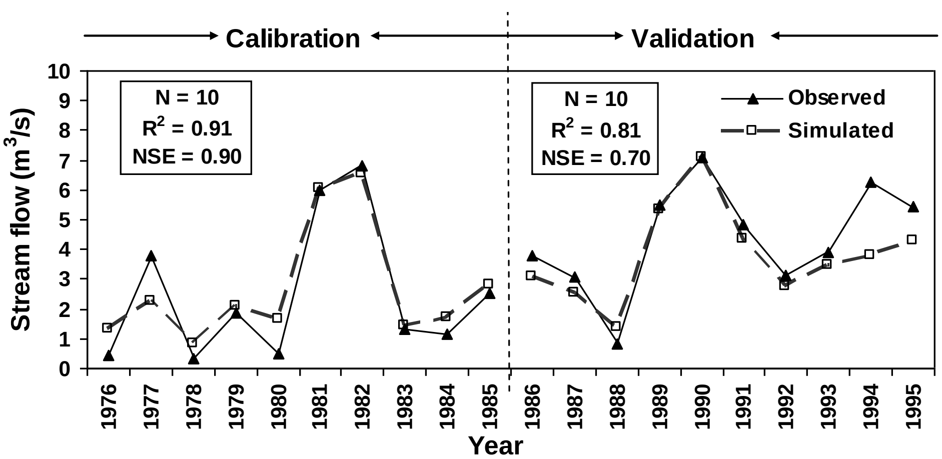


Figure 4.10 Measured and simulated yearly streamflow at USGS gauge 08058900 (East Fork Trinity River at McKinney) for the calibration (1976–1985) and validation periods (1986–1995).

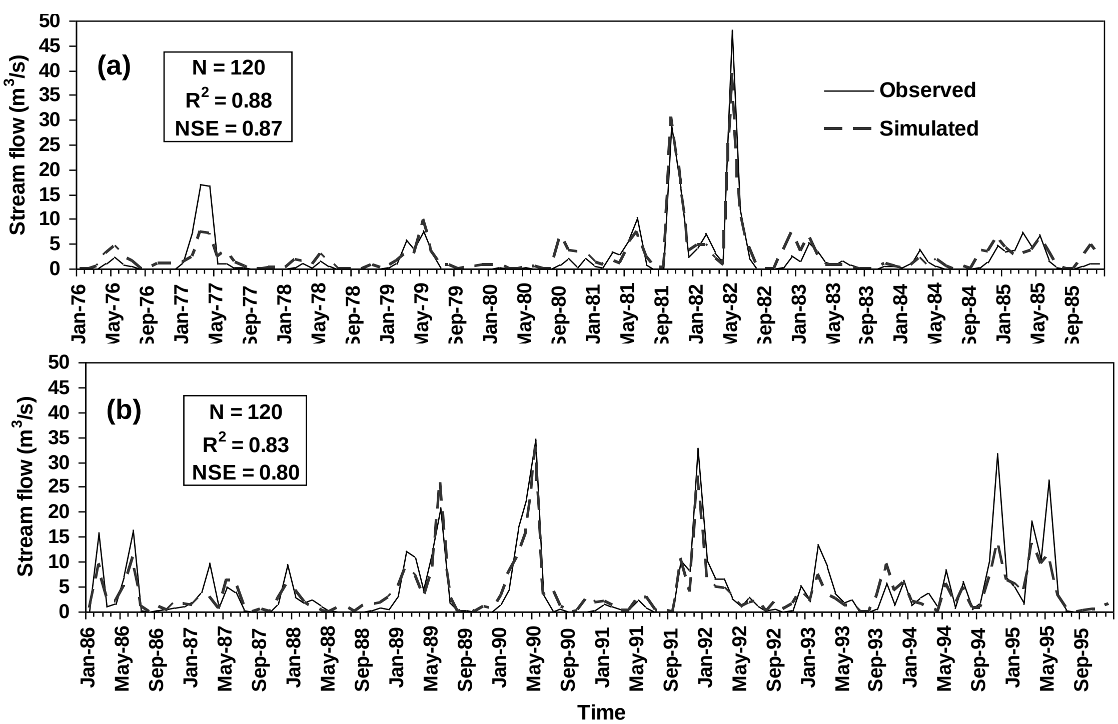


Figure 4.11 Measured and simulated monthly streamflow at USGS gauge 08058900 (East Fork Trinity River at McKinney) for the (a) calibration period (1976–1985) and (b) validation period (1986–1995).

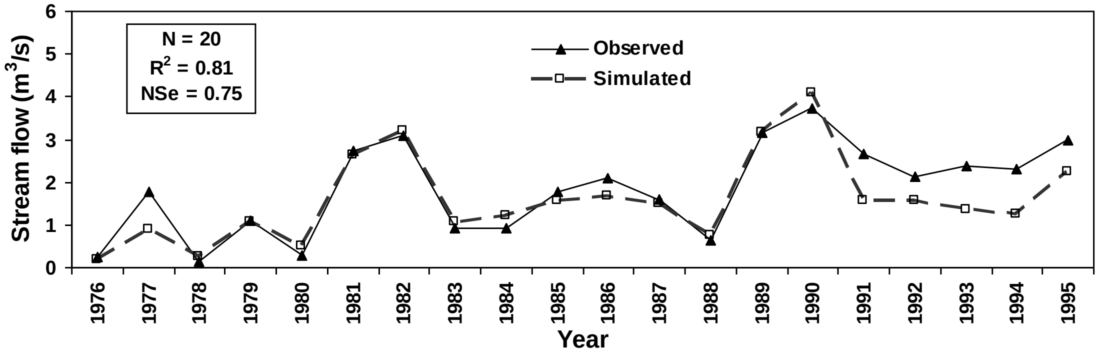


Figure 4.12 Validation of yearly streamflow at USGS gauge 08059400 (Sister Grove Creek near Blue Ridge).

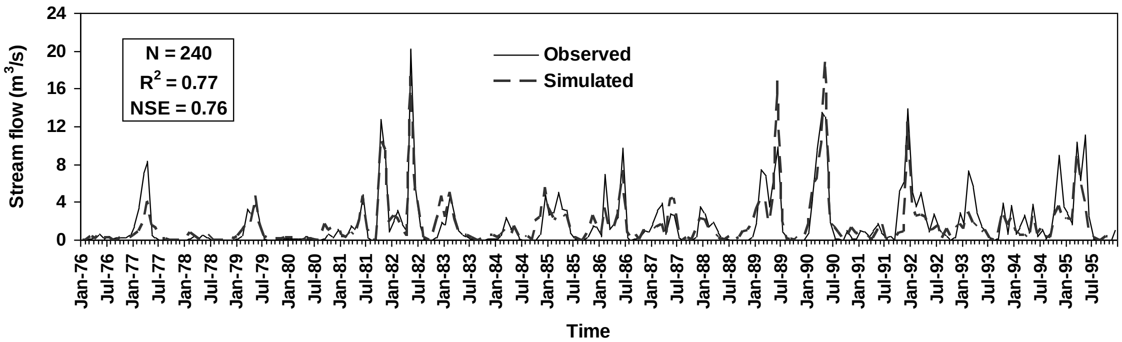


Figure 4.13 Validation of monthly streamflow (1976–1995) at USGS gauge 08059400 (Sister Grove Creek near Blue Ridge).

Sediment and Nutrients

Available water quality data from the Lavon Basin were used for calibration. While a downstream monitoring station (see Chapter 7: Ray Hubbard Basin) provided data for model validation. We compared measured and simulated average daily loads for days with available data. In general, simulated sediment, mineral phosphorus, total phosphorus and organic nitrogen loads compared well with corresponding measured values (table 4.4 and figure 4.14). However, the nitrate-nitrite component was clearly under-predicted.

Table 4.4. Water quality calibration for days with available data at the USGS gauging station in McKinney. Sediment yield, NO_3-NO_2 and total N were available from April 1993 to August 1995, giving 29 to 30 data points. Organic N and total P were available during some years from November 1981 to August 1995, with 48 data points for each. (Obs.= Observed values and Sim. = simulated values).

	Sediment metric tons/day (tons/day)		Mineral P		Total P		NO_3-NO_2		Organic N		Total N	
			------------------------------------ kg/day-- (lb/day)									
	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.
Mean	226.9 (250)	213.0 (235)	16.3 (35.9)	16.2 (35.7)	139.7 (308.0)	131.9 (290.8)	434.1 (957.2)	270.7 (596.9)	678.3 (1495.7)	627.8 (1384.3)	621.5 (1370.4)	640.3 (1141.9)
%Error		6.1		-0.8		-5.6		-37.6		-7.4		3.0

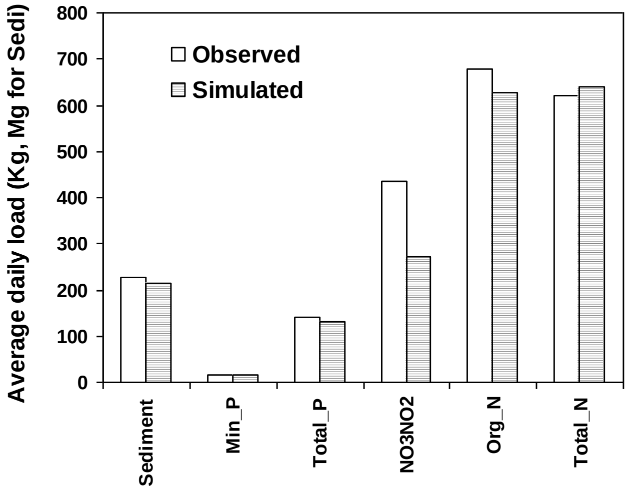


Figure 4.14
Measured and simulated average daily sediment in Megagrams (1 Megagram = 1 metric ton) and nutrient loads in kilograms at USGS gauge 08058900 (East Fork Trinity River at McKinney).

Model Predictions

During the simulation period (1968–2007), the model predicted that overland within Lavon Basin generated an average of 262,775 metric tons (289,660 tons) of sediment per year under current management scenarios, but the total sediment load actually reaching Lake Lavon was about 535,400 metric tons (590,177 tons) per year. The channel banks contributed about 55% of the sediment load reaching the reservoir. However, sediment was not generated uniformly throughout the basin. Overland range areas with steeper slopes generated higher sediment yields while overland areas with more ponds generated relatively less sediment (figure 4.15).

During the simulation period (1968–2007), the model predicted that overland in Lavon Basin generated phosphorus and nitrogen loads of 274,924 kilograms (606,104 pounds) per year and 1,622,043 kilograms (3,575,993 pounds) per year, respectively, under current management scenarios. Total phosphorus losses from overland flow are mapped in figure 4.16. Point sources generated 43,586 kilograms (96,091 pounds) per year of phosphorus and 336,979 kilograms (742,911 pounds) per year of nitrogen. The model predicted that, of the nutrients derived from overland areas and point sources, 210,750 kilograms (464,624 pounds) of total phosphorus and 2,671,500 kilograms (5,889,648 pounds) of total nitrogen reach Lake Lavon annually.

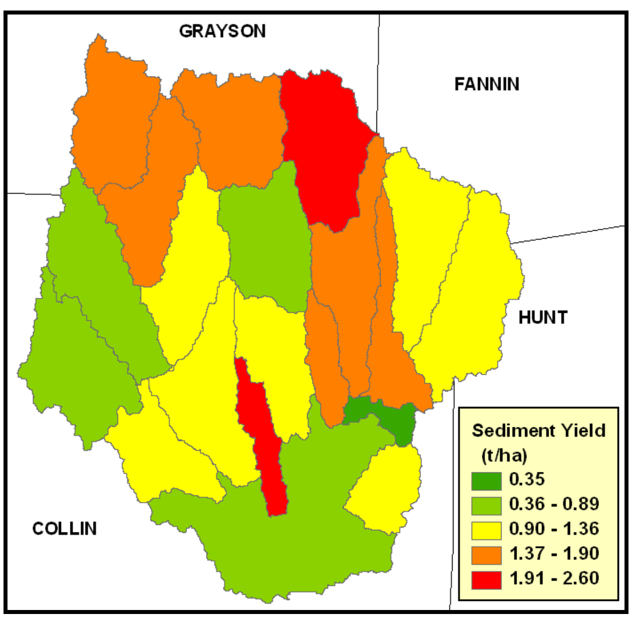


Figure 4.15 Sediment losses from Lavon Basin as predicted by the SWAT model.

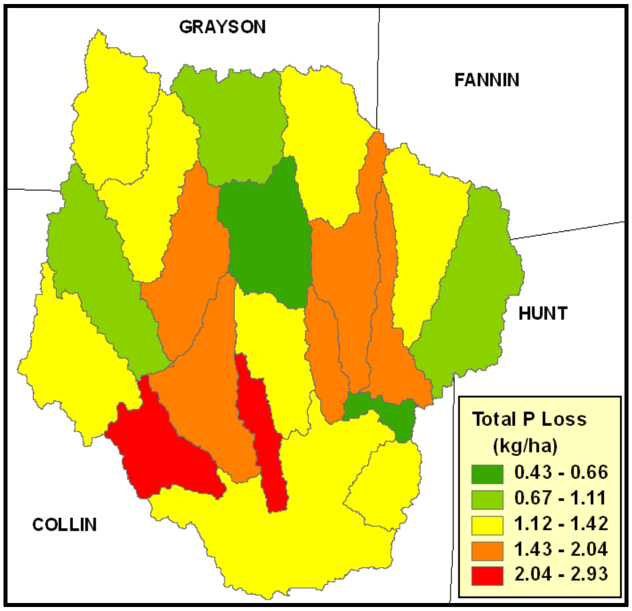


Figure 4.16 Phosphorus losses from Lavon Basin as predicted by the SWAT model.

Scenarios

Conservation Practices

SWAT simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 4.5.

Ponds

According to the model, the 185 ponds, determined based on the National Inventory of Dams' PL-566 structures (see Chapter 2, General Model Input Data), in the Lavon Basin reduce Lake Lavon's sediment loading by 19%. By removing all ponds from the basin model simulation, sediment (figure 4.17) and nutrient loads reaching the reservoir increased.

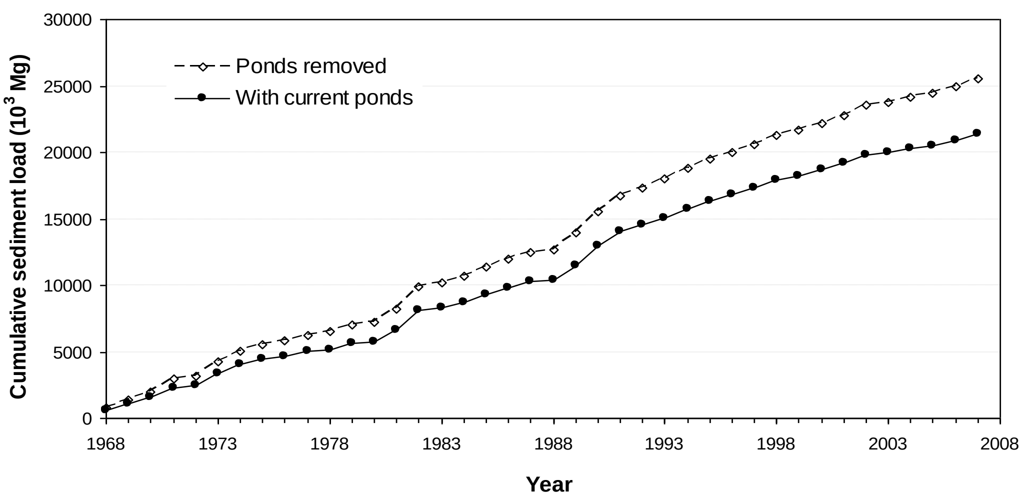


Figure 4.17 Cumulative sediment loads reaching Lake Lavon as predicted by the SWAT model.

Range Utilization

Removing grazing activity and all rangeland management resulted in an 9% reduction in the sediment load and 11% reduction in total phosphorus load received by Lake Lavon (table 4.5).

Point Source Load Elimination

The basin contains nineteen point sources. Eliminating these point sources would reduce total phosphorus and nitrogen loads received by Lake Lavon by 24% and 56%, respectively. Also, the sediment load received by Lake Lavon would be reduced by 6%. As stated in the Model Prediction section, river channels contribute about 55% of the sediment load received by the reservoir. According to the SWAT simulation, eliminating point source discharges would reduce flow energy, the driving force carrying sediment particles. Therefore, the total sediment yield was substantially reduced by eliminating these point source discharges in this watershed.

Urbanization

Urban area within Lavon Basin is expected to increase from 49,668 acres in 2000 to 92,665 acres in 2030. Put differently, the percentage of urban area in Lavon Basin is predicted to change from 10% to 19% in 30 years. In SWAT, urban expansion caused total phosphorus and total nitrogen loads from overland areas to increase by 14% and 9%, respectively. Predicted changes in sediment, phosphorus and nitrogen are given by subbasin in figures 4.19–4.21.

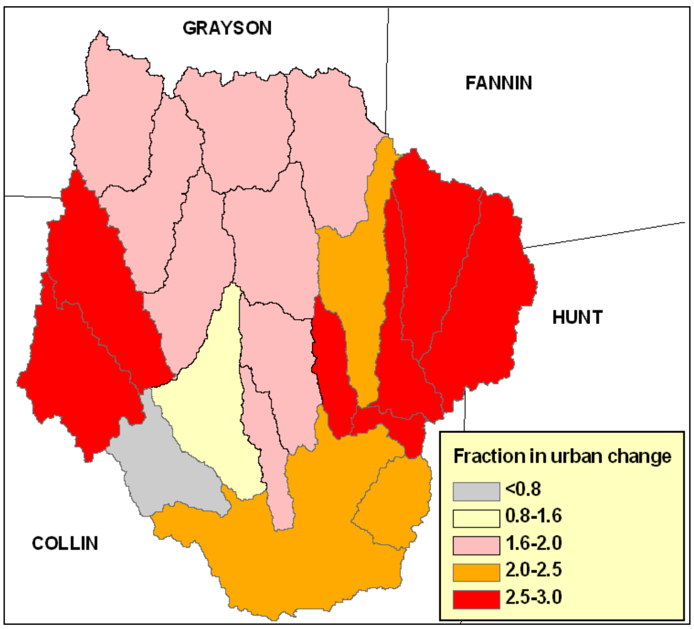


Figure 4.18
Projected change in urban areas from 2000 to 2030 (urban area in 2030 = urban area in 2000 * fraction of urban change).

Table 4.5 Changes in sediment and nutrient loads received by Lake Lavon under different scenarios as derived from SWAT model simulations (1968–2007).

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	535,400 metric tons/yr (590,177 tons/yr)	210,750 kg/yr (464,624 lb/yr)	2,671,500 kg/yr (5,889,648 lb/yr)
No Ponds	19.4%	12.7%	7.5%
No Range Grazing	-9.2%	-11.3%	-0.3%
Urban *	-2.9%	14.4%	9.2%
No Point Sources	-5.5%	-24.2%	-55.7%

* load changes from overland only

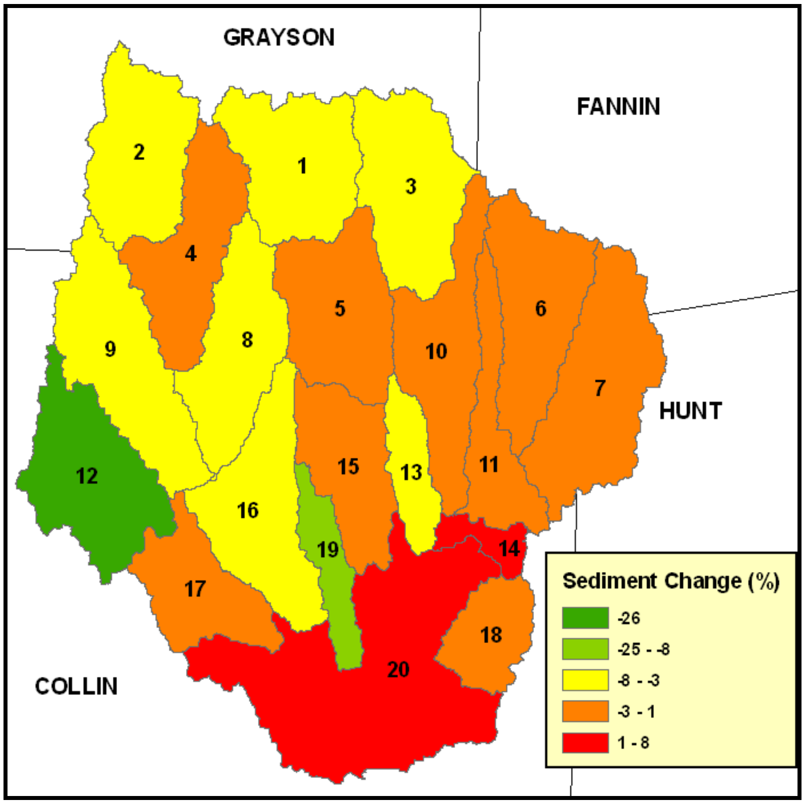


Figure 4.19 Change in basin sediment losses from 2000 to 2030 as predicted by SWAT.

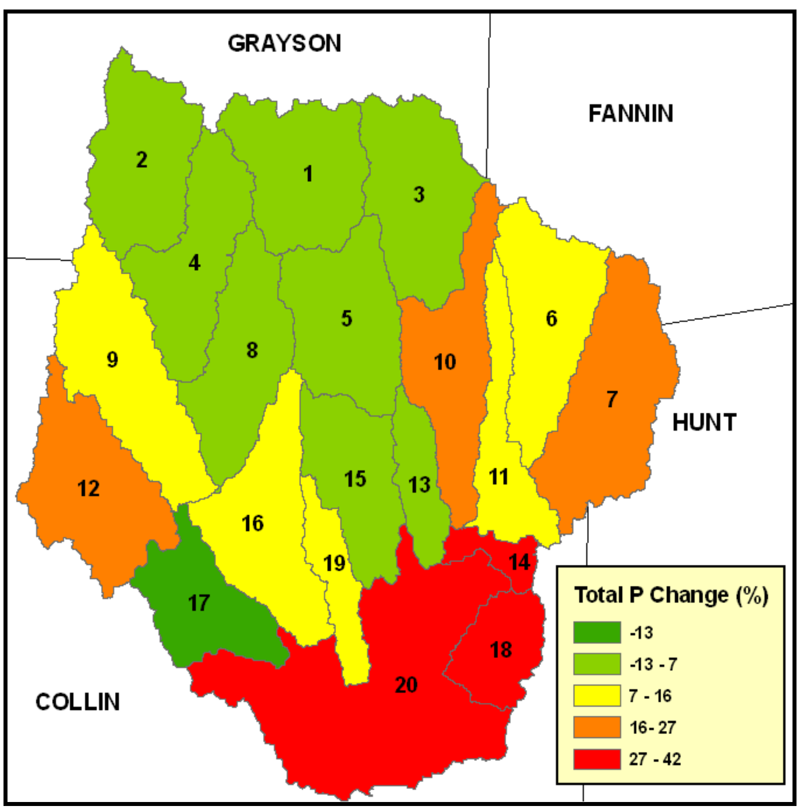


Figure 4.20 Change in basin phosphorus losses from 2000 to 2030 as predicted by SWAT.

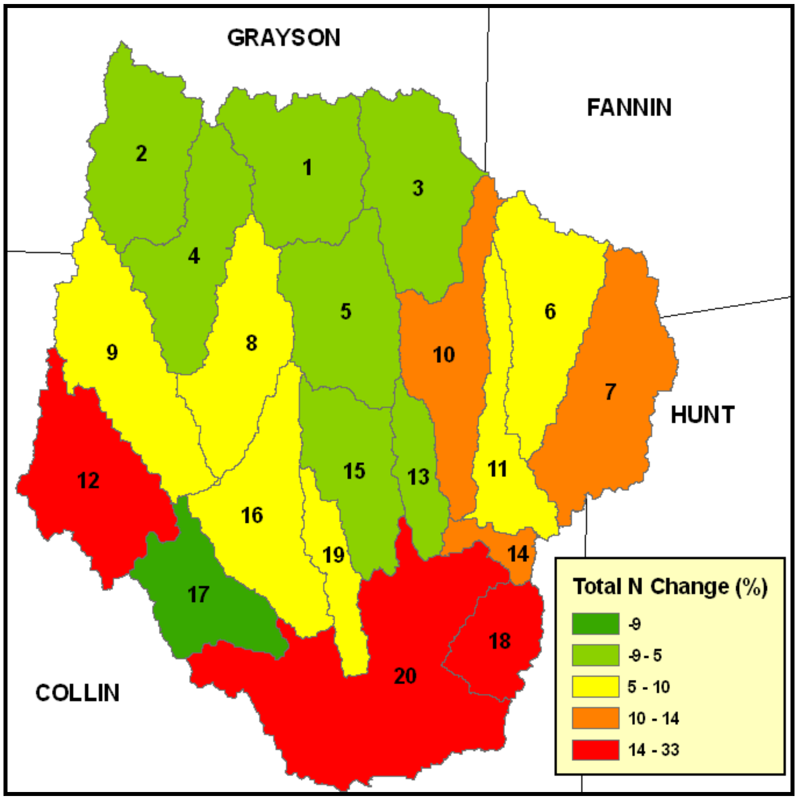


Figure 4.21 Change in basin nitrogen losses from 2000 to 2030 as predicted by SWAT.

Conclusions

The SWAT model was used to simulate the effects of urbanization and other land use changes on hydrologic and water quality processes in the Lavon Basin. Using flow data available in the watershed, SWAT calibration and validation resulted in a modeled representation of the watershed that was within acceptable standards. NSE values based on monthly flow comparisons ranged from 0.76 and 0.87. Also, simulated average daily sediment, mineral phosphorus, total phosphorus, organic nitrogen and total nitrogen loads were within ±10% of the corresponding observed values. The results indicate that SWAT was able to realistically replicate conditions within Lavon Basin.

During the simulation period ranging from 1968–2007, the model predicted that the total sediment load reaching Lake Lavon was about 535,400 metric tons (590,177 tons) per year with channel banks contributing about 55% of the sediment load. The phosphorus and nitrogen loads resulting from overland areas were 274,924 kilograms (606,104 pounds) per year and 1,622,043 kilograms (3,575,993 pounds) per year, respectively, under current management scenarios. The model predicted that, of the nutrients derived from overland areas and point sources, 210,750 kilograms (464,624 pounds) of total phosphorus and 2,671,500 kilograms (5,889,648 pounds) of total nitrogen reach Lake Lavon annually.

In the scenario analyses, the complete removal of ponds, point sources and rangeland management demonstrated their overall impact on the water quality of Lavon Basin. Based on model predicted values, the removal of ponds would increase reservoir loading with sediment by approximately 19%, total phosphorus by 13% and total nitrogen by 8%. Removing point sources achieved nutrient reductions of approximately 24% in total phosphorus and 56% in total nitrogen. Finally, removing grazing from rangelands reduced the sediment load by approximately 9%, total phosphorus load by 11% and total nitrogen load by 0.3%. SWAT also predicted the effects of urban expansion in 2030. In the urbanization projections, urban areas are expected to increase from 10% in 2000 to 19% in 2030. Associated with this expansion, the model predicted approximate increases of 14% in total phosphorus and 9% in total nitrogen resulting from overland areas.

References

See the appendix.

Chapter 5: Lewisville Basin

Introduction

The U.S. Army Corps of Engineers constructed Lake Lewisville in 1954. The lake was built for flood control purposes and to serve as a water source for Dallas and its suburbs. However, the lake is primarily used recreationally for boating and watercraft. Lake Lewisville's watershed is located within the lower part of USGS designated Hydrologic Unit Code 12030103 and encompasses an area of 622,706 acres on the Elm Fork of the Trinity River in Denton County near Lewisville. The watershed covers parts of Montague, Cooke, Wise, Denton, Grayson and Collin counties (figure 5.1).

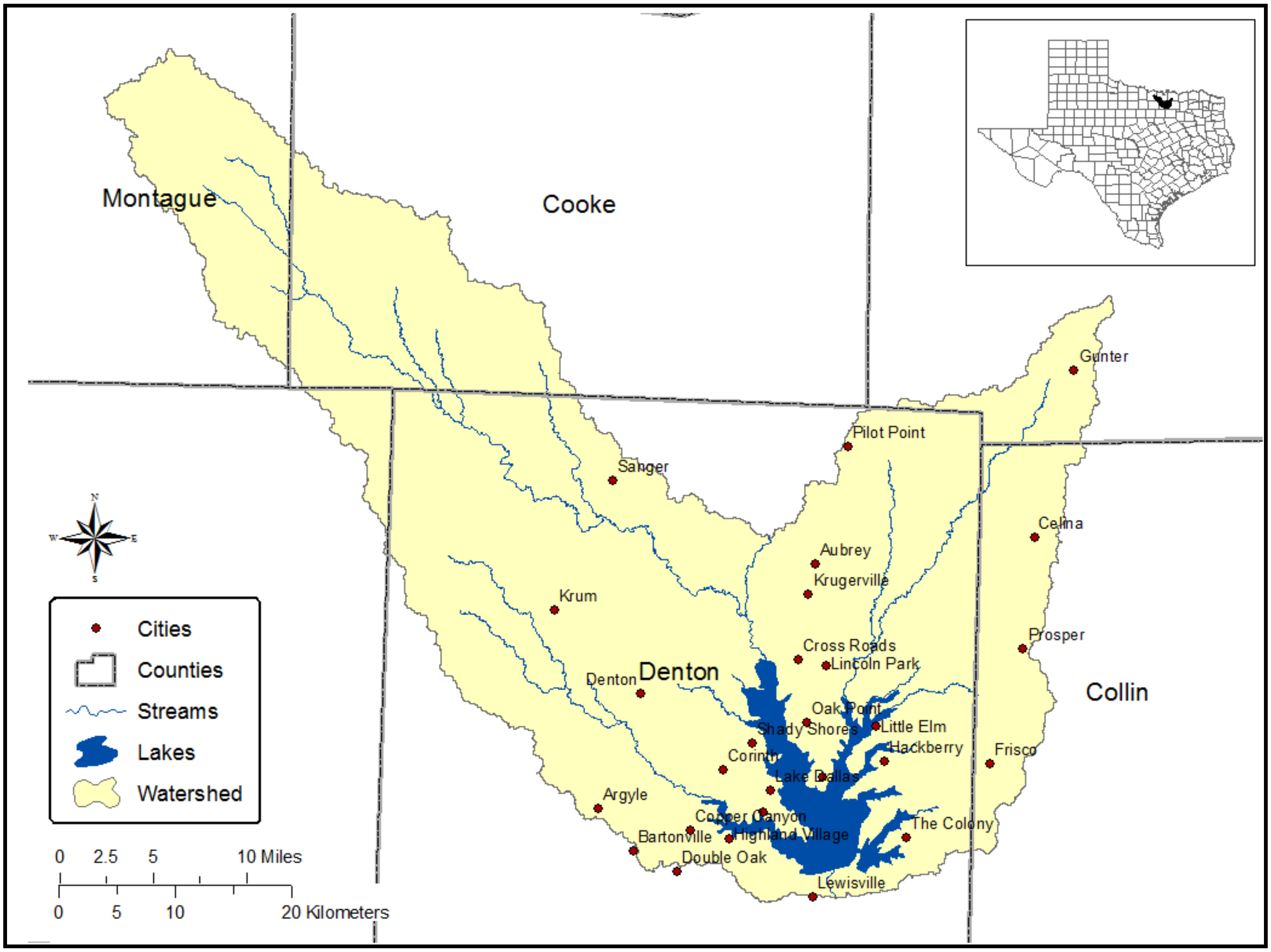


Figure 5.1 Location of Lewisville Basin.

Model Input Data Tables and Figures

Topography

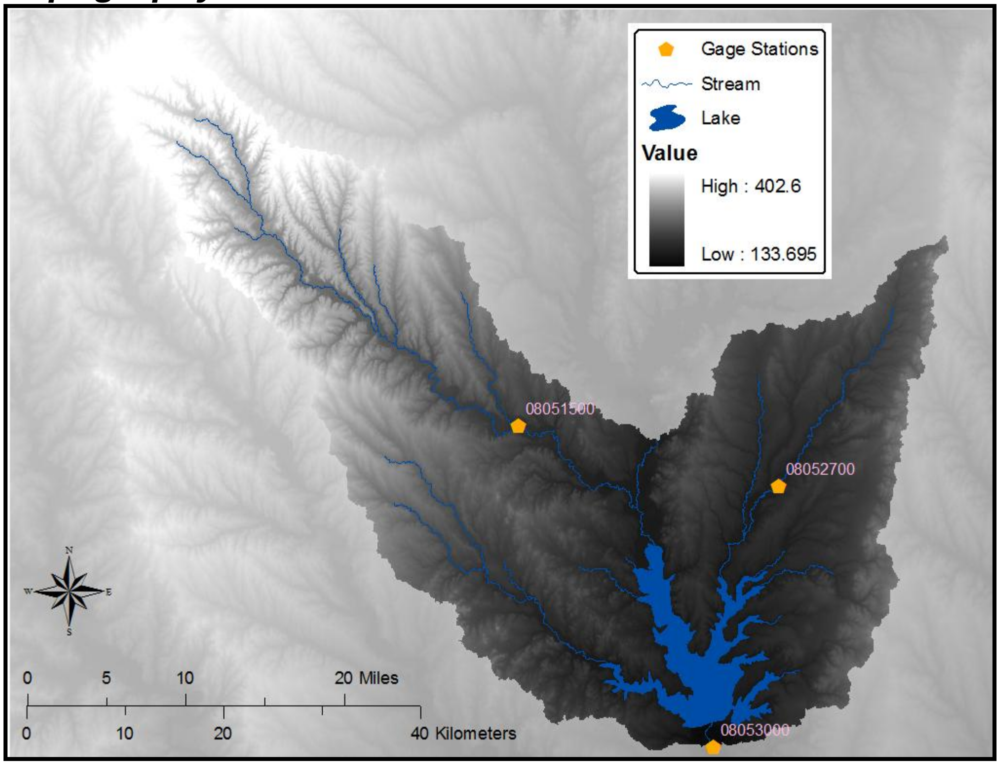


Figure 5.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of Lewisville Basin.

Soils

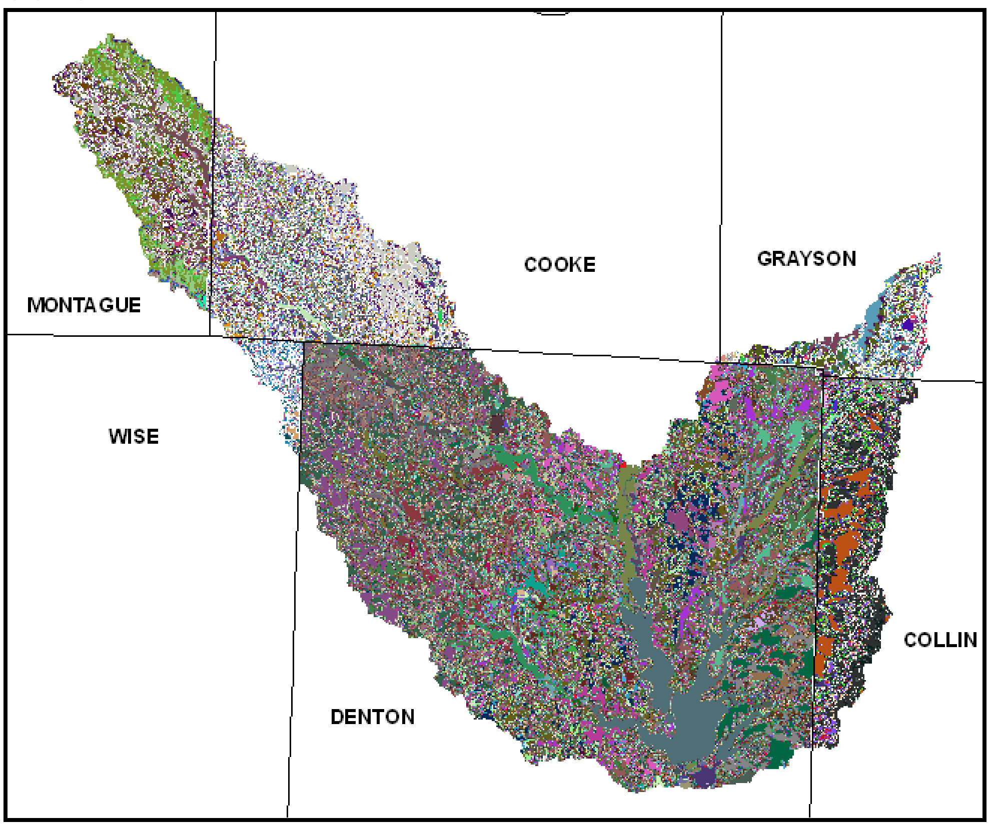


Figure 5.3 Soil Survey Geographic (SSURGO) data defined soil attributes in the SWAT model.

Land use

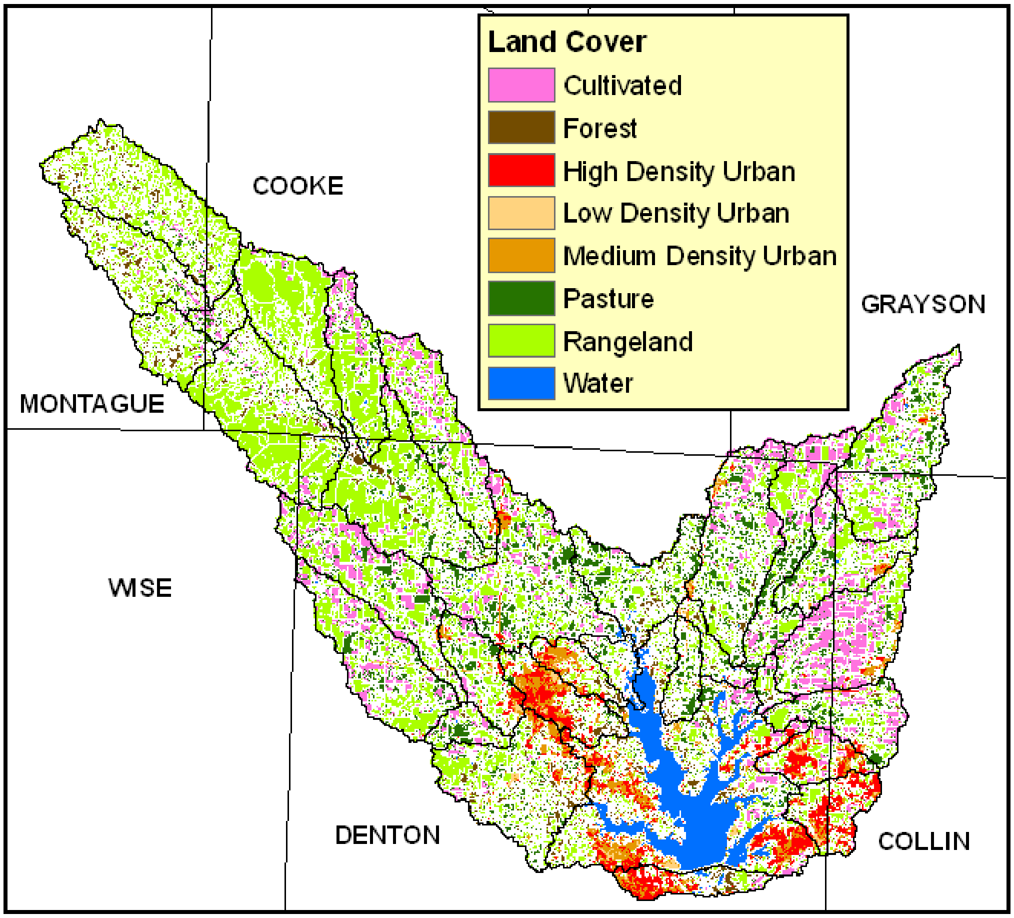


Figure 5.4 National Land Cover Data (2001) defined land uses within Lewisville Basin.

Weather

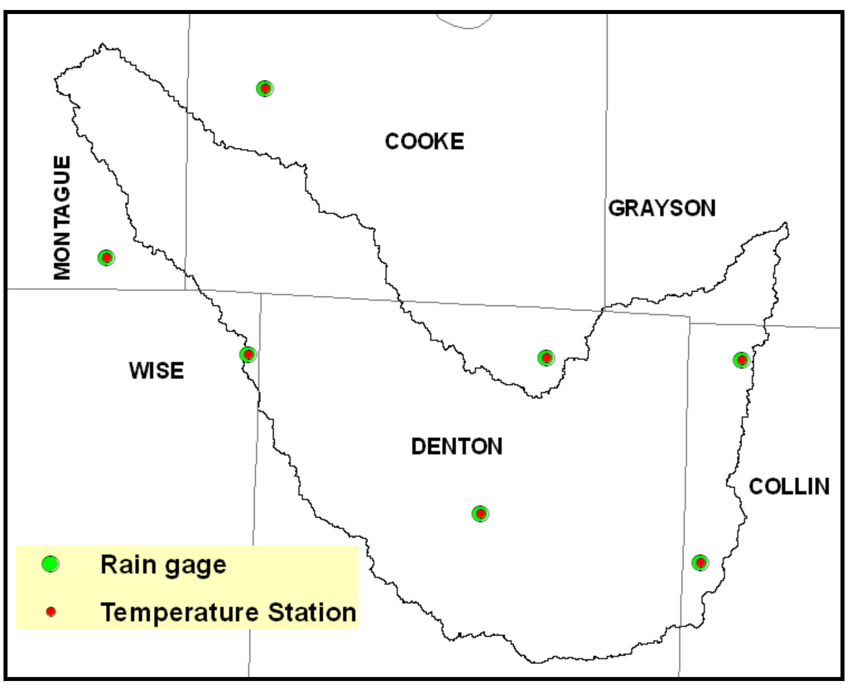


Figure 5.5 Seven precipitation and temperature stations located in and around Lewisville Basin provided daily rainfall and temperature values (maximum and minimum) for the SWAT simulation model.

Point Sources

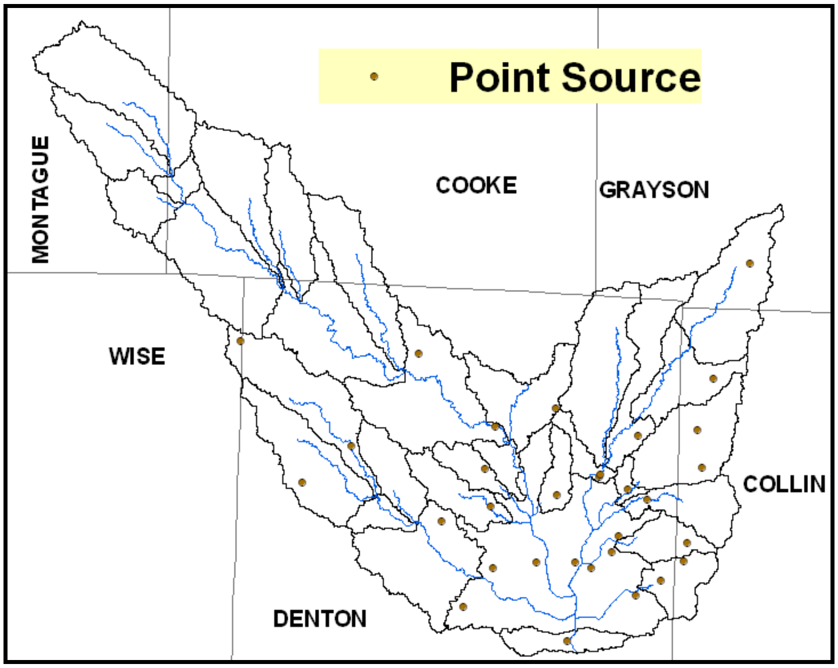


Figure 5.6 The USGS Water Resource Database identified 19 point sources with available discharge data actively operating in the Lewisville Basin. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

Reservoir

Table 5.1 The basin contains one large reservoir, Lake Lewisville, which was constructed prior to the simulation period. The National Inventory of Dams provided reservoir characteristics used in SWAT model development.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Lewisville	31	23,279	46.5	73,039	145.7	Measured

Subbasin Delineation

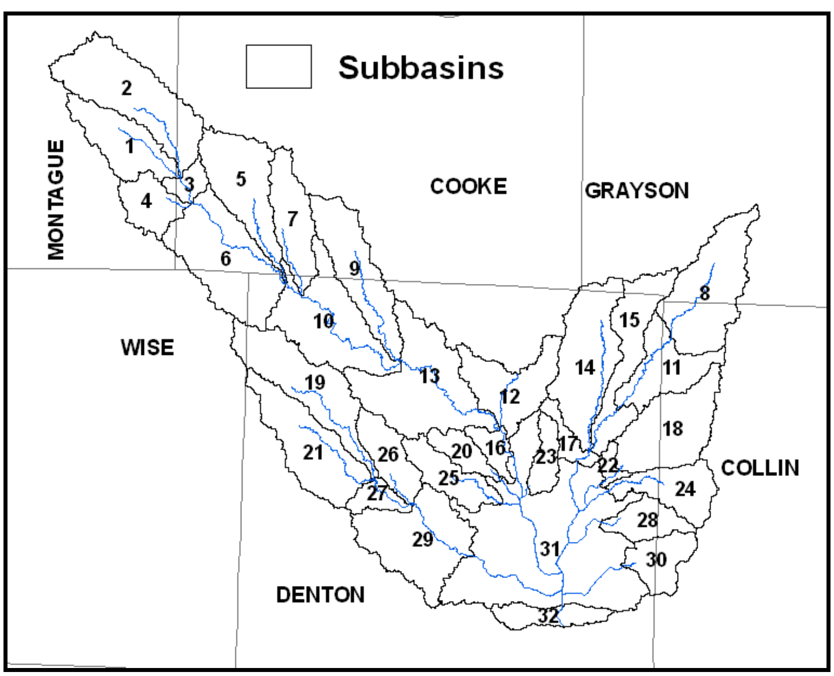


Figure 5.7 We selected a stream threshold value of 6,178 acres to delineate subbasins, creating a total of 32 subbasins in the Lewisville SWAT Model.

HRU Distribution

Using a predetermined threshold, SWAT split the 32 subbasins into 1,405 HRUs.

Ponds

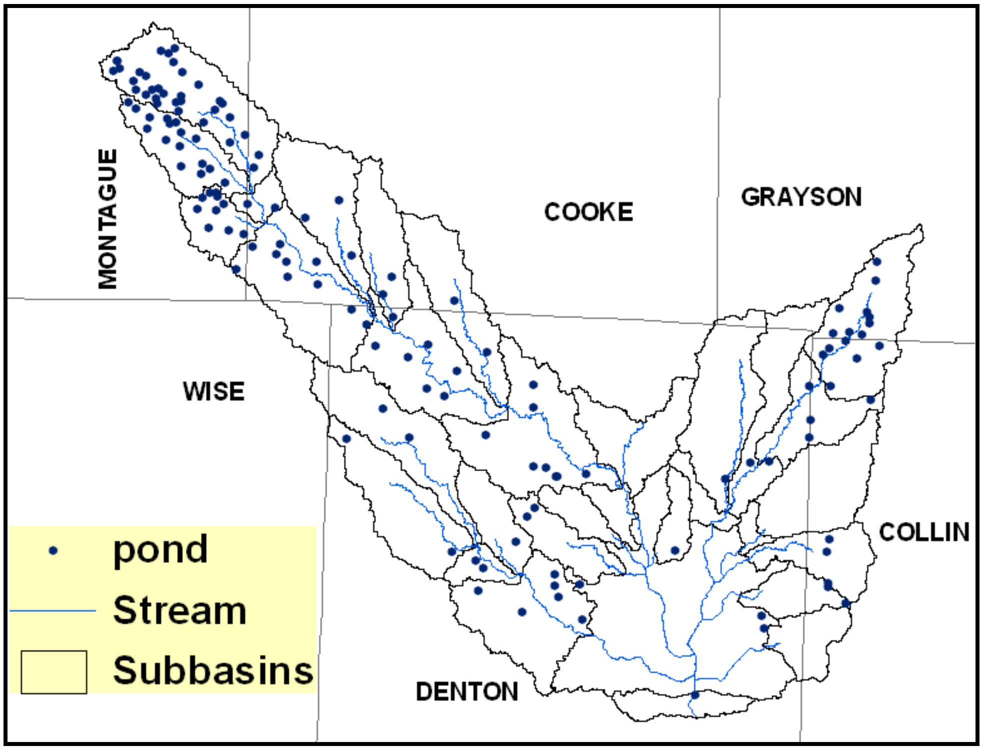


Figure 5.8 The locations of ponds based on the U.S. Army Corps of Engineers' National Inventory of Dams. Approximately 24% of the Lewisville Basin area drains to ponds.

Model Calibration and Validation

Model simulation began in 1960. Two USGS gauging stations, one at Little Elm Creek near Aubrey (08052700) and the other at Clear Creek near Sanger (08051500) (figure 5.9) provided streamflow records. We analyzed these records using the Baseflow Filter Program (Arnold and Allen, 1999; Arnold et al., 1995) to determine the appropriate baseflow proportion. During calibration, we carefully matched the proportions of surface flow and baseflow. The flow records from the two USGS gauging stations above allowed for SWAT calibration of annual and monthly streamflow from 1970 through 1986. Then, we validated the model for flow at Little Elm Creek near Aubrey for the period lasting from 1987 through 2006 and at Clear Creek near Sanger from 1987 to 2000. We also validated flow at the USGS gauging station at Elm Fork Trinity River near Lewisville (08053000) for the period lasting from 1970 through 2007.

In addition to flow calibration and validation, we also calibrated for water quality variables including sediment, mineral phosphorus, total phosphorus, nitrate-nitrite and organic nitrogen. The Little Elm Creek near Aubrey and Elm Fork Trinity River near Lewisville monitoring stations provided data for water quality calibration (figure 5.9). Then, we used about half of the available data points at Elm Fork Trinity River near Lewisville for validation. Calibrated parameters are listed in table 5.2.

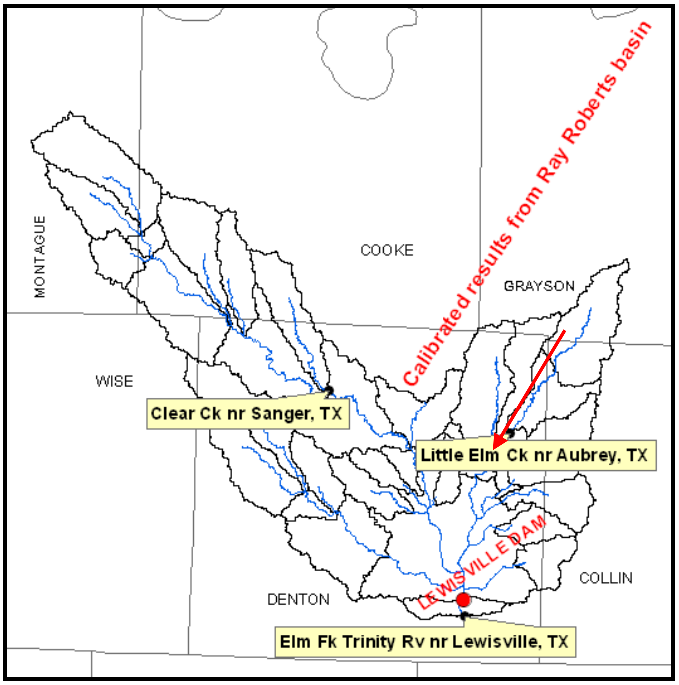


Figure 5.9 USGS gauges within Lewisville Basin that provided data for model calibration and validation.

Table 5.2 Model parameters and ranges used for calibration and final calibrated values.

Component	Parameter (file)	Description	Range	Calibrated value
Flow	CN2 (.mgt)	Initial NRCS runoff curve number for moisture condition II	-5 – 5	Default 2 –3
	ESCO (.bsn)	Soil evaporation compensation factor	0.01–1.0	0.95
	EPCO (.bsn)	Plant uptake compensation factor	0.01–1.0	1.0
	GW_REVAP (.gw)	Groundwater revap coefficient	0.02–0.4	0.12–0.2
	GWQMN (.gw)	Threshold depth of water in the shallow aquifer required for return flow to occur	0.0–300.0	0.0–1.2
	REVAPMN (.gw)	Threshold depth of water in the shallow aquifer for "revap" to occur [mm]	1.0–15.0	1.0
Sediment	SPCON (.bsn)	Linear parameter for estimating maximum amount of sediment that can be reentrained during channel sediment routing	0.0001–0.01	0.006
	CH_COV (.rte)	Channel cover factor	0.0–1.0	0.03
	CH_EROD (.rte)	Channel erodibility factor	0.0–1.0	0.01
Nutrients	RHOQ (.wwq)	Algal respiration rate at 20ºC (day^{-1})	0.05–0.50	0.1
Mineral nitrogen	SDNCO (.bsn)	Denitrification threshold water content (fraction of field capacity water content above which denitrification takes place)		1.5
	NPERCO (.bsn)	Nitrate percolation coefficient		1.0
Nitrogen in reach	AI1 (.wwq)	Fraction of algal biomass that is nitrogen	0.07–0.09	0.07
	RS4 (.swq)	Rate coefficient for organic N settling in the reach at 20ºC (day^{-1})		0.001–0.6
	BC3 (.swq)	Rate constant for hydrolysis of organic N to NH_4 in the reach at 20ºC (day^{-1})		0.001–0.02
Mineral phosphorus	PPERCO (.bsn)	Phosphorus percolation coefficient	10.0–17.5	17.5
	PHOSKD (.bsn)	Phosphorus soil partitioning coefficient	100–175	100
Phosphorus in reach	AI2 (.wwq)	Fraction of algal biomass that is phosphorus	0.01–0.02	0.015
	BC4 (.swq)	Rate constant for mineralization of organic P to dissolved P in the reach at 20ºC (day^{-1})		0.35
	RS5 (.swq)	Organic P settling rate in the reach at 20ºC (day^{-1})		0.16–0.8

Results and Discussion

Model Calibration and Validation

Flow

Calibration of annual streamflow (1970–1986) at the USGS gauging station at Little Elm Creek near Aubrey resulted in R^2 and NSE values of 0.96 and 0.92, respectively. While the monthly comparison for the same calibration period resulted in R^2 and NSE values of 0.81 and 0.74, respectively (table 5.3). Validation of annual and monthly streamflow at this gauging station (1987–2006) resulted in R^2 and NSE values ranging from 0.74 to 0.84 (table 5.3).

Calibration and validation of annual and monthly streamflow at Clear Creek near Sanger resulted in R^2 and NSE values ranging from 0.79 to 0.87 (table 5.3).

Finally, the Elm Fork Trinity River near Lewisville station is downstream from the Lewisville Dam. Because reservoir release is used as SWAT input, both R^2 and NSE values were close to 1. Overall, the model simulated flow showed trends similar to corresponding observed data (figures 5.10–5.15).

Table 5.3 Annual and monthly streamflow (m^3/s) calibration and validation results from the three USGS gauging stations used.

Gauge station ID (site name)		**Period**	**Measured**			**Simulated**			**Yearly**		**Monthly**	
			Mean	**Std_y**	**Std_m**	**Mean**	**Std_y**	**Std_m**	**R^2**	**NSE**	**R^2**	**NSE**
08052700 (Little Elm Creek near Aubrey)	Calibration	1970–1986	**1.52**	0.97	3.03	**1.37**	0.84	1.97	**0.96**	**0.92**	**0.81**	**0.74**
	Validation	1987–2006	**1.54**	0.89	2.74	**1.55**	0.79	2.17	**0.84**	**0.84**	**0.74**	**0.74**
08051500 (Clear Creek near Sanger)	Calibration	1970–1986	**2.71**	2.36	6.81	**2.50**	2.17	5.40	**0.86**	**0.85**	**0.81**	**0.79**
	Validation	1987–2000	**3.66**	2.83	7.14	**3.14**	2.38	6.42	**0.87**	**0.83**	**0.80**	**0.79**
08053000 (Elm Fork Trinity River near Lewisville)	Validation	1970–2007	**22.21**	14.36	32.13	**21.99**	14.00	31.74	**0.97**	**0.97**	**0.99**	**0.99**

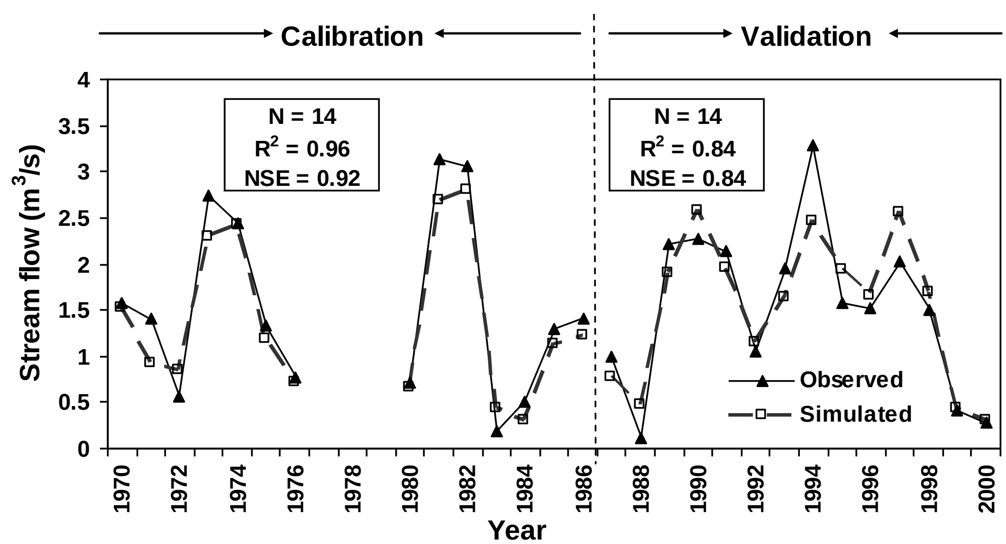


Figure 5.10 Measured and simulated yearly streamflow at Little Elm Creek near Aubrey for the (a) calibration period (1970–1986) and (b) validation period (1987–2000). Observed data was not available from 1977 to 1979.

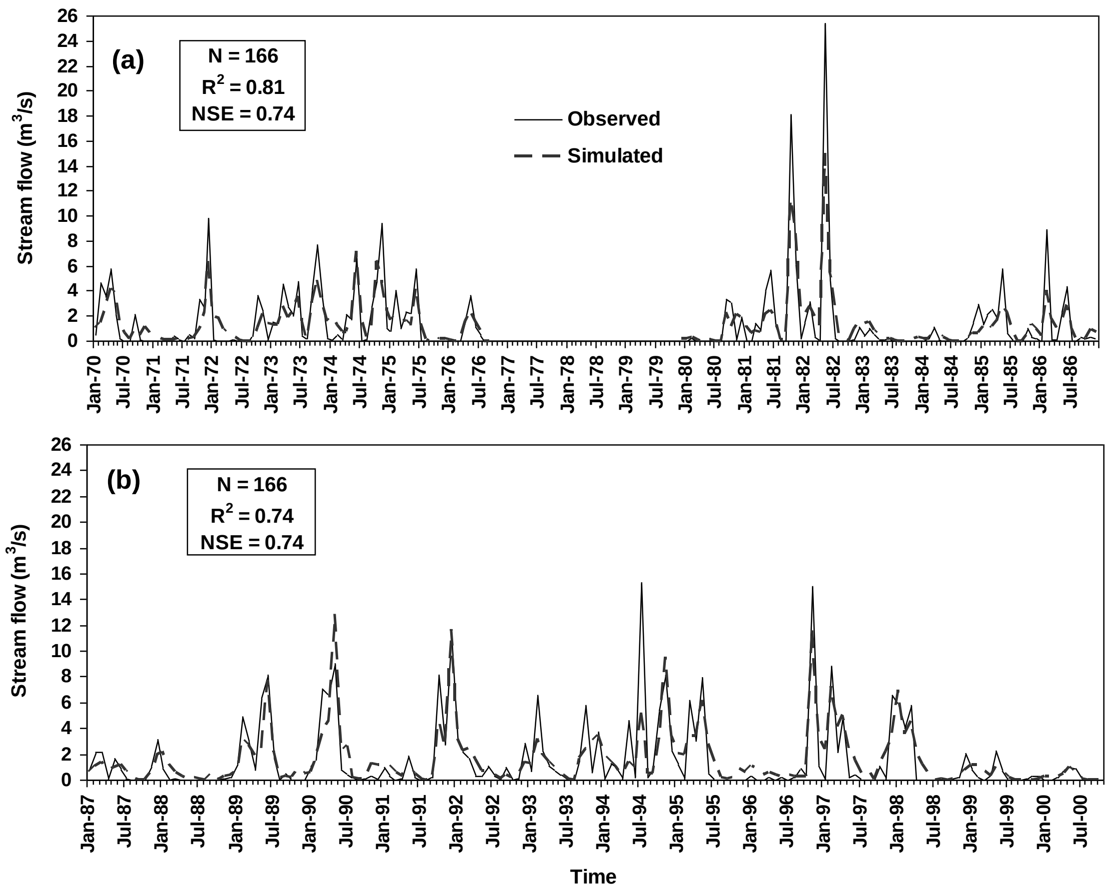


Figure 5.11 Measured and simulated monthly streamflow at Little Elm Creek near Aubrey for the (a) calibration period (1970–1986) and (b) validation period (1987–2000). Observed data was not available from October 1976 to November 1979.

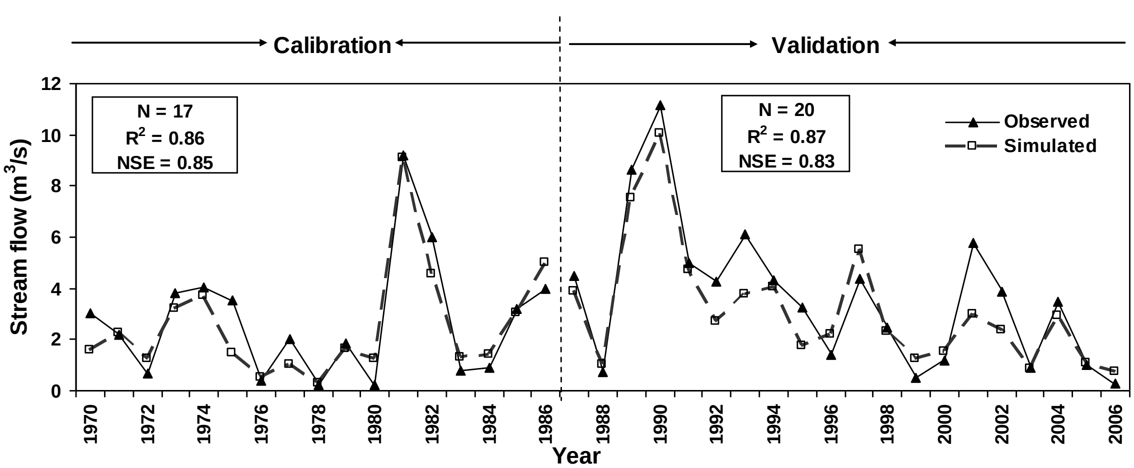


Figure 5.12 Measured and simulated yearly streamflow at Clear Creek near Sanger for the calibration (1970–1986) and validation periods (1987–2006).

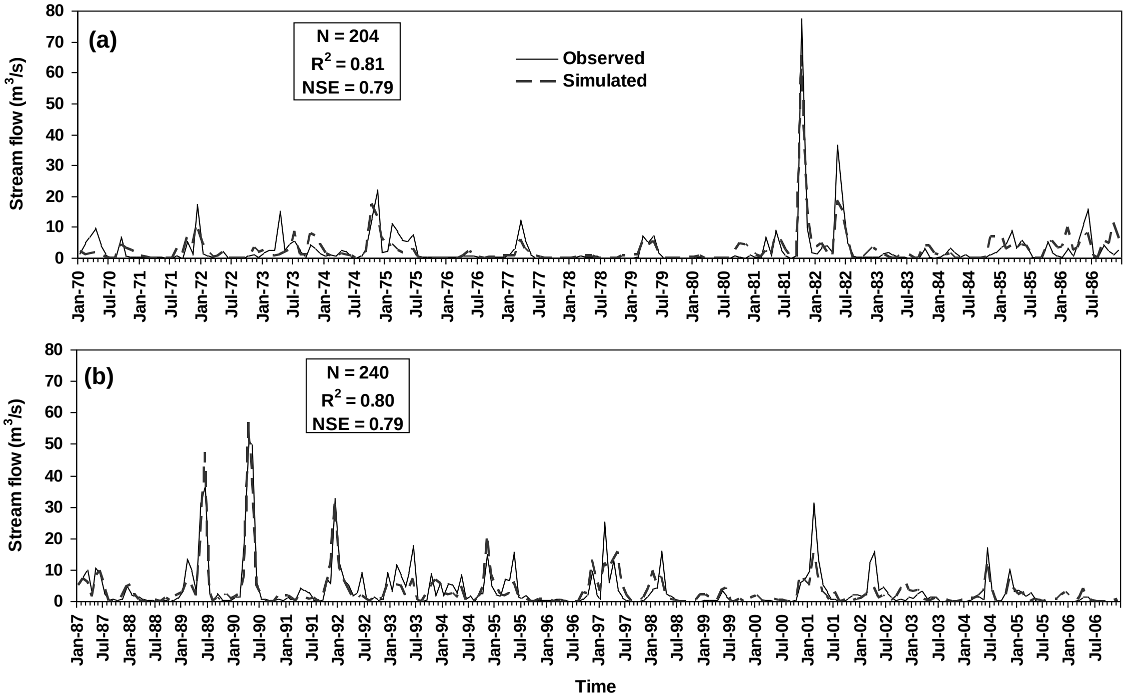


Figure 5.13 Measured and simulated monthly streamflow at Clear Creek near Sanger for the (a) calibration period (1970–1986) and (b) validation period (1987–2006).

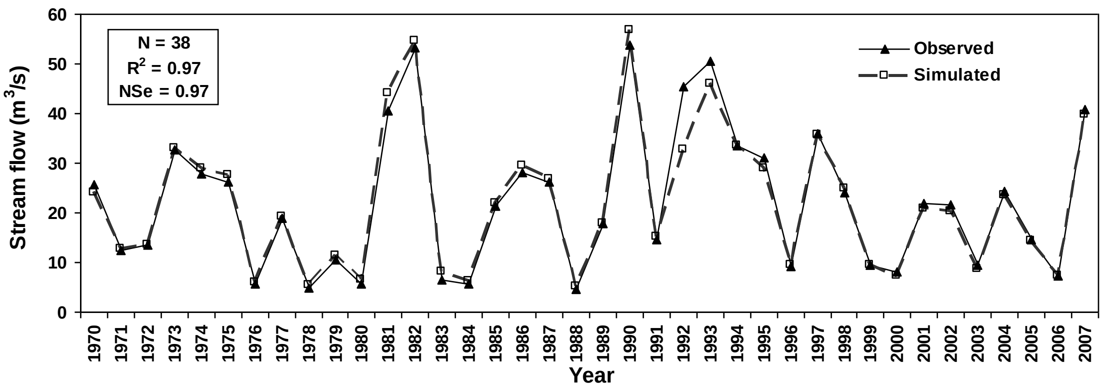


Figure 5.14 Validation of simulated yearly streamflow at the Elm Fork Trinity River near Lewisville for 1970–2007. Note: This site is downstream of the Lewisville Dam. Therefore, reservoir release, which is a SWAT input, dominates flow data for this gauge.

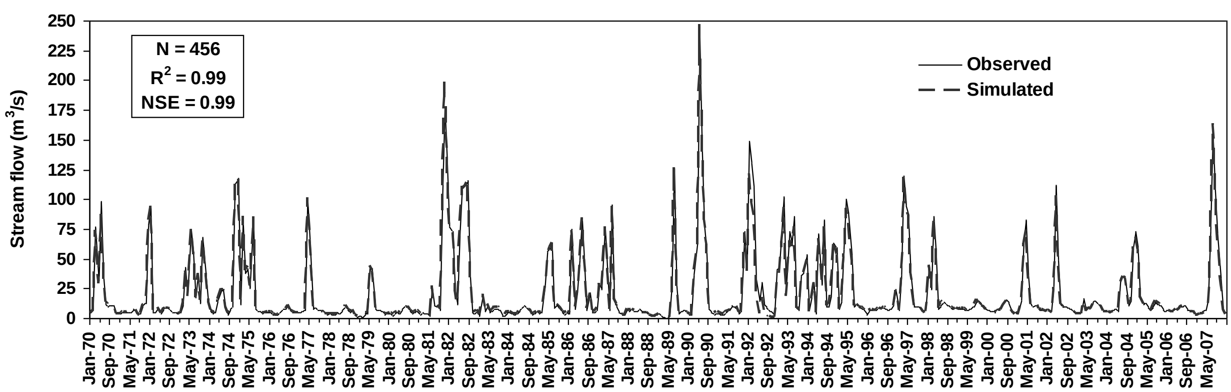


Figure 5.15 Validation of simulated monthly streamflow at the Elm Fork Trinity River near Lewisville (1970–2007). Note: This site is downstream of the Lewisville Dam. Therefore, reservoir release, which is a SWAT input, dominates flow data for this gauge.

Water Quality

We calibrated the model for average daily sediment and nutrient loading at the USGS gauging stations at Little Elm Creek near Aubrey (08052700) and Elm Fork Trinity River near Lewisville using days with available data. The Elm Fork Trinity River near Lewisville station was also used for validation. Simulated sediment, mineral phosphorus, total phosphorus, nitrate-nitrite and organic nitrogen daily loads compared well with the corresponding values measured at Little Elm Creek near Aubrey. Percent error was within about ±10% (table 5.4 and figure 5.16). However, average daily loads calibrated and validated at Elm Fork Trinity River near Lewisville were relatively weaker, with the percent error ranging from -21.1% to 10.8%.

Table 5.4 Water quality calibration and validation for days with available observations at USGS gauging stations in the Lewisville Basin (Obs. = observed data; Sim. = simulated data).

		Sediment metric tons (tons/day)		Mineral P		Total P		NO3-NO2		Organic N	
				---------- kg ---------- (lb/day)							
		Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.
Little Elm Creek near Aubrey	**Calibration** Mean	1764.0 (1943.9)	1708.6 (1882.9)	135.4 (298.6)	134.6 (296.8)	225.5 (497.2)	227.8 (502.3)	830.2 (1830.6)	744.5 (1641.6)	1208.9 (2665.6)	1196.4 2638.0
	% error		-3.1		-0.5		1.1		-10.3		-1.0
Elm Fork Trinity River near Lewisville	**Calibration** Mean	-	-	96.6 (213.0)	98.1 (216.3)	216.1 (476.5)	237.4 (523.5)	2265.9 (4996.3)	2301.4 (5074.6)	1924.3 (4243.1)	1517.8 (3346.7)
	% error		-		1.6		9.9		1.6		-21.1
	Validation Mean	-	-	87.8 (193.6)	85.1 (187.6)	261.2 (575.9)	217.8 (480.2)	1133.9 (2500.2)	1230.9 (2714.1)	957.2 (2110.6)	853.5 (1882.0)
	% error		-		-3.0		-16.6		8.6		10.8

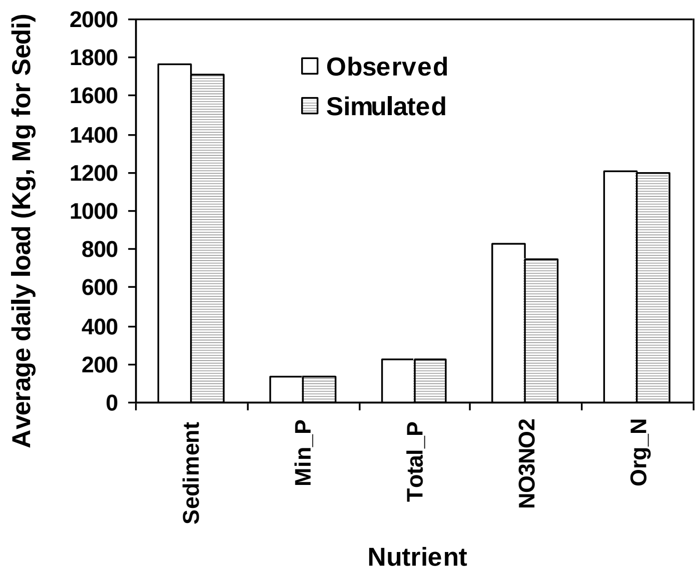


Figure 5.16 Measured and simulated average daily sediment (measured in megagrams) and nutrient loads (measured in kg) at Little Elm Creek near Aubrey.

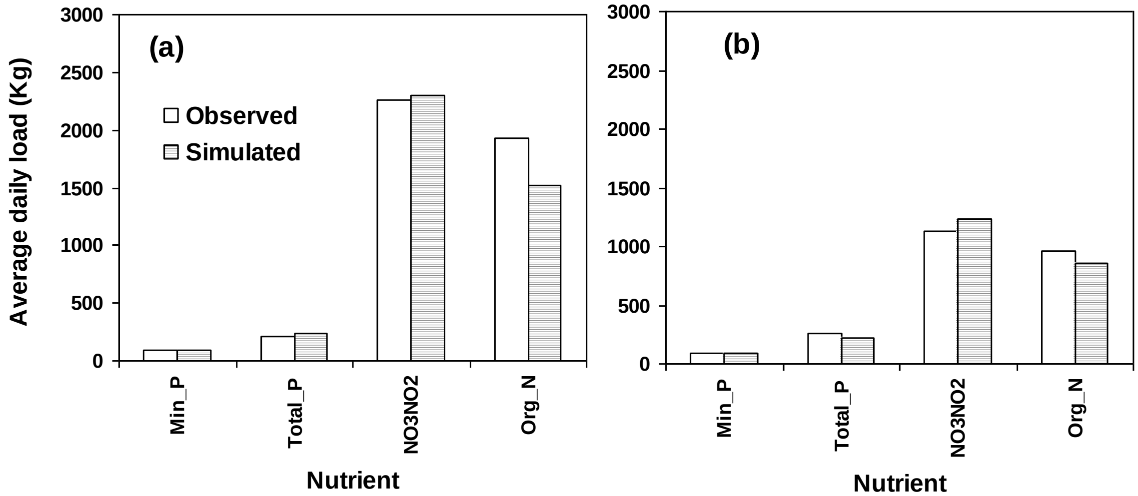


Figure 5.17 Observed and simulated total sediment and nutrient loads at Elm Fork Trinity River near Lewisville for the (a) calibration period and (b) validation period. The availability of water quality monitoring data was not the same for all variables. Thus, we used different time periods for calibration and validation of certain water quality variables.

Model Predictions

During the simulation period lasting from 1968 through 2007, the model predicted that overland flow within Lewisville Basin generated 400,796 metric tons (441,802 tons) of sediment per year under current management scenarios. According to the model, the total sediment actually reaching Lake Lewisville was about 440,800 metric tons (485,899 tons) per year, with channel banks contributing about 9% of that sediment load. The sediment derived from channel banks within Lewisville Basin (9%) was less than that reported for the Lavon Basin (58%) (see Chapter 4: Lavon Basin). Streambank erosion processes are driven by two major factors: stream bank characteristics (erodibility) and hydraulic/gravitational forces. Vegetation rooting characteristics can protect banks from fluvial entrainment and collapse and can also provide internal bank strength. The channel cover factor and channel erodibility factor used in this study were calibrated as 0.03 and 0.01, respectively, based on available sediment yield data at USGS gauge 08052700. However, in the Lake Lavon watershed, calibrated values for the channel cover factor and the channel erodibility factor were 0.6 and 0.7, respectively, based on available sediment yield data at USGS gauge 08058900 (see Chapter 4: Lavon Basin). Therefore, channel banks within Lavon Basin contributed much more sediment than those of this watershed, according to SWAT simulations. However, relative uncertainty is high due to the lack of data necessary to calibrate channel degradation and deposition of sediments. The sediment data available at USGS gauging stations are sediment loads from both overland areas and streambanks. Sediment yield from each subbasin, as simulated by SWAT, is given in figure 5.18.

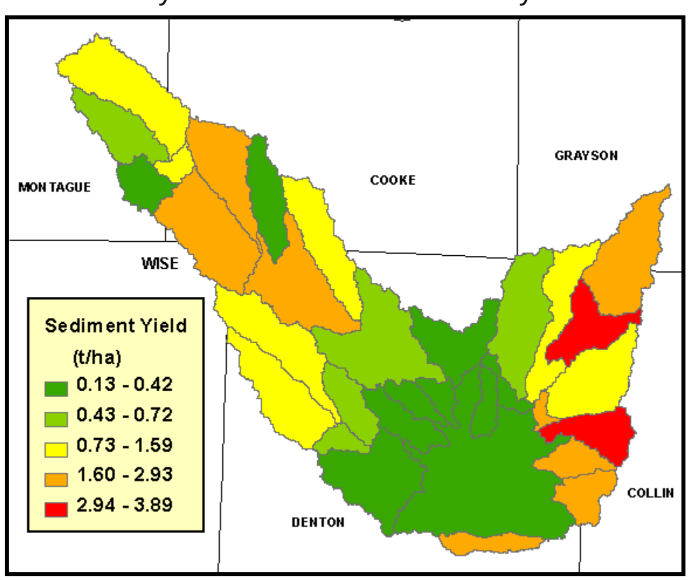


Figure 5.18 Sediment losses generated within the Lewisville Basin as predicted by the SWAT model.

Over the simulation period, the model also predicted that phosphorus and nitrogen losses from overland areas were 182,805 kilograms (403,016 pounds) per year and 1,706,778 kilograms (3,762,801 pounds) per year, respectively, under current management scenarios. The total phosphorus and total nitrogen losses from each subbasin are mapped in figures 5.19 and 5.20, respectively. According to SWAT, point sources generated 43,938 kilograms (96,867 pounds) of total phosphorus and 273,206 kilograms (602,316 pounds) of total nitrogen. Of the nutrients derived from Lewisville Basin, the model predicted that 168,080 kilograms (370,553 pounds) of total phosphorus and 2,518,920 kilograms (5,553,268 pounds) of total nitrogen reach Lake Lewisville annually.

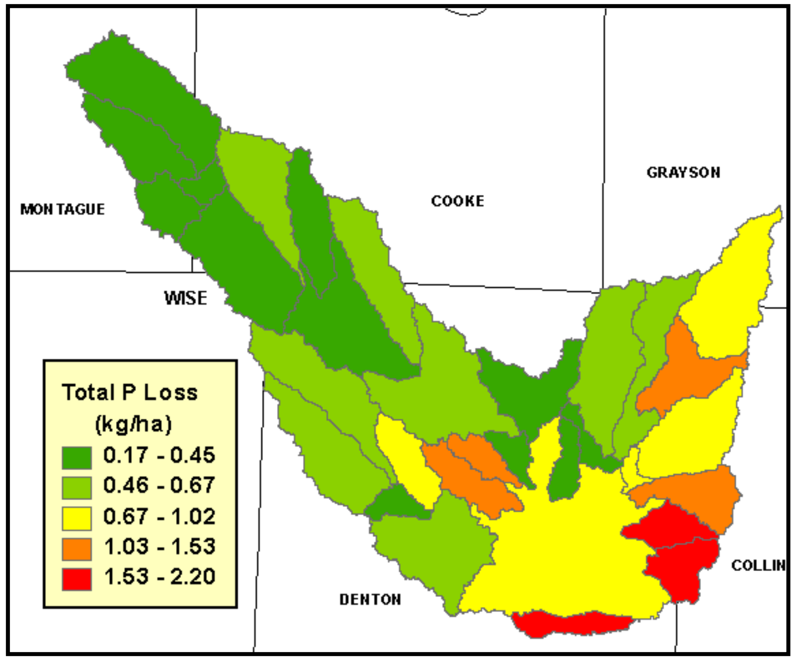


Figure 5.19 Phosphorus losses from Lewisville Basin as predicted by the SWAT model.

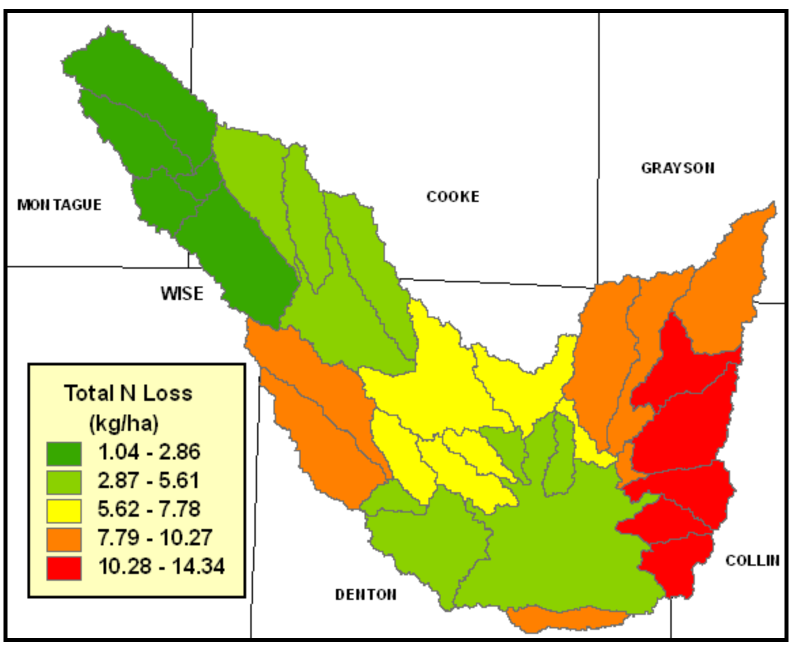


Figure 5.20 Total nitrogen losses from Lewisville Basin as predicted by the SWAT model.

Scenarios

Conservation Practices

We simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 5.5.

Ponds

According to SWAT, the 139 ponds, determined based on the National Inventory of Dams' PL-566 structures (see Chapter 2, General Model Input Data), reduce Lewisville Basin's sediment load by 48% (figure 5.21).

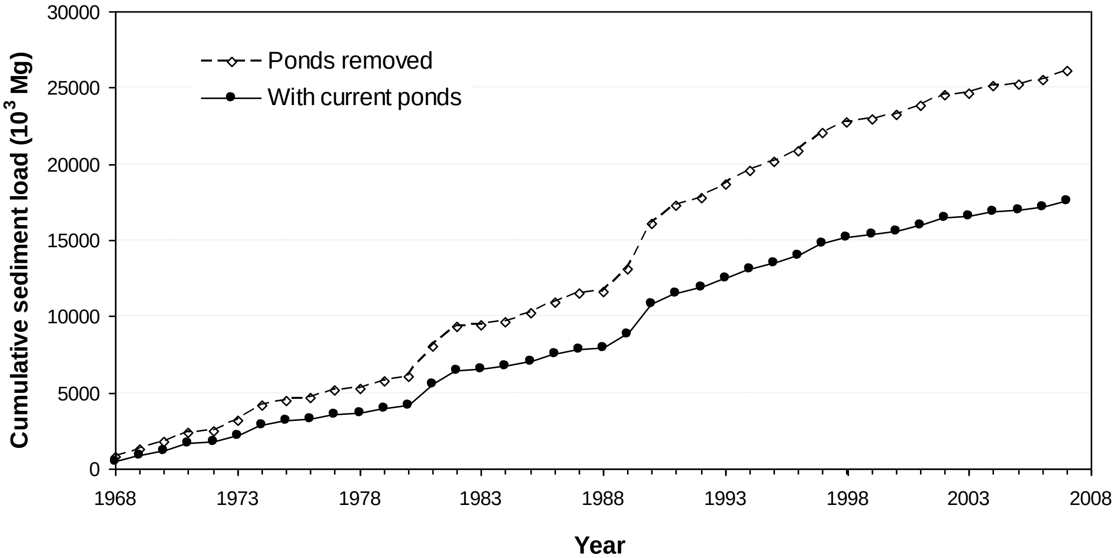


Figure 5.21 SWAT model predictions of the cumulative sediment loading of Lake Lewisville with and without currently existing ponds.

Range Utilization

Removing grazing from rangelands resulted in a 37% reduction in sediment loads from overland areas.

Point Source Load Elimination

The basin contains nineteen point sources. The elimination of these WWTPs reduced total phosphorus loads by 14% and nitrogen loads by 27%.

Urbanization

According to SWAT, urban area in the Lewisville Basin is expected to increase from 66,966 acres in 2000 to 119,599 acres in 2030, changing the percentage of urban areas from 11% to 19% (figure 5.22).

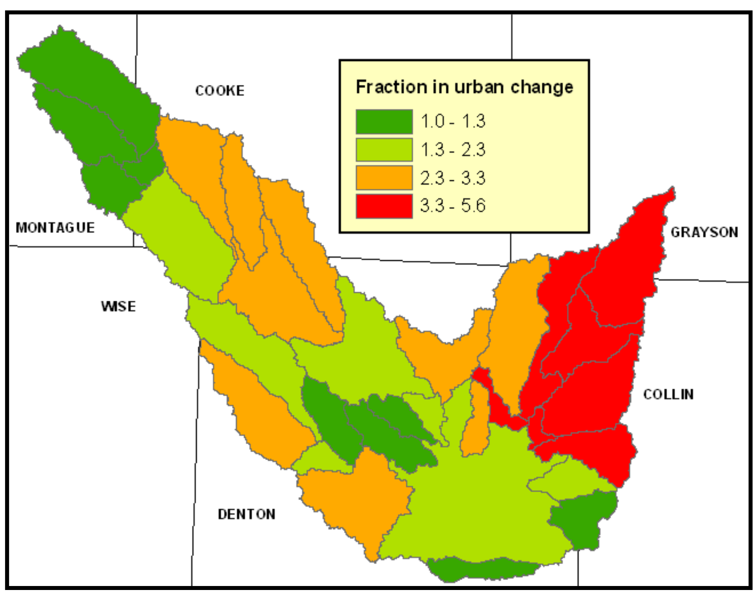


Figure 5.22 Projected changes in urban area from 2000 to 2030 (urban area in 2030 = urban area in 2000 * fraction of urban change).

Table 5.5 Changes in sediment and nutrient loading of Lake Lewisville under different scenarios as derived from SWAT model simulations (1968–2007).

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	440,800 metric tons/yr (485,899 tons/yr)	168,080 kg/yr (370,553 lb/yr)	2,518,920 kg/yr (5,553,268 lb/yr)
No Ponds	48.3%	10.0%	6.3%
No Range Grazing	-37.2%	-9.1%	-3.3%
Urban *	-2.5	23.5%	2.9%
No Point Sources	-0.2%	-14.4%	-27.2%

* load changes from upland only

Conclusions

We used the SWAT model to simulate the effects of urbanization and other land use changes on hydrologic and water quality processes in the Lewisville Basin. Using flow data available in the watershed, SWAT calibration and validation resulted in a modeled representation of the watershed that was within acceptable standards. NSE values based on monthly flow comparisons ranged from 0.74 to 0.99, and yearly comparisons ranged from 0.83 to 0.97. We calibrated water quality components of the SWAT model using data from Little Elm Creek near Aubrey and Elm Fork Trinity River near Lewisville. In addition, the later gauge was also used to validate the model for water quality. In general, simulated average daily sediment, mineral phosphorus, total phosphorus, nitrate-nitrite and organic nitrogen loads were within ±10% of the corresponding observed values. Only total phosphorus and organic nitrogen from Elm Fork Trinity River near Lewisville did not compare well with observed values.

During the simulation period lasting from 1968 to 2007, the model predicted that the total sediment reaching Lake Lewisville was 440,800 metric tons (485,899 tons) per year with channel banks contributing about 9% of the sediment load. The total phosphorus and nitrogen loads from overland areas were 182,805 kilograms (403,016 pounds) per year and 1,706,778 kilograms (3,762,801 pounds) per year, respectively, under current management scenarios. Of the nutrients resulting from overland areas and point sources, the model predicted that 168,080 kilograms (370,553 pounds) of total phosphorus and 2,518,920 kilograms (5,553,268 pounds) of total nitrogen actually reach Lake Lewisville annually.

In the scenario analyses, the complete removal of ponds, point sources and rangeland management demonstrated their overall contribution to the water quality of Lewisville Basin. Based on model predicted values, the removal of ponds would increase reservoir loading with sediment by approximately 48%, total phosphorus by 10% and total nitrogen by 6%. Removing point sources achieved nutrient reductions of approximately 14% in total phosphorus and 27% in total nitrogen. Finally, removing grazing from rangelands reduced the sediment load by approximately 37%, total phosphorus by 9% and total nitrogen by 3%. SWAT also predicted the effects of urban expansion in 2030. In the urbanization projections, urban areas are expected to increase from 11% in 2000 to 19% in 2030. Associated with this expansion, the model predicted approximate increases of 23% in total phosphorus and 3% in total nitrogen.

References

See the appendix.

Chapter 6: Ray Roberts Basin

Introduction

Lake Ray Roberts was completed in 1986. It is located in and supplies water to Cooke, Grayson, and Denton counties. Ray Roberts Basin lies within the upper part of Hydrologic Unit Code 12030103 and encompasses an area of 442,319 acres in the Trinity River Basin in Texas (figure 6.1). The primary land use within the basin is rangeland.

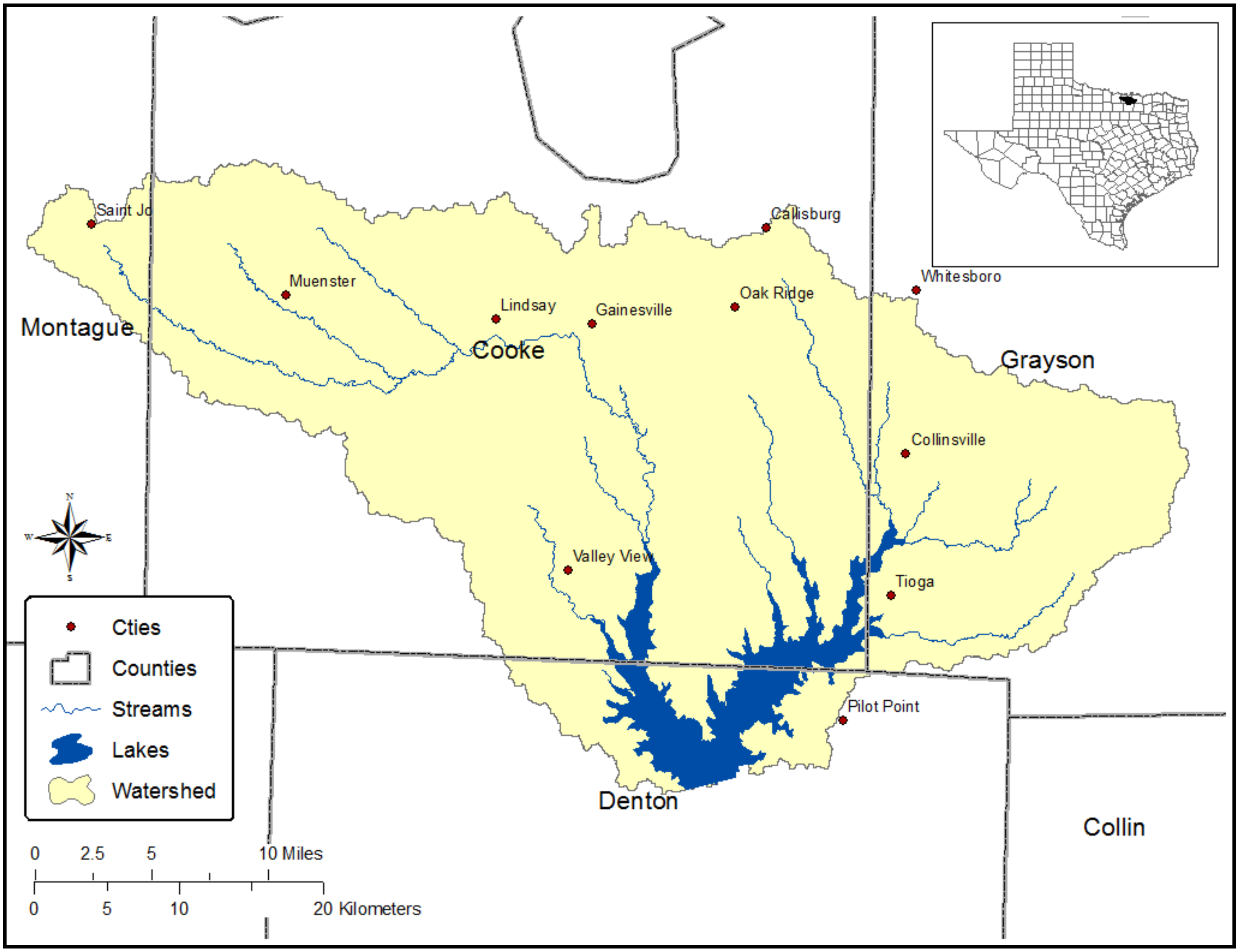


Figure 6.1 Location of Ray Roberts Basin.

Model Input Data Tables and Figures

Topography

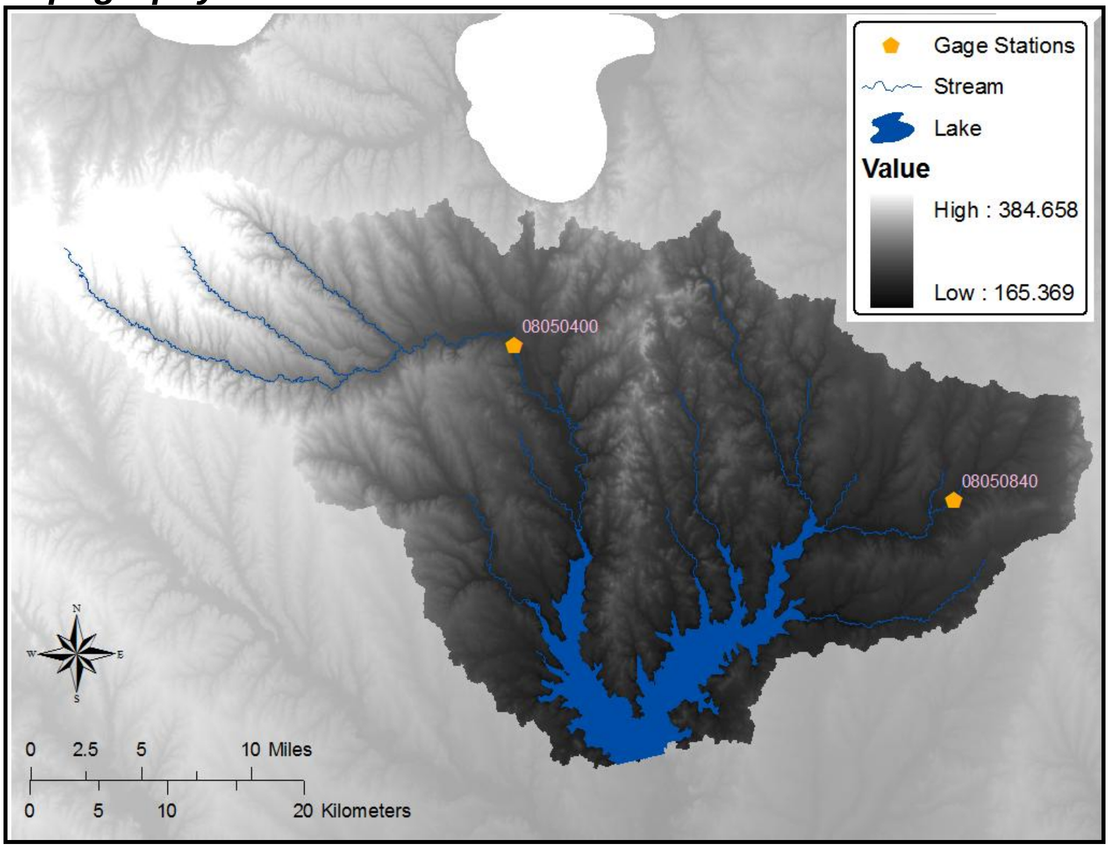


Figure 6.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of Ray Roberts Basin.

Soils

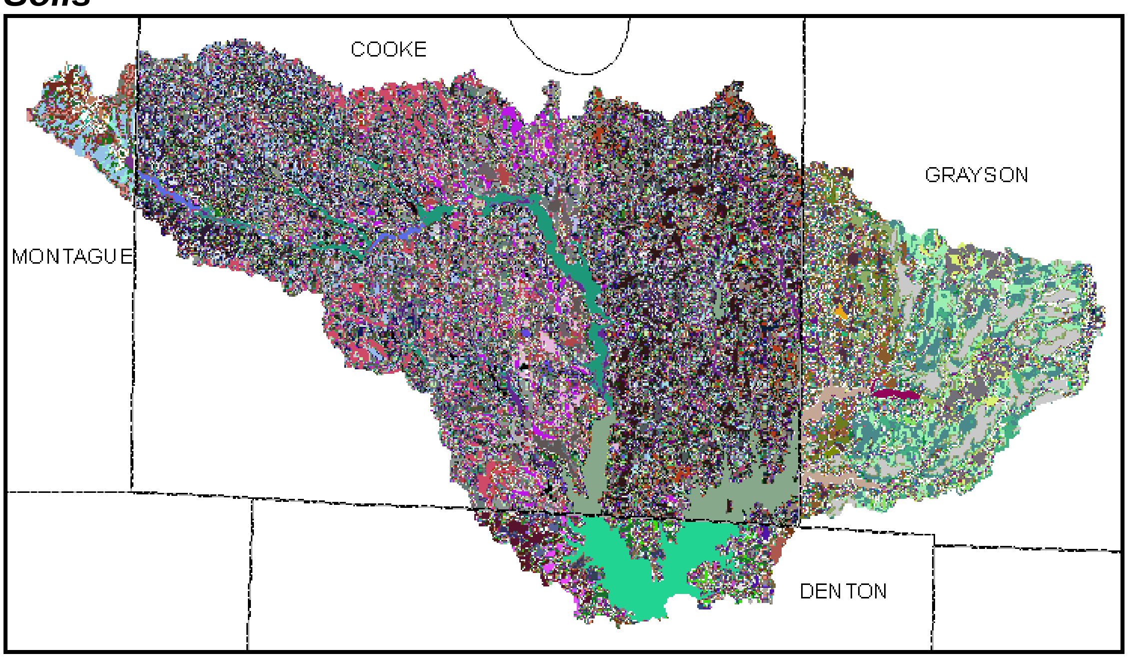


Figure 6.3 Soil Survey Geographic (SSURGO) data was used to define soil attributes in SWAT.

Land use

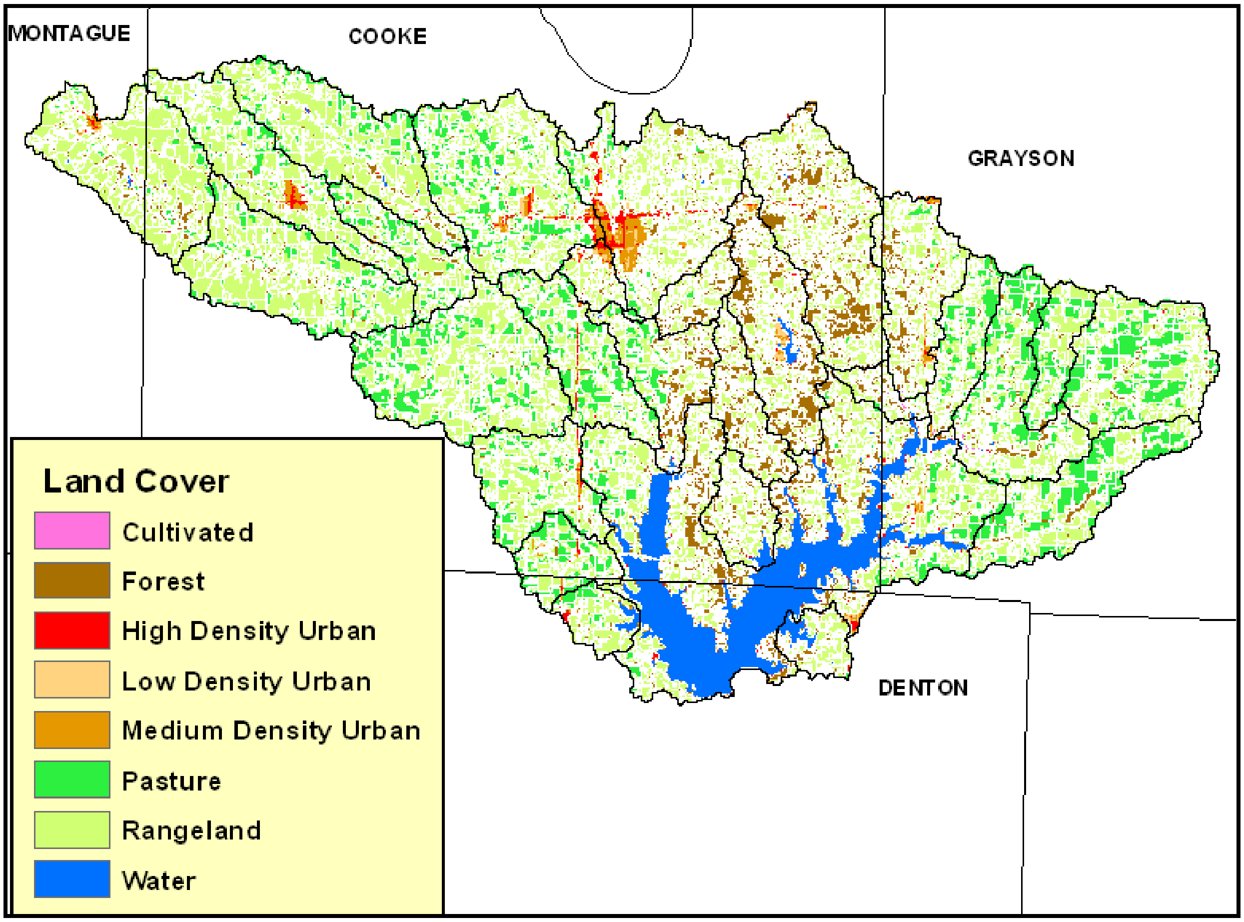


Figure 6.4 National Land Cover Data (2001) defined land use inputs for the Ray Roberts Basin.

Weather

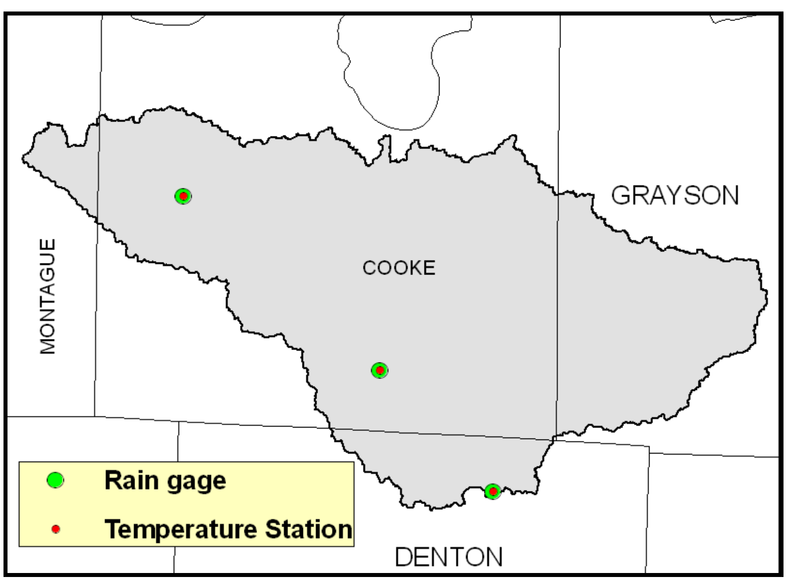


Figure 6.5 Three precipitation and temperature stations located in and around Ray Roberts Basin provided daily rainfall and temperature values (maximum and minimum) for the SWAT simulation model.

Point Sources

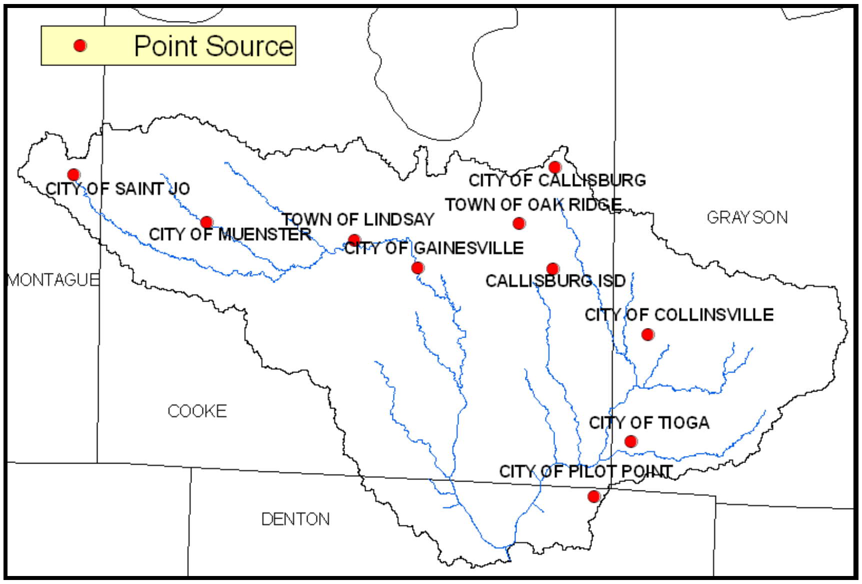


Figure 6.6 The USGS Water Resource database provided SWAT input data on the ten actively operating point sources within the Ray Roberts Basin. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

Reservoir

Table 6.1 The Ray Roberts Basin contains one large reservoir, Lake Ray Roberts, which began operations in 1987. SWAT utilized available daily reservoir outflow data and reservoir characteristics from the National Inventory of Dams.

Reservoir	Subbasin	Surface area at principle spillway (acres)	Volume at principle spillway (10^4 acre-feet)	Surface area at emergency spillway (acres)	Volume at emergency spillway (10^4 acre-feet)	Release
Ray Roberts	28	29,349	80.0	54,848	149.4	Measured

Subbasin Delineation

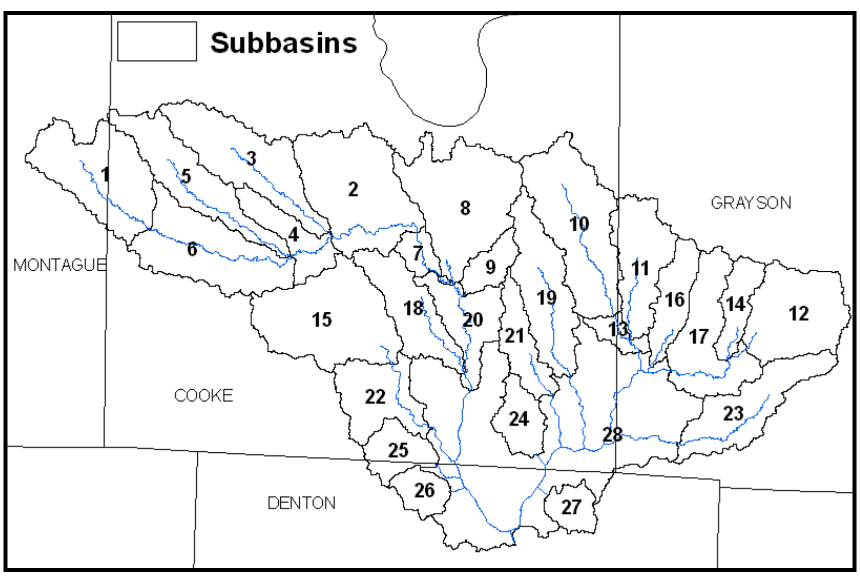


Figure 6.7 The Ray Roberts SWAT model used a stream threshold value of 4,448 acres to delineate subbasins, resulting in 28 subbasins.

HRU Distribution

SWAT split the 28 subbasins into HRUs based on researcher-defined thresholds, resulting in 1,570 HRUs.

Ponds

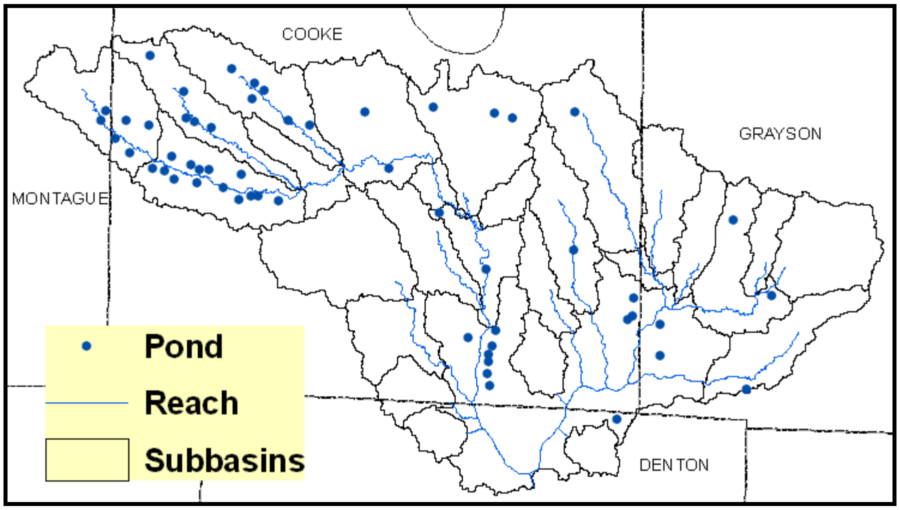


Figure 6.8 The U.S. Army Corps of Engineers' National Inventory of Dams provided SWAT with the location of ponds in the Ray Roberts Basin. According to SWAT, approximately 18% of the basin area drains to ponds.

Model Calibration and Validation

Model simulation began in 1960. To determine the appropriate baseflow proportion, we used the Baseflow Filter Program (Arnold and Allen, 1999; Arnold et al., 1995), analyzing streamflow records at two USGS gauging stations: Elm Fork Trinity River at Gainesville (08050400, outlet of subbasin two) and Range Creek near Collinsville (08050840, outlet of subbasin 12) (figure 6.9). During calibration, we carefully matched the proportions of surface flow and baseflow. We calibrated SWAT for annual and monthly streamflow using flow records from the Gainesville station (1985–1995) and the station near Collinsville (1992–1997). We then validated the model for flow at Gainesville (1996–2007) and Collinsville (1992–2004).

One available monitoring station at Range Creek near Collinsville provided water quality calibration data including mineral phosphorus, total phosphorus, nitrate-nitrite and organic nitrogen (figure 6.9). Due to limited available data, all water quality data were used for calibrating nutrient loads. Calibrated parameters are listed in table 6.2.

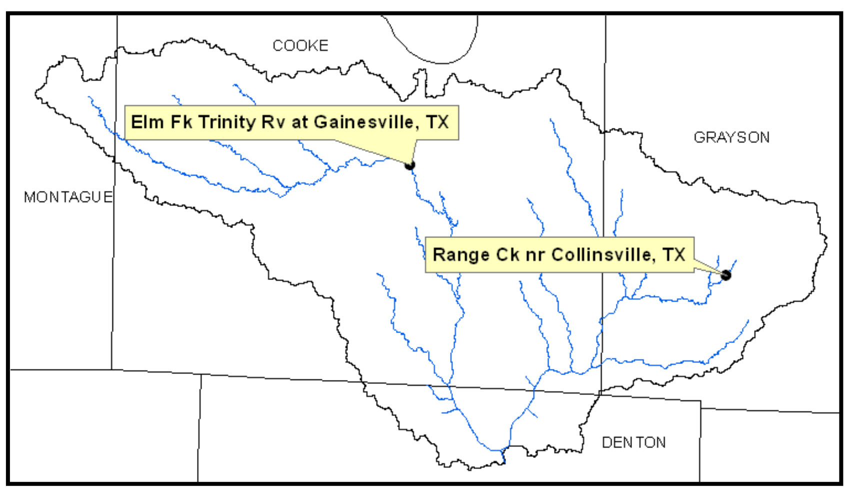


Figure 6.9 USGS gauges used for calibration and validation in the Ray Roberts Basin.

Table 6.2 Model parameters and ranges used for calibration and final calibrated values.

Component	Parameter (file)	Description	Range	Calibrated value
Flow	CN2 (.mgt)	Initial NRCS runoff curve number for moisture condition II	-5– 5	- 4.0–0.0
	ESCO (.bsn)	Soil evaporation compensation factor	0.01– 1.0	0.8
	EPCO (.bsn)	Plant uptake compensation factor	0.01–1.0	0.8
	GW_REVAP (.gw)	Groundwater revap coefficient	0.02– 0.4	0.3
	GWQMN (.gw)	Threshold depth of water in the shallow aquifer required for return flow to occur	0.0–300.0	0.0
	REVAPMN (.gw)	Threshold depth of water in the shallow aquifer for "revap" to occur [mm]	1.0–15.0	1.0
Sediment	SPCON (.bsn)	Linear parameter for estimating the maximum amount of sediment that can be reentrained during channel sediment routing	0.0001– 0.01	0.006
	CH_COV (.rte)	Channel cover factor	0.0–1.0	0.02
	CH_EROD (.rte)	Channel erodibility factor	0.0–1.0	0.02
Nutrients	RHOQ (.wwq)	Algal respiration rate at 20ºC (day^{-1})	0.05–0.50	0.3
Mineral nitrogen	SDNCO (.bsn)	Denitrification threshold water content (fraction of field capacity water content above which denitrification takes place)		1.2
	NPERCO (.bsn)	Nitrate percolation coefficient		0.18
Nitrogen in reach	AI1 (.wwq)	Fraction of algal biomass that is nitrogen	0.07–0.09	0.08
	RS4 (.swq)	Rate coefficient for organic N settling in the reach at 20ºC (day^{-1})		0.05–0.4
	BC2 (.swq)	Rate constant for biological oxidation of NO_2 to NO_3 in the reach at 20ºC (day^{-1})	0.2–2.0	2.0
	BC3 (.swq)	Rate constant for hydrolysis of organic N to NH_4 in the reach at 20ºC (day^{-1})		0.21
Mineral phosphorus	PPERCO (.bsn)	Phosphorus percolation coefficient	10.0–17.5	10
	PHOSKD (.bsn)	Phosphorus soil partitioning coefficient	100–175	175
Phosphorus in reach	AI2 (.wwq)	Fraction of algal biomass that is phosphorus	0.01–0.02	0.016
	BC4 (.swq)	Rate constant for mineralization of organic P to dissolved P in the reach at 20ºC (day^{-1})		0.45
	RS5 (.swq)	Organic phosphorus settling rate in the reach at 20ºC (day^{-1})		0.001

Results and Discussion

Model Calibration and Validation

Flow

Simulated flow for both calibration and validation periods was similar to corresponding data observed at USGS gauging stations (figures 6.10-6.13), as demonstrated by NSE and R^2 values. Calibration and validation of annual streamflow at Elm Fork Trinity River (08050400) and Range Creek near Collinsville (08050840) resulted in R^2 values ranging from 0.76 to 0.95 and NSE values from 0.65 to 0.90 (table 6.3). A monthly comparison resulted in R^2 and NSE values ranging from 0.72 to 0.85 (table 6.3).

Table 6.3 Annual and monthly streamflow (m^3/s) calibration and validation results at both USGS gauging stations.

Gauge station ID (site name)		Period	Measured			Simulated			Yearly		Monthly	
			Mean	Std_y	Std_m	Mean	Std_y	Std_m	R^2	NSE	R^2	NSE
08050400 (Elm Fork Trinity River at Gainesville)	Calibration	1985–1995	**3.66**	2.15	5.93	**3.42**	1.77	4.58	**0.91**	**0.88**	**0.85**	**0.82**
	Validation	1996–2007	**2.01**	1.30	3.80	**2.56**	1.51	3.67	**0.89**	**0.65**	**0.83**	**0.81**
08050840 (Range Creek near Collinsville)	Calibration	1992–1999	**0.56**	0.20	1.00	**0.50**	0.20	0.77	**0.76**	**0.67**	**0.73**	**0.72**
	Validation	2000–2004	**0.37**	0.27	0.75	**0.41**	0.22	0.81	**0.95**	**0.90**	**0.77**	**0.73**

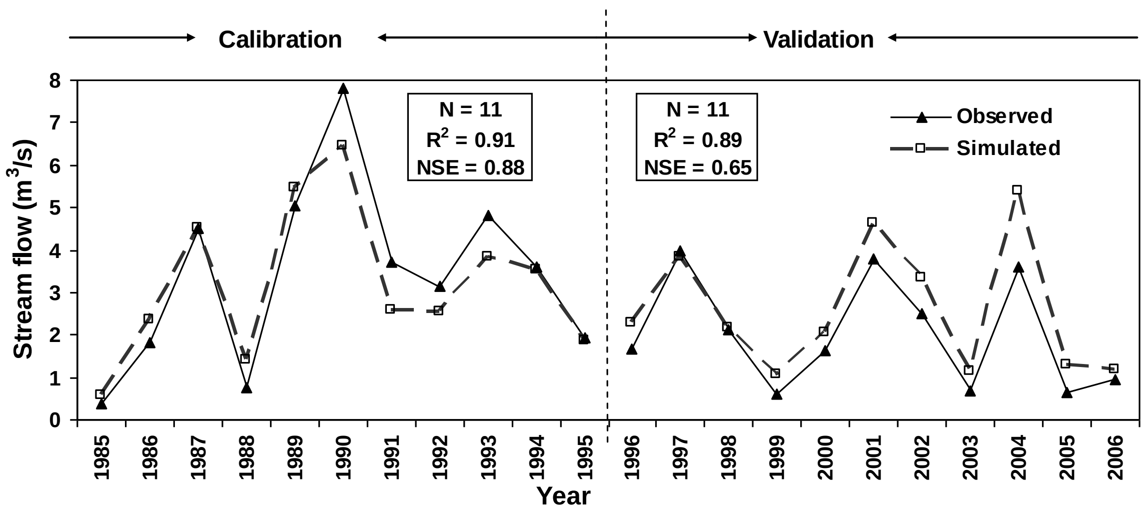


Figure 6.10 Measured and simulated yearly streamflow at USGS gauge 08050400 (Elm Fork Trinity River at Gainesville) for the calibration (1985–1995) and validation periods (1996–2007).

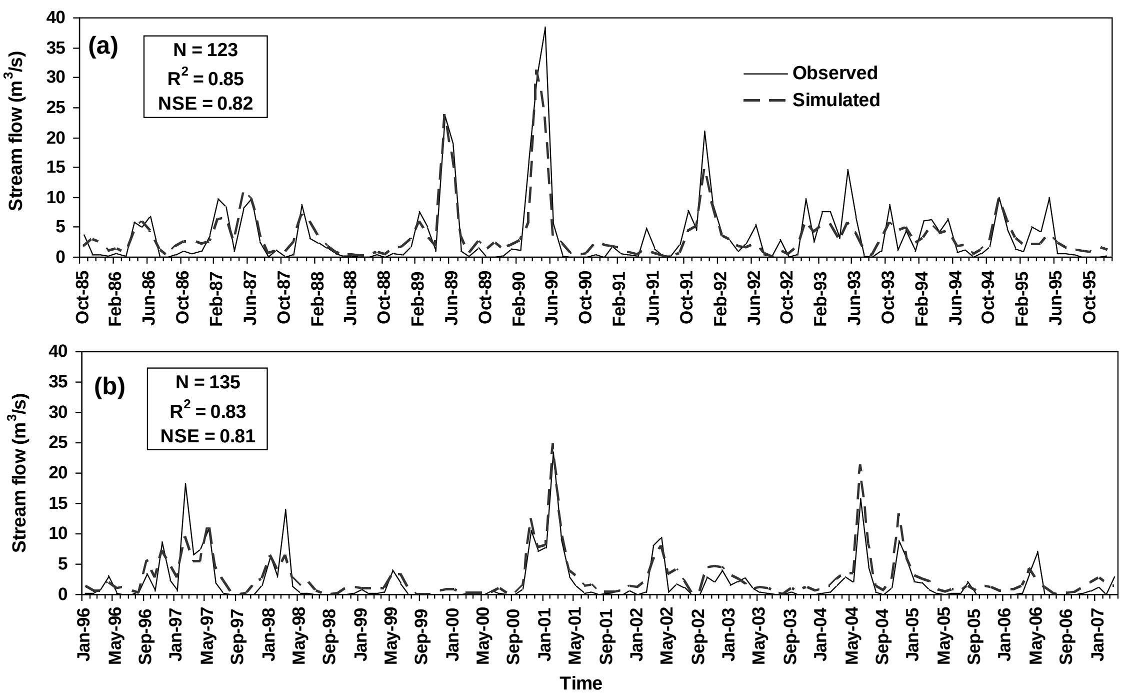


Figure 6.11 Measured and simulated monthly streamflow at USGS gauge 08050400 (Elm Fork Trinity River at Gainesville) for the (a) calibration period (10/1985–12/1995) and (b) validation period (1/1996–3/2007).

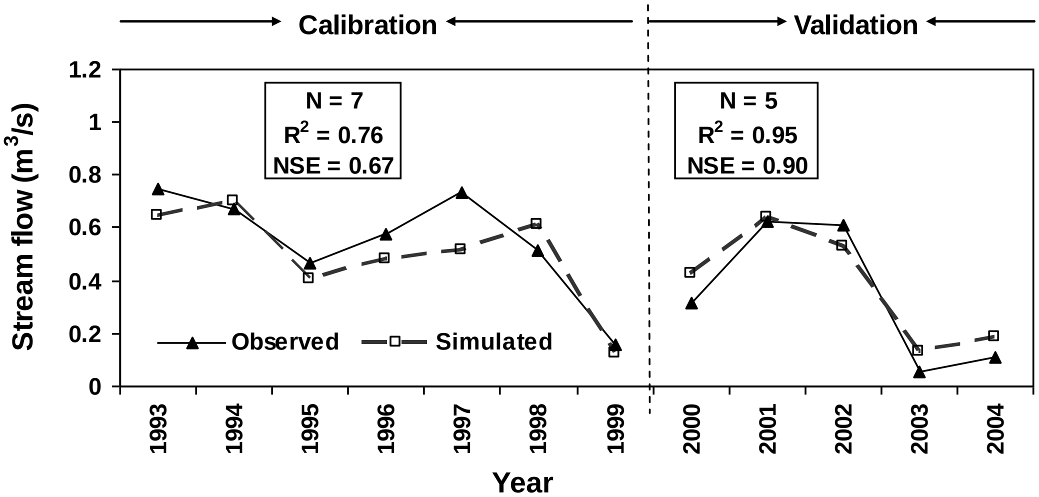


Figure 6.12 Measured and simulated yearly streamflow at USGS gauge 08050840 (Range Creek near Collinsville) for the calibration (1993–1999) and validation periods (2000–2004).

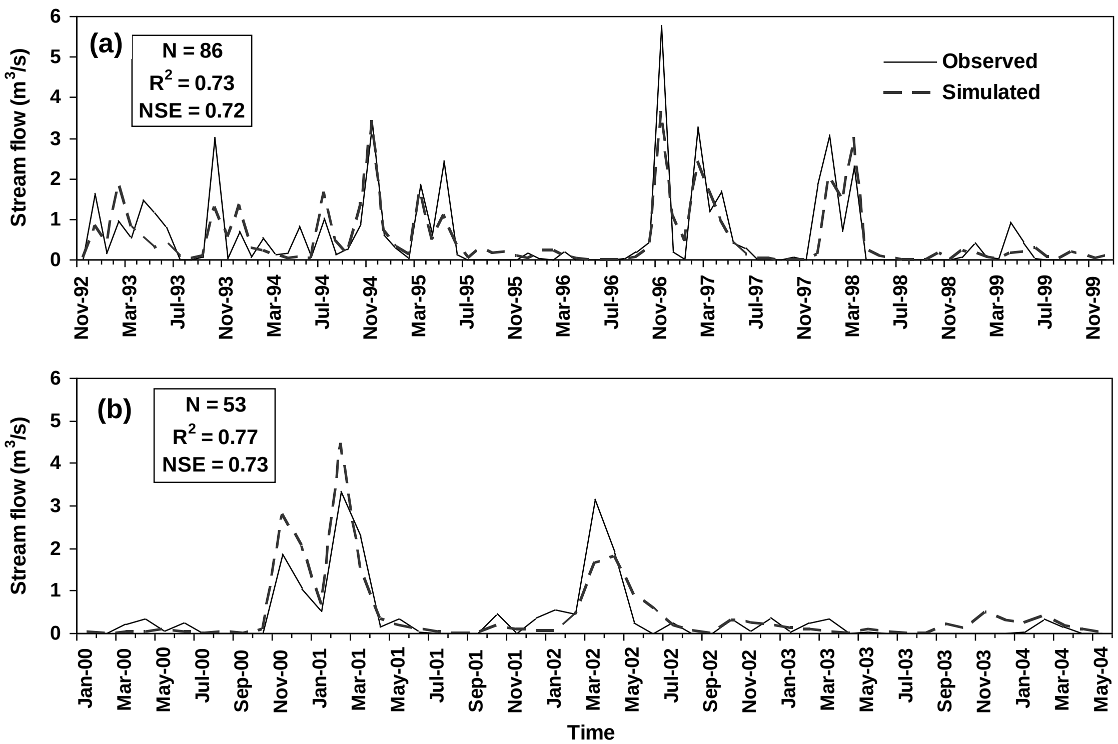


Figure 6.13 Measured and simulated monthly streamflow at USGS gauge 08050840 (Range Creek near Collinsville) for the (a) calibration period (11/1992–12/1999) and (b) validation period (1/2000–4/2004).

Water Quality

Insufficient water quality data were available for the Ray Roberts Basin. At USGS gauge 08050840, only 17 daily water quality sampling data were available, and USGS gauge 08050400 provided only three. For days with available data, we used average daily loads measured at USGS gauging station 08050840 to calibrate the SWAT model. In general, simulated mineral phosphorus, nitrate-nitrite and organic nitrogen loads compared well with corresponding measured values, as the percent error was within ±10% (table 6.4). However, total phosphorus was clearly under-predicted. Additional monitoring data would be very helpful in adequately calibrating and validating model predicted loadings.

Table 6.4 Water quality calibration for days with available data at the USGS gauging station at Range Creek near Collinsville.

	Mineral P		Total P		NO3-NO2		Organic N	
	Observed	Simulated	Observed	Simulated	Observed	Simulated	Observed	Simulated
Mean in kg/day (lb/day)	16.6 (36.6)	17.5 (38.6)	1453.6 (3205.2)	1110.4 (2448.4)	101.5 (223.8)	95.5 (210.6)	3097.8 (6830.6)	3125.0 (6890.6)
% error		5.6		-23.6		-5.9		0.9

Model Predictions

During the simulation period lasting from 1987 through 2007, the model predicted that Ray Roberts Basin generated 76,748 metric tons (84,600 tons) of sediment per year under current management scenarios. According to SWAT, the total sediment load reaching Ray Roberts Reservoir is about 73,860 metric tons (81,417 tons) per year. Sediment is generated non-uniformly across the basin, but in general, overland range areas with steeper slopes produced higher sediment yields (figure 6.14).

From 1987 through 2007, the model predicted that phosphorus and nitrogen losses from Ray Roberts Basin were 79,854 kilograms (176,048 pounds) per year and 743,769 kilograms (1,639,730 pounds) per year, respectively, under current management scenarios. The total phosphorus and nitrogen losses from each subbasin are mapped in figures 6.15 and 6.16, respectively. Basin point sources generated 9,477 kilograms (20,893 pounds) of total phosphorus and 16,575 kilograms (36,542 pounds) of total nitrogen. The model predicted that, of the nutrients produced within Ray Roberts Basin, 103,590 kilograms (228,376 pounds) of total phosphorus and 646,060 kilograms (1,424,318 pounds) of total nitrogen actually reach Ray Roberts Reservoir annually.

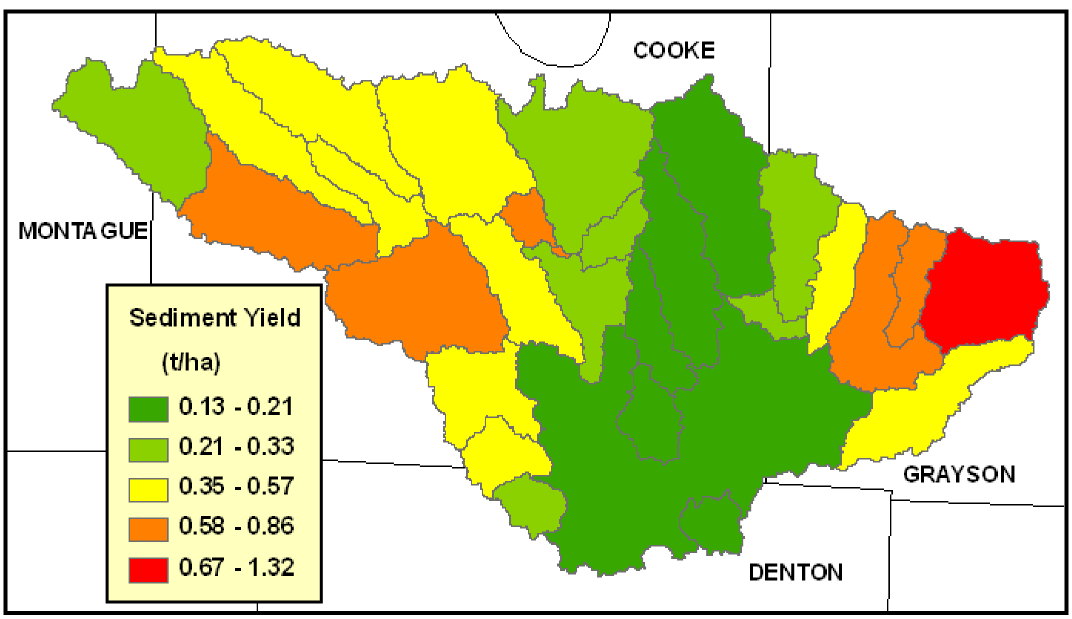


Figure 6.14 Sediment losses from Ray Roberts Basin as predicted by the SWAT model.

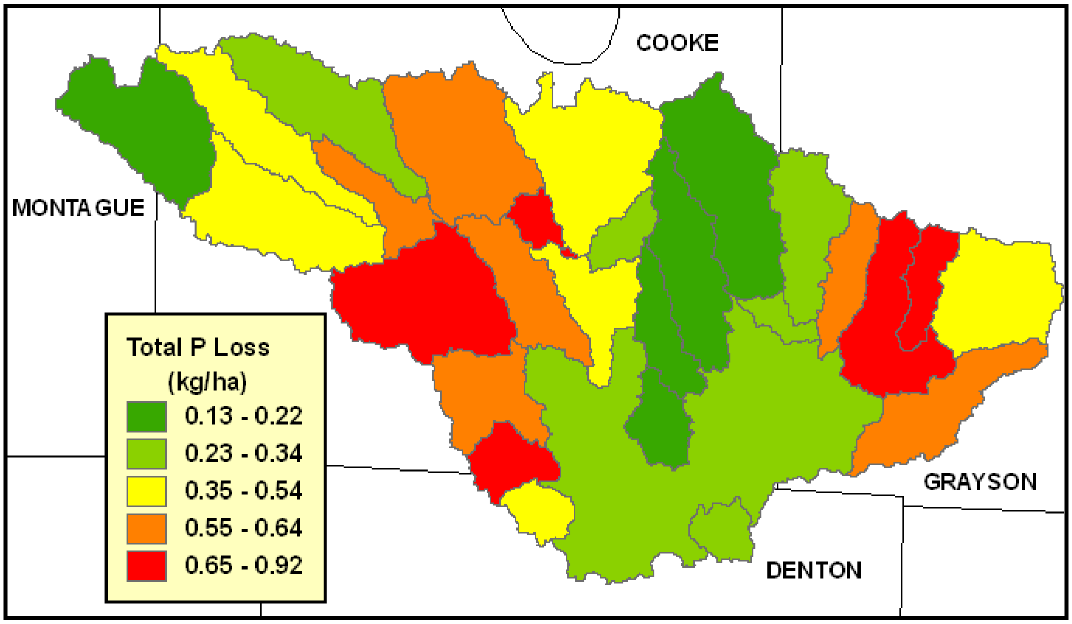


Figure 6.15 Phosphorus losses from Ray Roberts Basin as predicted by the SWAT model.

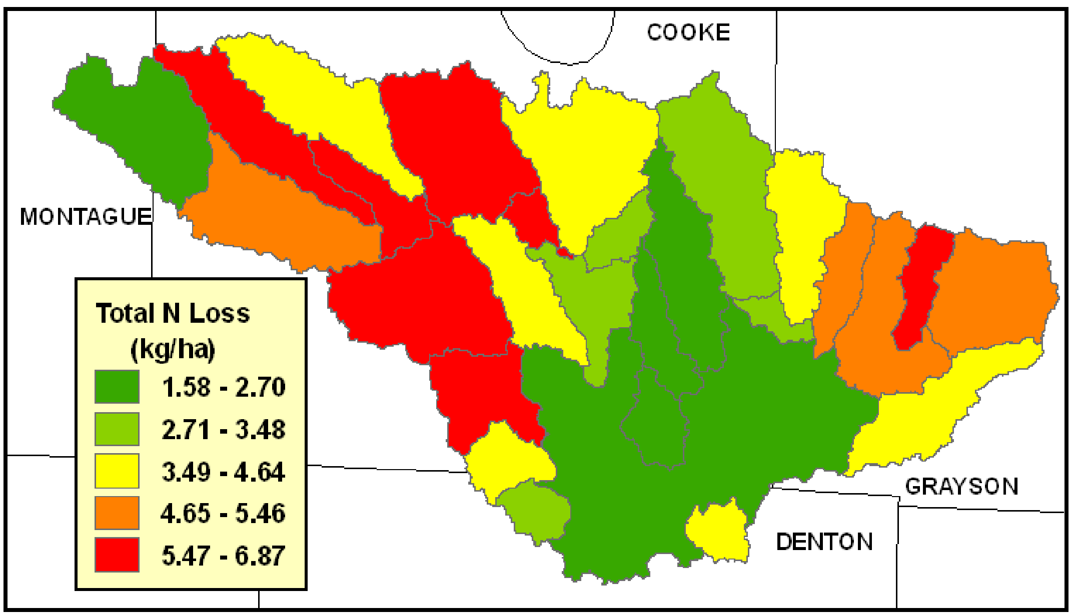


Figure 6.16 Total nitrogen losses from Ray Roberts Basin as predicted by the SWAT model.

Scenarios

Conservation Practices

SWAT simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 6.5.

Ponds

According to SWAT, ponds within Ray Roberts Basin, determined based on the National Inventory of Dams' PL-566 structures (see Chapter 2, General Model Input Data), reduce sediments by 19.4%. When modeling the removal of Ray Roberts Basin's 56 ponds, sediment (figure 6.17) and nutrient loads increased.

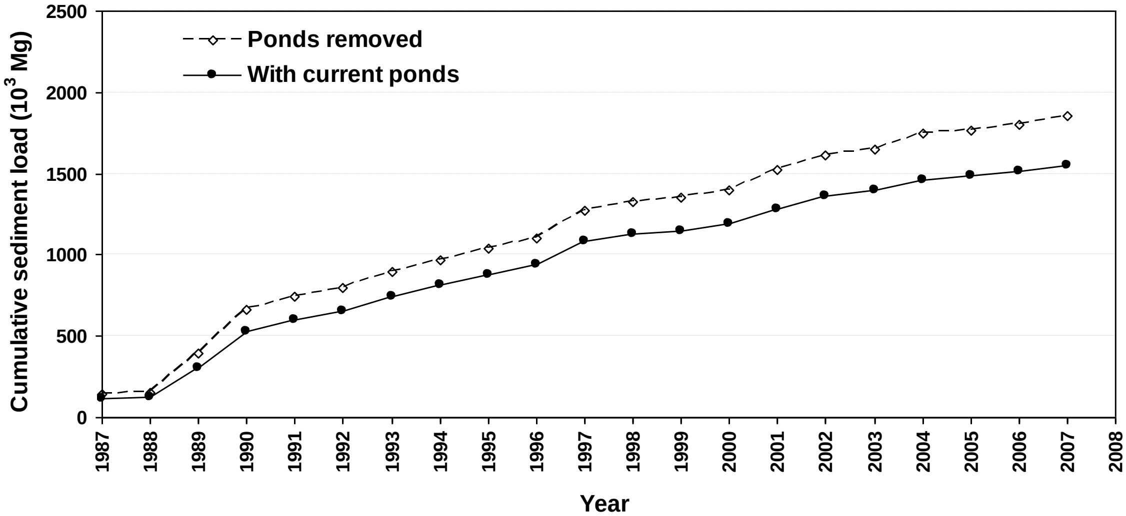


Figure 6.17 SWAT-predicted cumulative sediment loads received by Ray Roberts Reservoir with and without currently existing ponds.

Range Utilization

Removing grazing from rangelands resulted in a 3% reduction in sediment loading of the Ray Roberts Reservoir (table 6.5).

Point Source Load Elimination

The basin contains ten point sources. The elimination of these WWTPs reduced both total phosphorus and total nitrogen loading of the Ray Roberts Reservoir by 6%.

Urbanization

The North Central Texas Council of Governments expects portions of the Ray Roberts Basin to urbanize in the near future. However, population projection data only cover a small part of Denton County. We summarized increases in urban area at the subbasin level (figure 6.18), so for subbasins not covered in the population projections, we assumed the fraction of urban change to be 1.1–2.3. This resulted in an

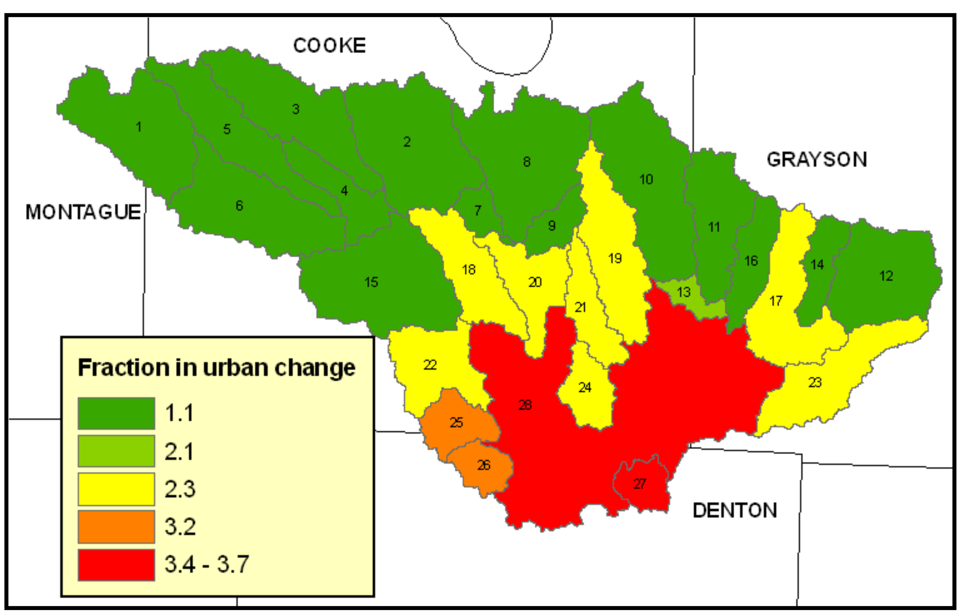


Figure 6.18 Projected change in urban area from 2000 to 2030 (urban area in 2030 = urban area in 2000 * fraction of urban change).

increase in urban area from 27,429 acres in 2000 to 46,209 acres in 2030. Stated differently, urban area in the Ray Roberts Basin is expected to increase from 6% to 11%. Coinciding with urban expansion, SWAT predicted increases in sediment (1%), total phosphorus (17%) and total nitrogen (5%) loads from overland areas (table 6.5).

Table 6.5 Changes in sediment and nutrient loading of Lake Ray Roberts under different scenarios, as derived from SWAT model simulations (1987–2007).

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	73,860 metric tons/yr (81,417 tons/yr)	103,590 kg/yr (228,377 lb/yr)	646,060 kg/yr (1,424,318 lb/yr)
No Ponds	19.4%	6.4%	7.9%
No Range Grazing	-2.9%	-5.0%	-3.1%
Urban *	1.0%	16.6%	4.7%
No Point Sources	-0.1%	-6.0%	-5.9%

* load changes from overland only

Conclusions

We used the SWAT model to simulate the effects of urbanization and other land use changes on hydrologic and water quality processes in the Ray Roberts Basin. Using flow data from USGS gauging stations 08050840 and 08050400, SWAT calibration and validation resulted in a modeled representation of the watershed that was within acceptable standards. For example, NSE values based on monthly flow comparisons ranged from 0.72 to 0.82. While, yearly flow comparisons ranged from 0.65 to 0.90. Due to limited water quality data, all data from USGS gauging station 08050840 were used to calibrate SWAT for sediments and nutrients. We compared total loads from days with available data (measured) with SWAT simulated data. Simulated mineral phosphorus, nitrate-nitrite and organic nitrogen loads were within ±10% of their corresponding values. However, total phosphorus was under-predicted with a percent error of -24%.

During the model simulation period lasting from 1987 through 2007, total sediment reaching Lake Ray Roberts was about 73,860 metric tons (81,417 tons) per year. The model also predicted that 103,590 kilograms (228,377 pounds) of total phosphorus and 646,060 kilograms (1,424,318 pounds) of total nitrogen reach Ray Roberts Reservoir annually.

In the scenario analyses, the complete removal of ponds, point sources and rangeland management demonstrated their overall contribution to the water quality of Lake Ray Roberts. Based on model predicted values, the removal of ponds would increase the sediment load received by the Ray Roberts Reservoir by 19% while total phosphorus and total nitrogen would increase by approximately 6% and 8%, respectively. Removing point sources achieved reductions of approximately 6% in both total phosphorus and total nitrogen received by the Ray Roberts Reservoir. Finally, removing grazing from rangelands reduced the sediment load by approximately 3%, total phosphorus by 5% and total nitrogen by 3%. SWAT also predicted the effects of urban expansion in 2030. In the urbanization projections, urban areas are expected to increase from 6% in 2000 to 11% in 2030. Associated with this expansion, the model predicts increases of approximately 17% in total phosphorus and 5% in total nitrogen resulting from overland areas.

References

See the appendix.

Chapter 7: Ray Hubbard Basin

Introduction

Lake Ray Hubbard is situated in the Ray Hubbard Basin, which is located in the lower part of Hydrologic Unit Code 1203010. The basin encompasses an area of 224,125 acres in the Trinity River Basin in Texas. The watershed covers parts of Dallas, Kaufman, Collin and Rockwall counties (figure 7.1). Lake Ray Hubbard is on the East Fork of the Trinity River, just east of Dallas. This lake was built for municipal water supply but also features a lakeside power generating plant. Much of the north end retains its original standing timber. The watershed modeling objective of this project was to use the Soil and Water Assessment Tool (SWAT) to assess the effects of urbanization and other land use changes on sediment and nutrient delivery to Lake Ray Hubbard in the Trinity River Basin.

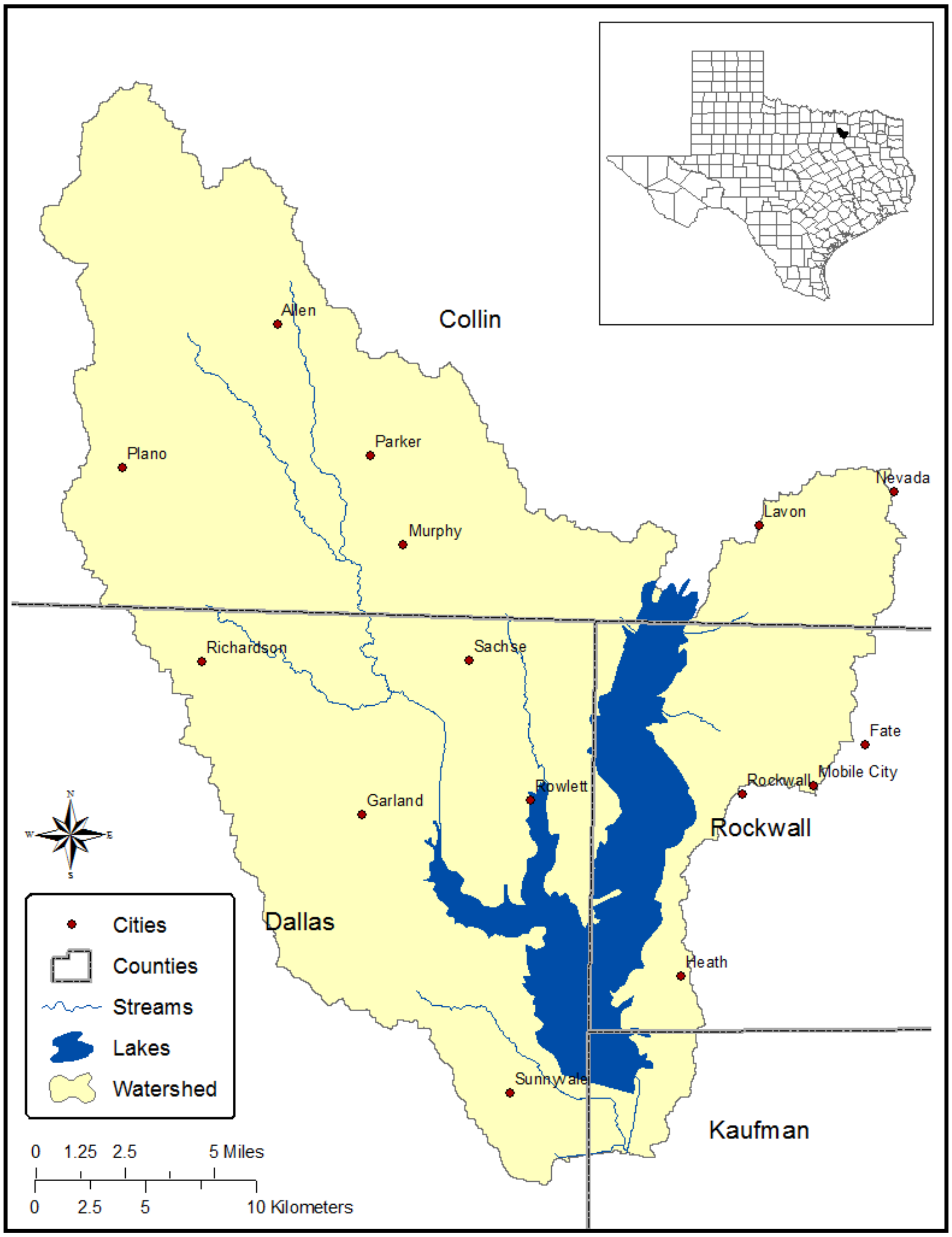


Figure 7.1 Location of Ray Hubbard Basin.

Model Input Data Tables and Figures

Topography

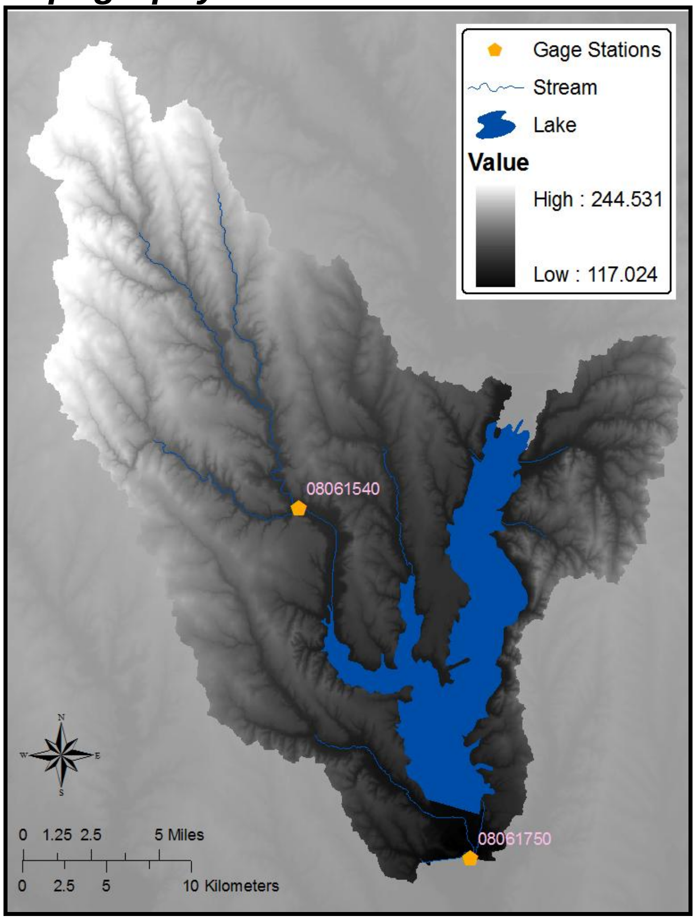


Figure 7.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of Rockwall - Forney Basin.

Soils

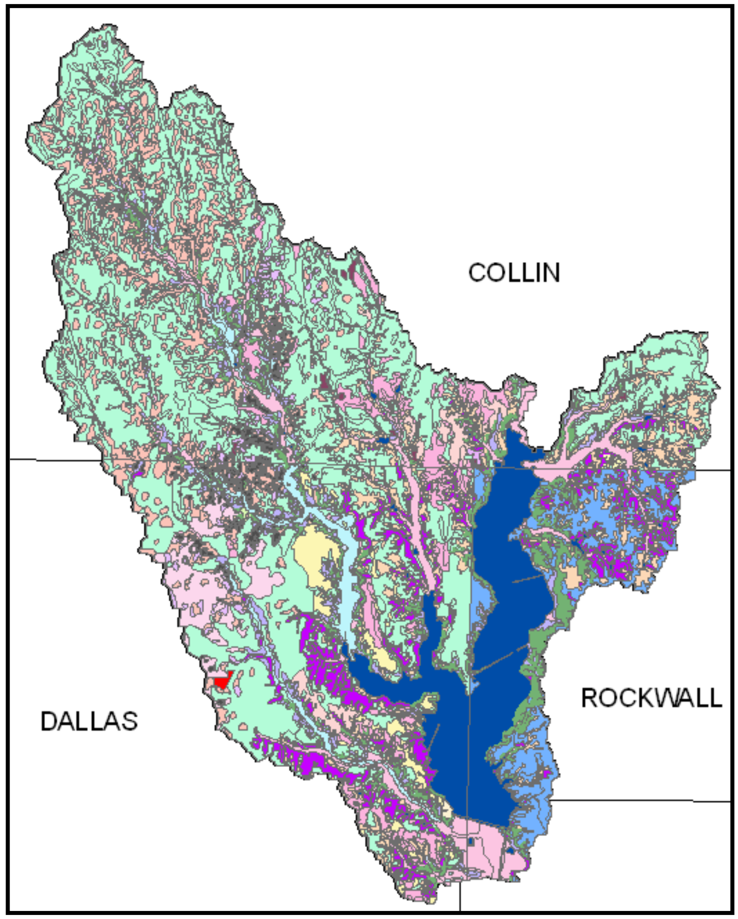


Figure 7.3 We used Soil Survey Geographic (SSURGO) data to define soil attributes in the SWAT model.

Weather

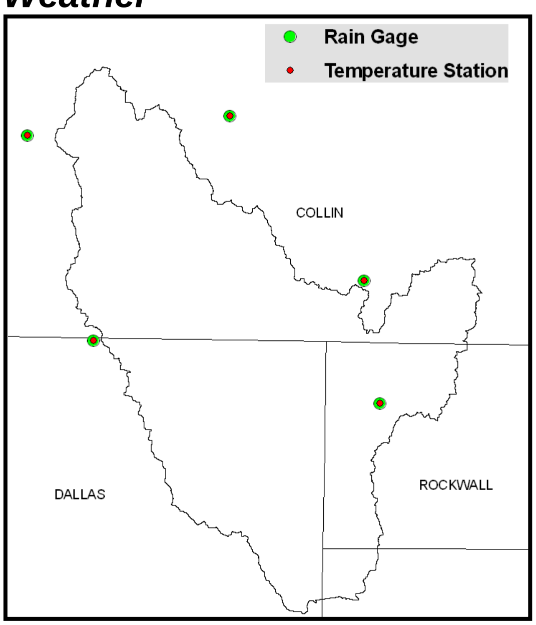


Figure 7.4 Five precipitation and temperature stations located in and around Ray Hubbard Basin provided daily rainfall and temperature values (maximum and minimum) for the SWAT simulation model.

Land use

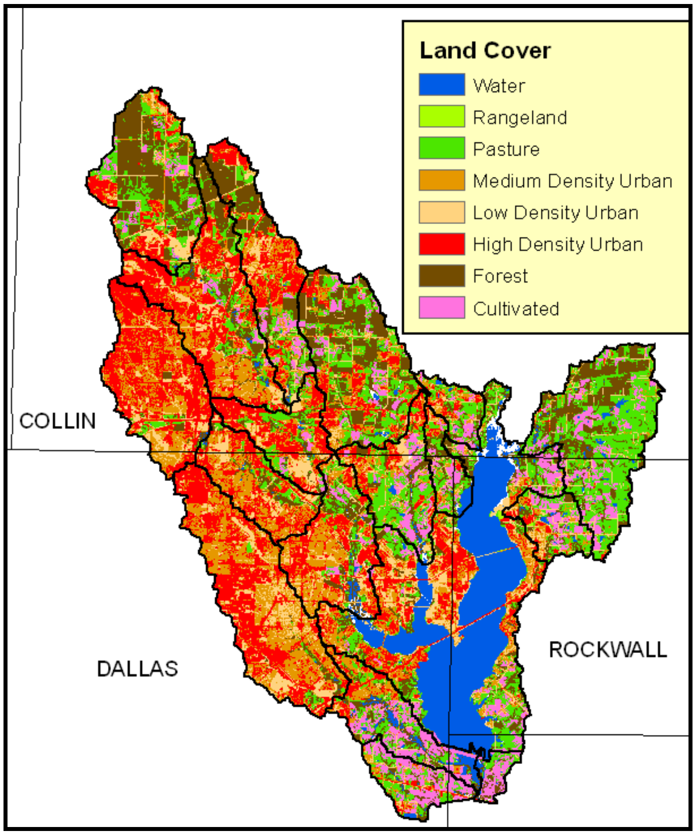
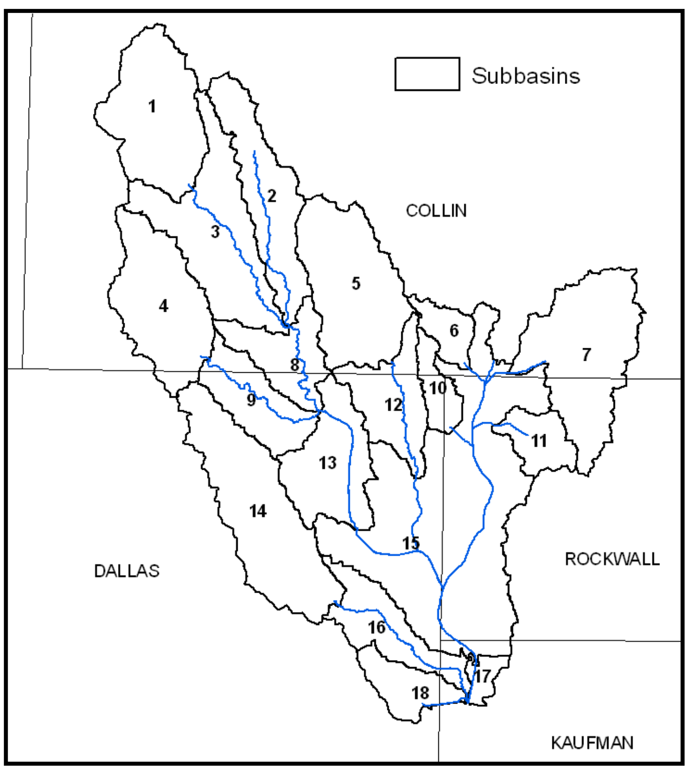


Figure 7.5 National Land Cover Data (2001) defined land use inputs for the Ray Hubbard Basin.

Subbasin Delineation



Point Sources

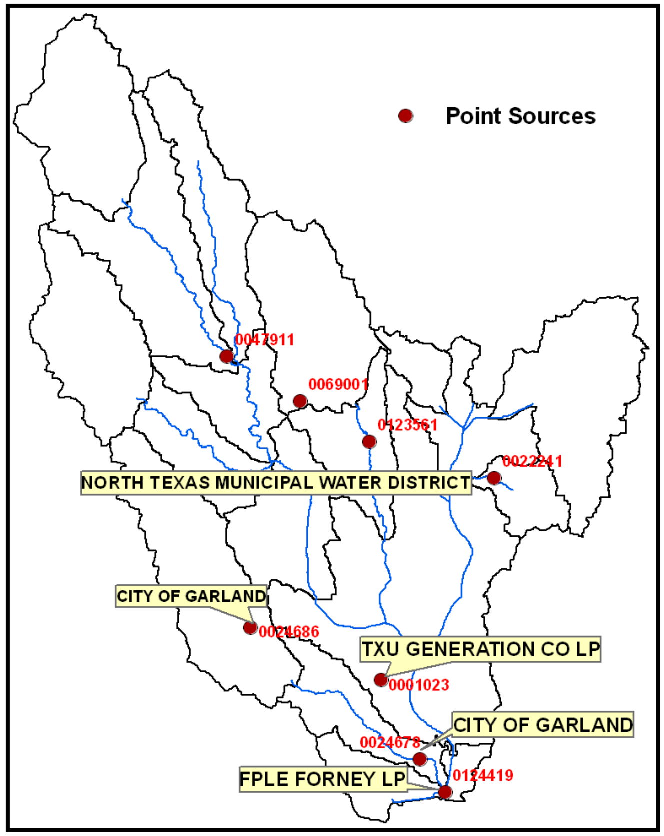


Figure 7.6 The USGS Water Resource database provided SWAT input data on the 11 actively operating point sources within the Ray Hubbard Basin. All 11 had discharge data, but three had discharges less than 0.005 m^3/sec. These were excluded from the model as insignificant. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

Figure 7.7 The Ray Hubbard SWAT model used a stream threshold value of 3,089 acres to delineate subbasins, resulting in 18 subbasins.

HRU Distribution

Each of the 18 subbasins were further split into HRUs based on researcher-defined thresholds, resulting in 548 HRUs.

Reservoir

Table 7.1 Ray Hubbard Basin contains one large reservoir, Lake Ray Hubbard, which was constructed prior to the simulation period. SWAT utilized available daily reservoir outflow data and reservoir characteristics from the National Inventory of Dams.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Ray Hubbard	15	23,998	49.0	26,813	79.1	Measured

Ponds

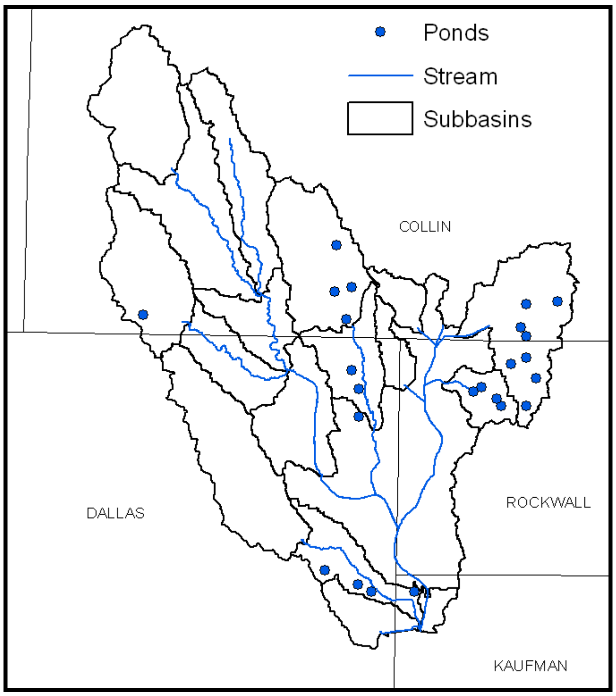


Figure 7.8 The U.S. Army Corps of Engineers' National Inventory of Dams provided SWAT with the location of 23 ponds in the Ray HubbardBasin. According to SWAT, approximately 15% of the basin area drains into ponds.

Model Calibration and Validation

Model simulation began in 1960. Using the Baseflow Filter Program (Arnold and Allen, 1999; Arnold et al., 1995), we analyzed streamflow records at the USGS gauging station at Rowlett Creek near Sachse (08061540) (figure 7.9) to determine the appropriate baseflow proportion. During calibration, we carefully matched the proportions of surface flow and

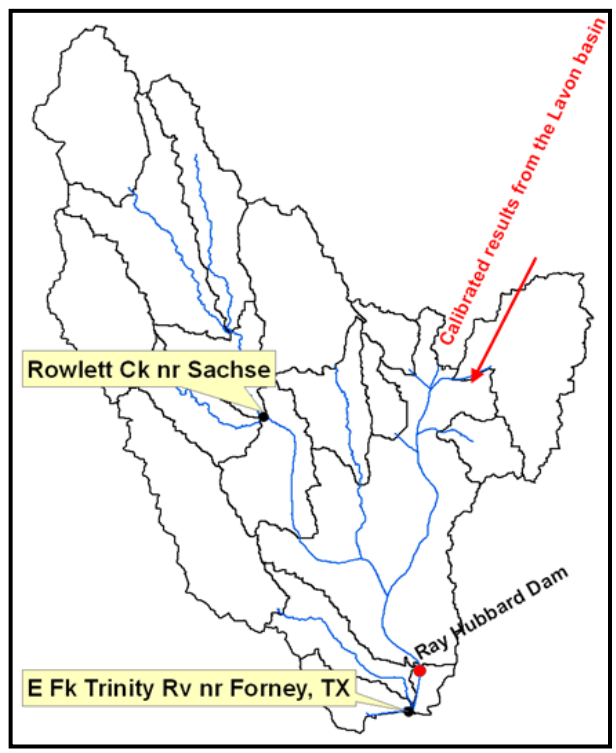


Figure 7.9 The locations of the USGS gauges used in the calibration and validation of the Ray Hubbard SWAT model.

baseflow. We calibrated SWAT for annual and monthly streamflow using flow records from the Sachse gauging station for the period lasting from 1968 through 1987. The model was validated for flow at the same gauge (1988–2006) and at the East Fork Trinity River near Forney gauging station (08061750, outlet of subbasin No. 17) (1973–2006). Calibrated parameters are listed in table 7.2.

Due to limited data, SWAT water quality components were calibrated using data from an upstream USGS gauging station at McKinney (see Chapter 4: Lavon Basin). To validate the model for water quality, we used available data from the East Fork Trinity River near Forney gauging station.

Table 7.2 Model parameters and ranges used for calibration and final calibrated values.

Component	Parameter (file)	Description	Range	Calibrated value
Flow	CN2 (.mgt)	Initial NRCS runoff curve number for moisture condition II	-4–4	Default + 2.0
	ESCO (.bsn)	Soil evaporation compensation factor	0.01–1.0	0.95
	EPCO (.bsn)	Plant uptake compensation factor	0.01–1.0	1.0
	GW_REVAP (.gw)	Groundwater revap coefficient	0.02–0.4	0.02
	REVAPMN (.gw)	Threshold depth of water in the shallow aquifer for "revap" to occur [mm]	1.0–15.0	1.0
Sediment	SPCON (.bsn)	Linear parameter for estimating maximum amount of sediment that can be reentrained during channel sediment routing	0.0001–0.01	0.0015
	CH_COV (.rte)	Channel cover factor	0.0–1.0	0.6
	CH_EROD (.rte)	Channel erodibility factor	0.0–1.0	0.07
Nutrients	RHOQ (.wwq)	Algal respiration rate at 20ºC (day^{-1})	0.05–0.50	0.1
Mineral nitrogen	SDNCO (.bsn)	Denitrification threshold water content (fraction of field capacity water content above which denitrification takes place)		1.5
	NPERCO (.bsn)	Nitrate percolation coefficient		1.0
Nitrogen in reach	AI1 (.wwq)	Fraction of algal biomass that is nitrogen	0.07–0.09	0.07
	RS4 (.swq)	Rate coefficient for organic N settling in the reach at 20ºC (day^{-1})		0.001
	BC3 (.swq)	Rate constant for hydrolysis of organic N to NH_4 in the reach at 20ºC (day^{-1})		0.001
Mineral phosphorus	PPERCO (.bsn)	Phosphorus percolation coefficient	10.0–17.5	10
	PHOSKD (.bsn)	Phosphorus soil partitioning coefficient	100–175	105
Phosphorus in reach	AI2 (.wwq)	Fraction of algal biomass that is phosphorus	0.01–0.02	0.01
	BC4 (.swq)	Rate constant for mineralization of organic P to dissolved P in the reach at 20ºC (day^{-1})		0.01
	RS5 (.swq)	Organic P settling rate in the reach at 20ºC (day^{-1})		0.001

Results and Discussion

Model Calibration and Validation

Flow

Calibration of annual streamflow (1968–1987) at the Sachse station (figure 7.10) resulted in R^2 and NSE values of 0.67 and 0.64, respectively. A monthly comparison resulted in R^2 and NSE values of 0.74 and 0.73, respectively (table 7.3). Validation of annual and monthly streamflow for the Sachse station (1988–2006) and for the East Fork Trinity River near Forney station (1973–2006) resulted in R^2 and NSE values ranging from 0.60 to 0.98 (table 7.3). Simulated flow for both the calibration and validation periods produced trends similar to corresponding observed data (figures 7.10–7.13).

Table 7.3 Annual and monthly streamflow (m^3/s) calibration and validation results by USGS gauging station.

Gauge station ID (site name)		Period	Measured			Simulated			Yearly		Monthly	
			Mean	Std_y	Std_m	Mean	Std_y	Std_m	R^2	NSE	R^2	NSE
08061540 (Rowlett Creek near Sachse)	Calibration	1968-1987	2.82	1.25	3.73	3.06	1.11	3.19	0.67	0.64	0.74	0.73
	Validation	1988-2006	4.80	1.40	4.15	4.23	1.39	4.10	0.77	0.60	0.79	0.76
08061750 (East Fork Trinity River near Forney)	Validation	1973-2006	18.98	13.76	30.97	17.86	13.17	29.23	0.98	0.97	0.98	0.97

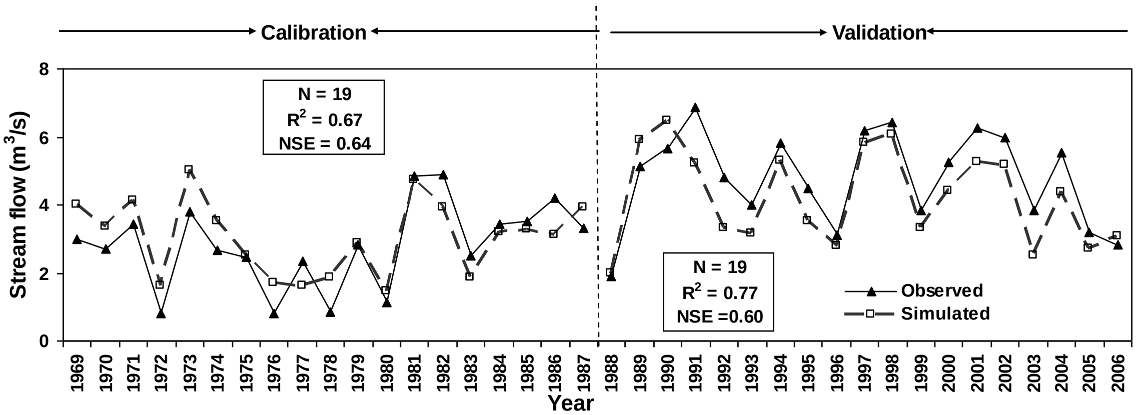


Figure 7.10 Measured and simulated yearly streamflow at Rowlett Creek near Sachse for the calibration (1968–1987) and validation periods (1988–2006).

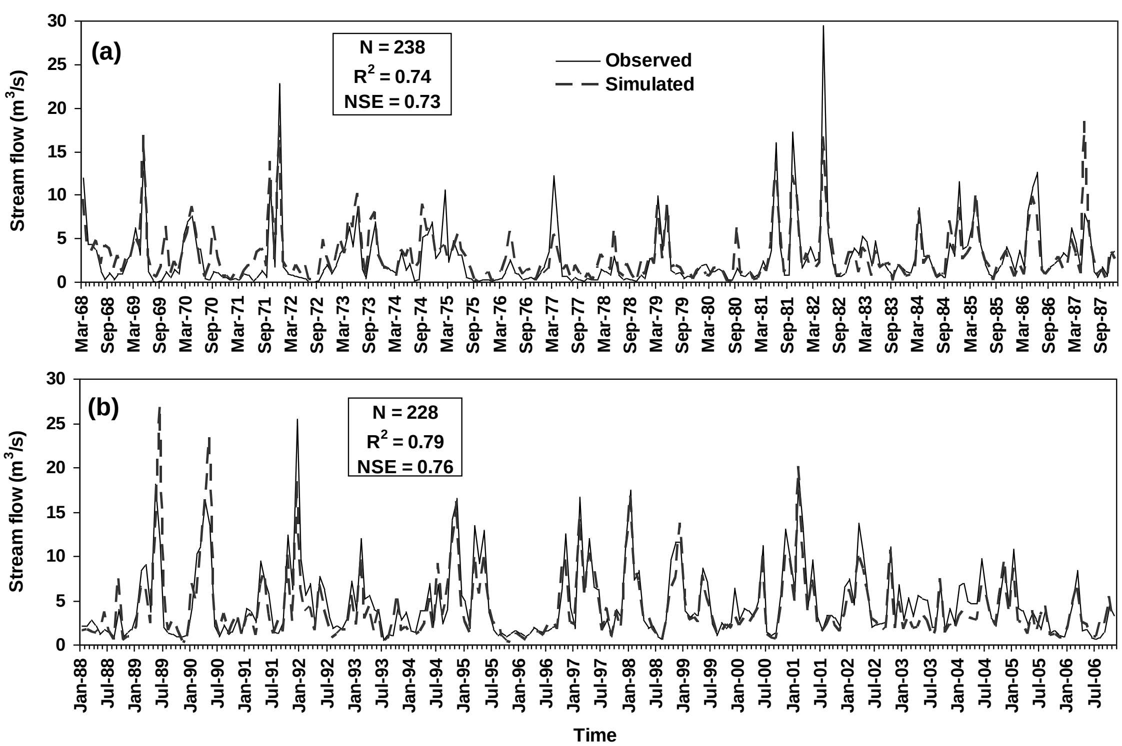


Figure 7.11 Measured and simulated monthly streamflow at Rowlett Creek near Sachse for the (a) calibration (3/1968–12/1987) and (b) validation period (1/1988–12/2006).

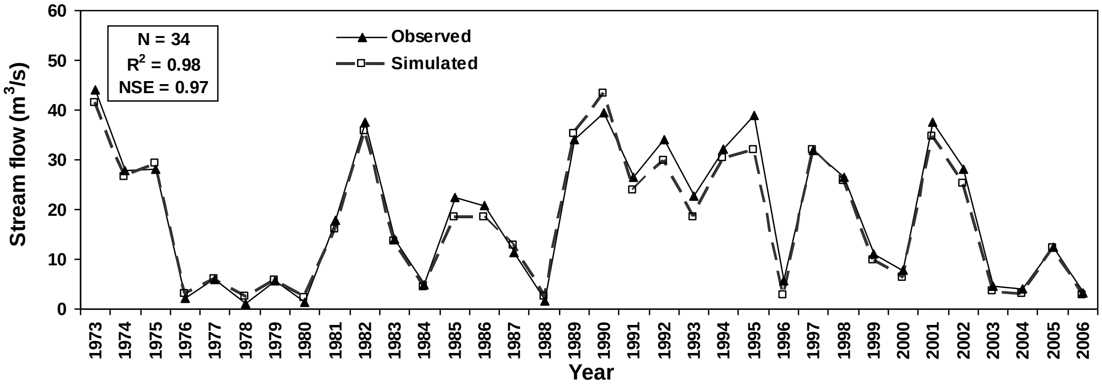


Figure 7.12 Validation of yearly streamflow at East Fork Trinity River near Forney. Note: This site is downstream of the Ray Hubbard Dam. Therefore, reservoir release, which is a SWAT input, dominates flow data for this gauge.

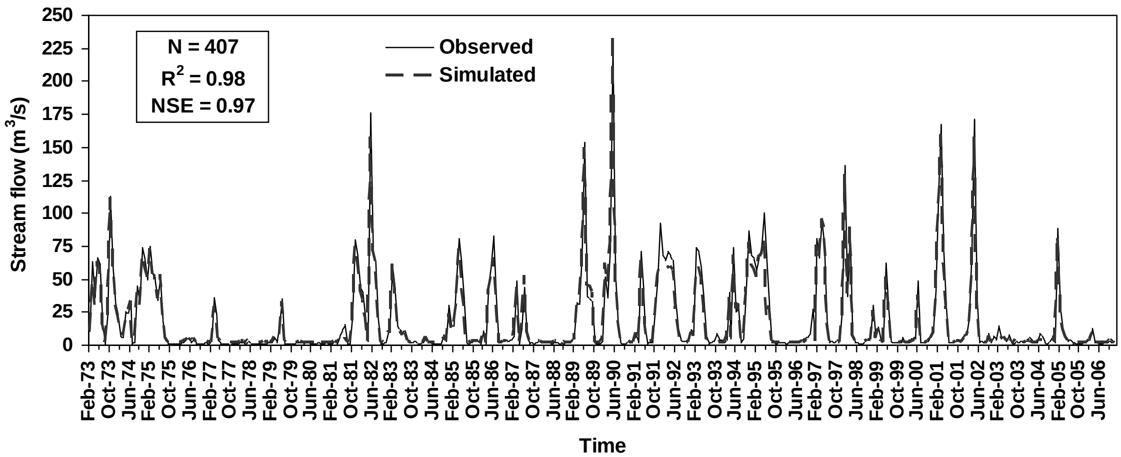


Figure 7.13 Validation of simulated monthly streamflow at East Fork Trinity River near Forney for 1973–2006. Note: This site is downstream of the Ray Hubbard Dam. Therefore, reservoir release, which is a SWAT input, dominates flow data for this gauge.

Water Quality

We used available water quality data to validate the Ray Hubbard SWAT model from November 1981 to November 1992. However, no sediment data were available for this basin. For days with available data, we compared simulated and measured values at the East Fork Trinity River near Forney (table 7.4 and figure 7.14). Organic nitrogen was clearly under-predicted, which led to the under-prediction of total nitrogen.

Table 7.4 Water quality validation for days with available data at the USGS gauging station at East Fork Trinity River near Forney.

	Total P		NO3-NO2		Organic N		Total N	
	Observed	Simulated	Observed	Simulated	Observed	Simulated	Observed	Simulated
Mean in kg/day (lb/day)	817.2 (1801.9)	763.2 (1682.9)	1402.5 (3091.9)	1684.4 (3713.4)	1931.9 (4259.1)	1302.8 (2872.2)	4010.8 (8842.3)	3355.8 (7398.2)
% Error		-6.6		20.1		-32.6		-16.3

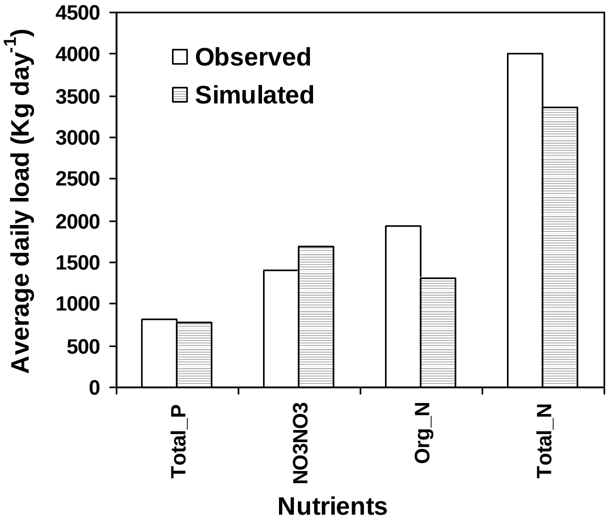


Figure 7.14 Validation of measured and simulated average daily nutrient loads at USGS gauge 08061750 (East Fork Trinity River near Forney).

Model Predictions

During the simulation period lasting from 1968–2007, SWAT predicted that overland generated 66,484 metric tons (73,286 tons) of sediment per year under current management scenarios. The model predicted that the total sediment actually reaching Lake Ray Hubbard was about 214,700 metric tons (236,666 tons) per year, with channel banks contributing about 69% of that sediment load. Sediment is generated non-uniformly across the basin, but in general, overland range areas with steeper slopes had higher sediment yields (figure 7.15).

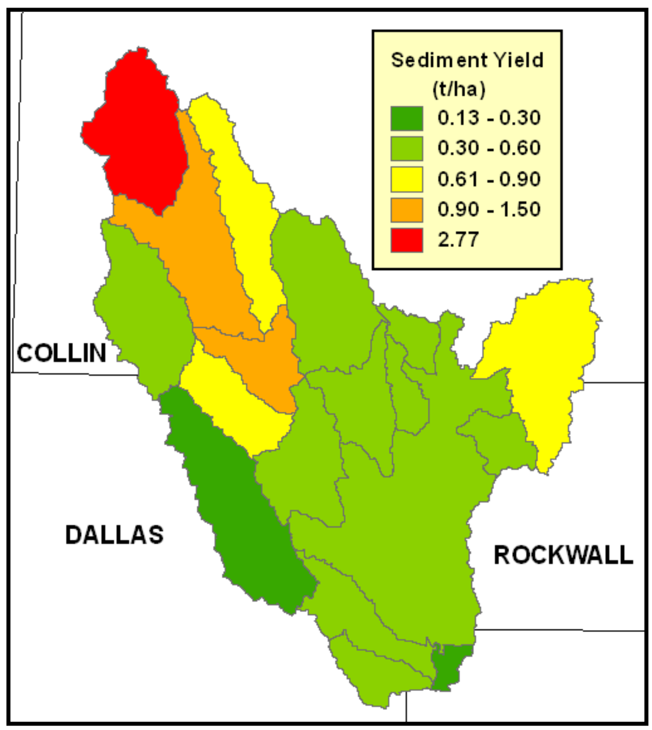


Figure 7.15 Sediment losses from Ray Hubbard Basin as predicted by the SWAT model.

During the simulation period (1968–2007), the model also predicted that phosphorus and nitrogen losses from overland were 186,734 kilograms (411,678 pounds) per year and 849,158 kilograms (1,872,073 pounds) per year, respectively, under current management scenarios. Total phosphorus loss from each subbasin is mapped in figure 7.16. Basin point sources generated 24,600 kilograms (54,235 pounds) of phosphorus and 41,248 kilograms (90,939 pounds) of nitrogen. The SWAT model predicted that 162,480 kilograms (358,207 pounds) of total phosphorus and 1,088,650 kilograms (2,400,062 pounds) of total nitrogen actually reach Lake Ray Hubbard annually.

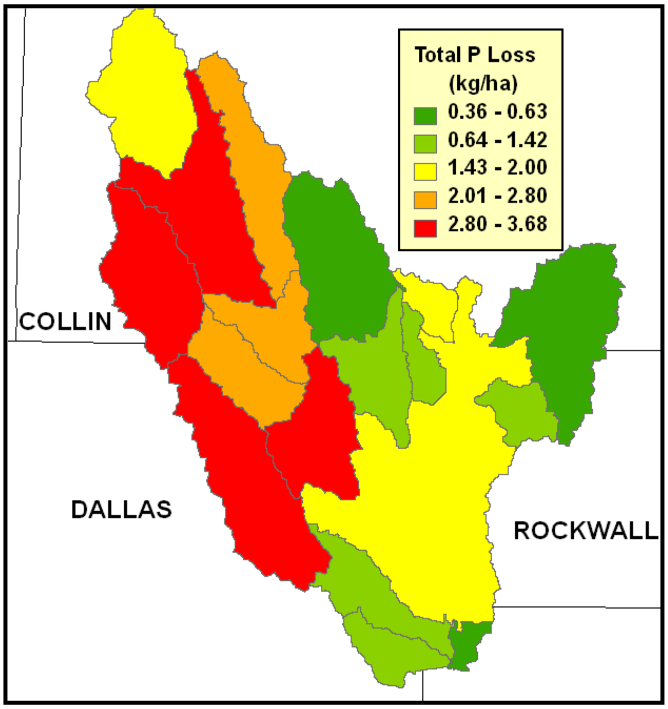


Figure 7.16 Phosphorus losses from Ray Hubbard Basin as predicted by the SWAT model.

Scenarios

Conservation Practices

SWAT simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 7.5.

Ponds

By removing the basin's 23 ponds, determined based on the National Inventory of Dams' PL-566 structures (see Chapter 2, General Model Input Data), SWAT estimated that the cumulative effect of those ponds reduced sediment loading of Lake Ray Hubbard by 4%.

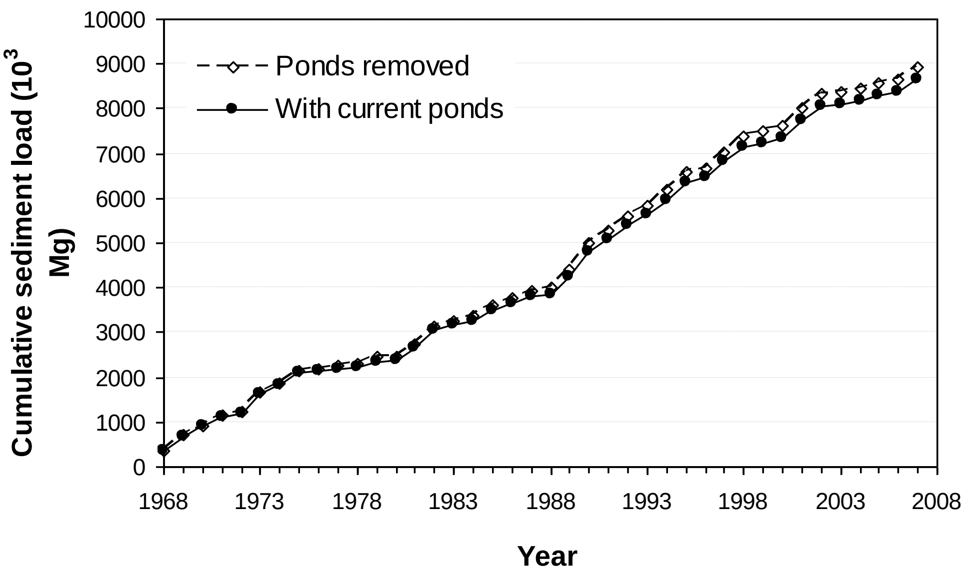


Figure 7.17 SWAT-predicted cumulative sediment loads received by Lake Ray Hubbard with and without currently existing ponds.

Range Utilization

Removing grazing from rangelands resulted in a 6% reduction in sediment loads reaching Lake Ray Hubbard.

Point Source Load Elimination

The Ray Hubbard Basin contains eight point sources. According to SWAT, the elimination of these WWTPs reduced reservoir loading of total phosphorus by 8% and nitrogen by 20%.

Urbanization

According to the North Central Texas Council of Governments' population projections, urban area within the Ray Hubbard Basin is expected to increase from 113,916 acres in 2000 to 119,599 acres in 2030, changing the percentage of urban area from 51% to 53%.

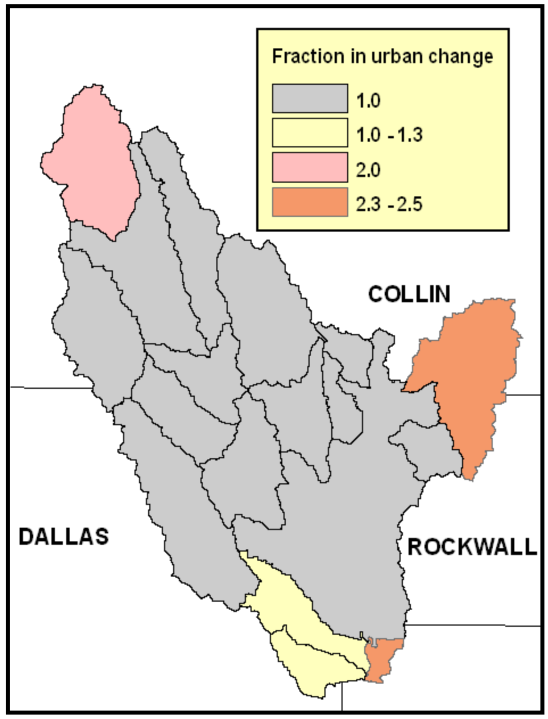


Figure 7.18 Projected changes in urban area from 2000 to 2030 (urban area in 2030 = urban area in 2000 * fraction of urban change).

Table 7.5 Changes in sediment and nutrient loads received by Lake Ray Hubbard under different scenarios as derived from SWAT model simulations (1968–2007).

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	214,700 metric tons/yr (236,666 tons/yr)	162,480 kg/yr (358,207 lb/yr)	1,088,650 kg/yr (2,400,062 lb/yr)
No Ponds	3.5%	7.4%	8.3%
No Range Grazing	-6.3%	-2.6%	-1.6%
Urban *	-10.3%	3.2%	-2.6%
No Point Sources	-0.6%	-7.8%	-20.4%

* load changes from overland only

Conclusions

We used SWAT to simulate the effects of urbanization and other land use changes on hydrologic and water quality processes in the Ray Hubbard Basin. SWAT calibration and validation resulted in a modeled representation of the watershed that was within acceptable standards. For example, NSE values based on monthly flow comparisons ranged from 0.76 to 0.97. Due to limited data, SWAT water quality components were calibrated using data from an upstream USGS gauging station at McKinney (see Chapter 4: Lavon Basin), and no recalibration was conducted for the Ray Hubbard Basin. We did, however, compare model predictions with the limited water quality data at the East Fork Trinity River near Forney gauging station. Percent error between simulated and observed average daily total phosphorus was -6.6%. However, percent errors for average daily loads of total nitrogen, organic nitrogen and soluble nitrogen were -16%, -33% and 20%, respectively.

During the simulation period lasting from 1968–2007, SWAT estimated that the total sediment load reaching Lake Ray Hubbard was about 214,700 metric tons (236,666 tons) per year with channel banks contributing about 69% of that sediment load. Total phosphorus and nitrogen losses from overland areas were approximately 186,734 kilograms (411,678 pounds) per year and 849,158 kilograms (1,872,166 pounds) per year, respectively, under current management scenarios. The model predicted that 162,480 kilograms (358,207 pounds) of total phosphorus and 1,088,650 kilograms (2,400,062 pounds) of total nitrogen reach Lake Ray Hubbard annually.

In the scenario analyses, the complete removal of ponds, point sources and rangeland management demonstrated their overall contribution to the water quality of the Lake Ray Hubbard. Based on model predicted values, the removal of ponds would increase the sediment load received by Lake Ray Hubbard by 4% while total phosphorus and total nitrogen would increase by about 7% and 8%, respectively. Removing point sources achieved reductions of approximately 8% in total phosphorus and 20% in total nitrogen received by Lake Ray Hubbard. Finally, removing grazing from rangelands reduced the sediment load by approximately 6%, total phosphorus by 3% and total nitrogen by 2%. SWAT also predicted the effects of urban expansion in 2030. In the urbanization projections, urban areas are expected to increase from 51% in 2000 to 53% in 2030. Associated with this expansion, the model predicts an increase of approximately 3% in total phosphorus and a reduction of 3% in total nitrogen produced in overland areas.

References

See the appendix.

Chapter 8: Benbrook Basin

Introduction

In this project, we used SWAT to address the effects of urbanization and other land use changes on sediment and nutrient delivery to Benbrook Reservoir. The U.S. Army Corps of Engineers constructed Benbrook Reservoir in 1947 as a flood control structure and water supply reservoir on the Clear Fork of the Trinity River, southeast of Fort Worth, Texas (figure 8.1). The reservoir covers 3,756 acres and drains 271,816 acres of Parker, Tarrant, Johnson and Hood counties. The basin is primarily rangeland on which cattle production is the most common agricultural activity. Upstream of the Benbrook Reservoir, near the city of Weatherford, lies Lake Weatherford. The catchment of this second large reservoir covers an area of 1,087 acres and is also used for flood control.

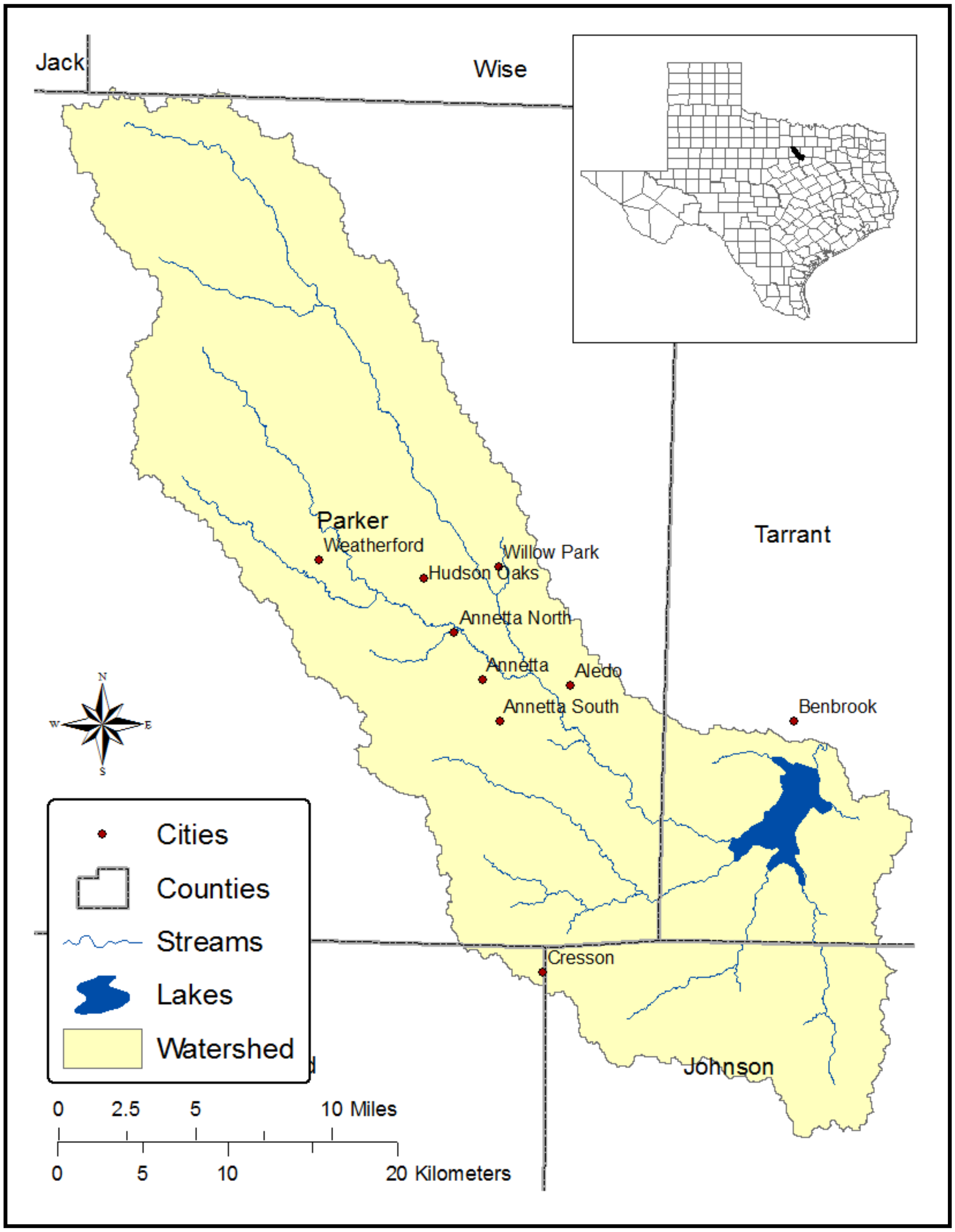


Figure 8.1 Location of Benbrook Basin.

Model Input Data Tables and Figures

Topography

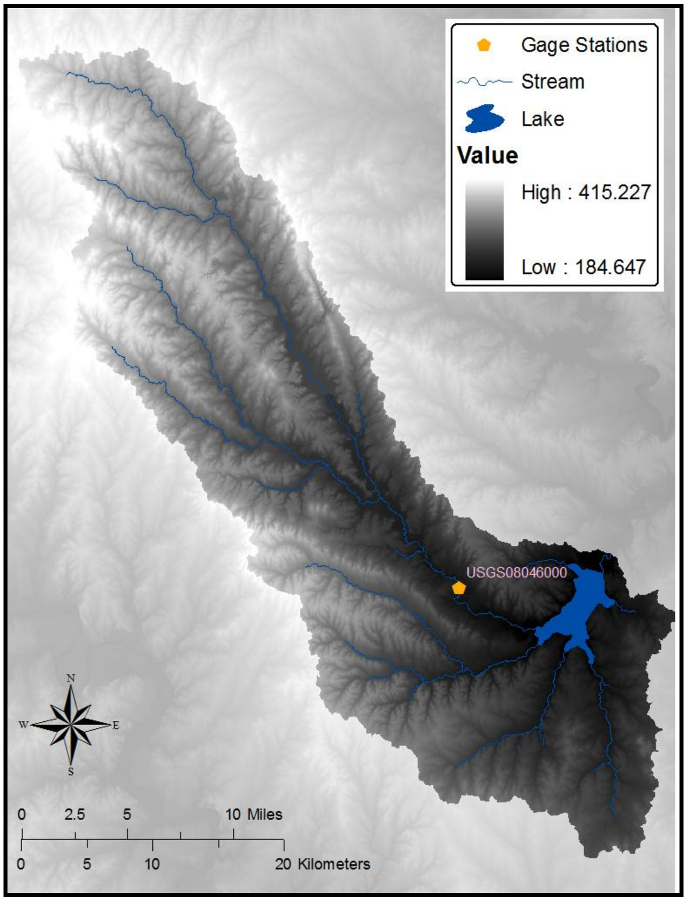


Figure 8.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of the Benbrook Basin.

Soils

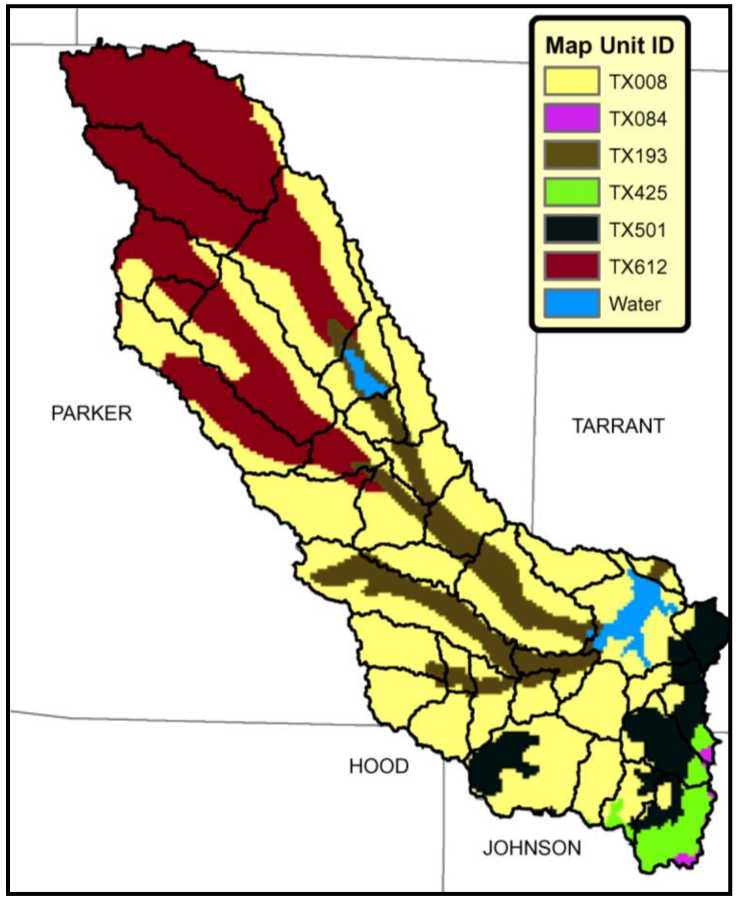


Figure 8.3 In other basins, SWAT utilized the SSURGO soils database. However, for the Benbrook Basin, we used State Soil Geographic (STATSGO) data bundled with the ArcSWAT interface instead.

Weather

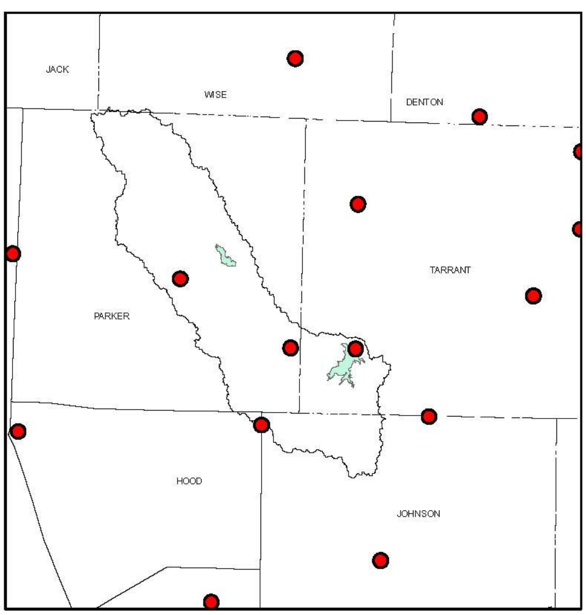


Figure 8.4 As described in the weather section of chapter 2, National Weather Service cooperative weather stations (shown here) provided minimum and maximum temperature and rainfall data from 1970 to 2005.

Land Use

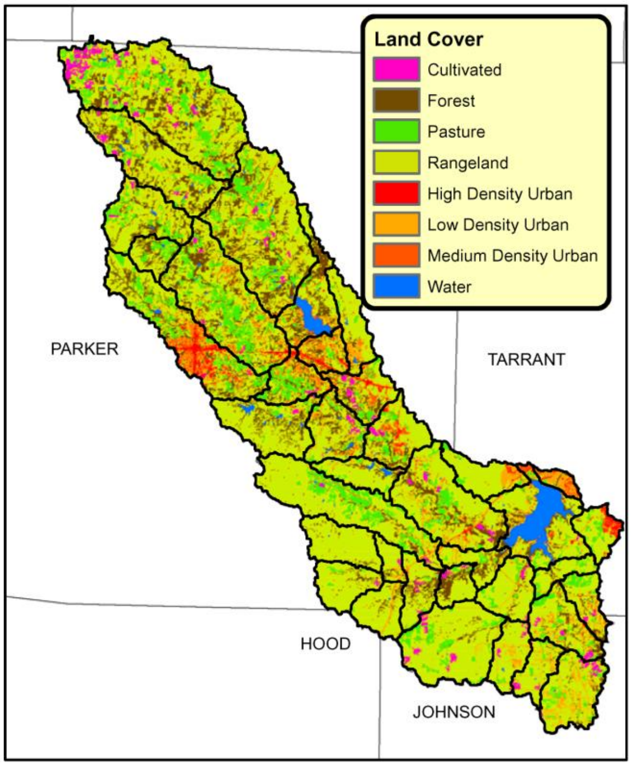


Figure 8.5 The 2001 National Land Cover Dataset defined Benbrook Basin land cover.

Point Sources

Six WWTPs were listed as active in the EPA's Permit Compliance System (figure 8.6). All six had measured discharge data but lacked useful nitrogen and phosphorus concentration data. We excluded point sources discharging less than 0.005 m^3 of effluent per second from the model as insignificant, leaving only two point sources (table A-1). Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

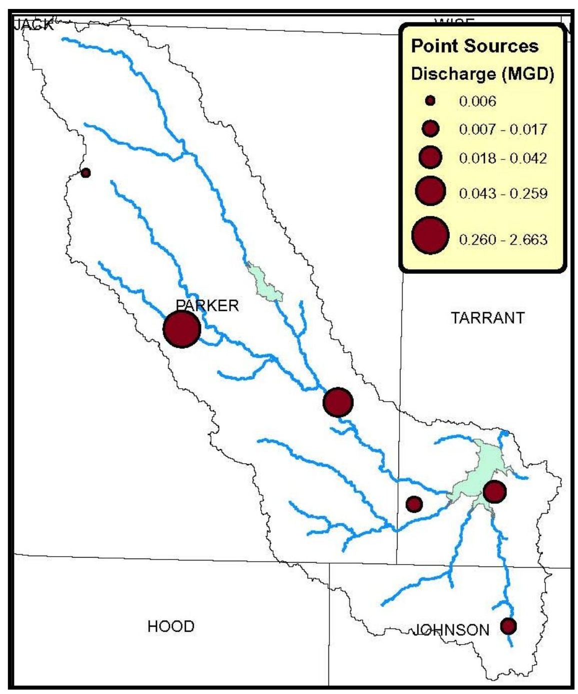


Figure 8.6 The location of point sources in the Benbrook Basin by discharge in Millions of Gallons per Day (MGD).

Major Impoundments

Benbrook Basin contains two large reservoirs, Lake Weatherford and Benbrook Reservoir (figure 8.7). Both were constructed prior to the simulation period. The National Inventory of Dams provided reservoir characteristics used to define each lake (table 8.1). SWAT utilized estimated daily discharge data at Benbrook. However, no discharge data was available for Lake Weatherford, so we simulated its discharge values. Lake Weatherford is the primary drinking water supply for the City of Weatherford, and approximately 3,648 acre-feet of water are removed from the reservoir annually for this purpose. We included this withdrawal amount in the SWAT model as average monthly consumption.

Table 8.1 Reservoir characteristics.

Reservoir	Subbasin	Surface Area at Principle Spillway (acre)	Volume at Principle Spillway (acre-feet)	Surface Area at Emergency Spillway (acre)	Volume at Emergency Spillway (acre-feet)	Release
Lake Weatherford	6	1,087	19,863	1,359	37,455	Simulated
Benbrook Lake	19	3,756	88,368	7,611	409,410	Measured

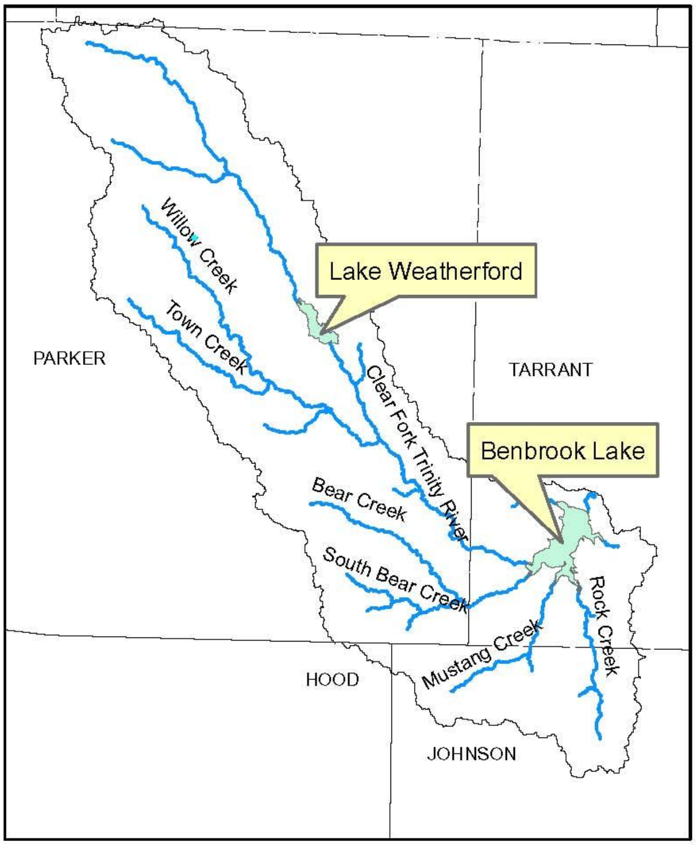


Figure 8.7 Major impoundments and streams in the Benbrook Basin.

Subbasin Delineation

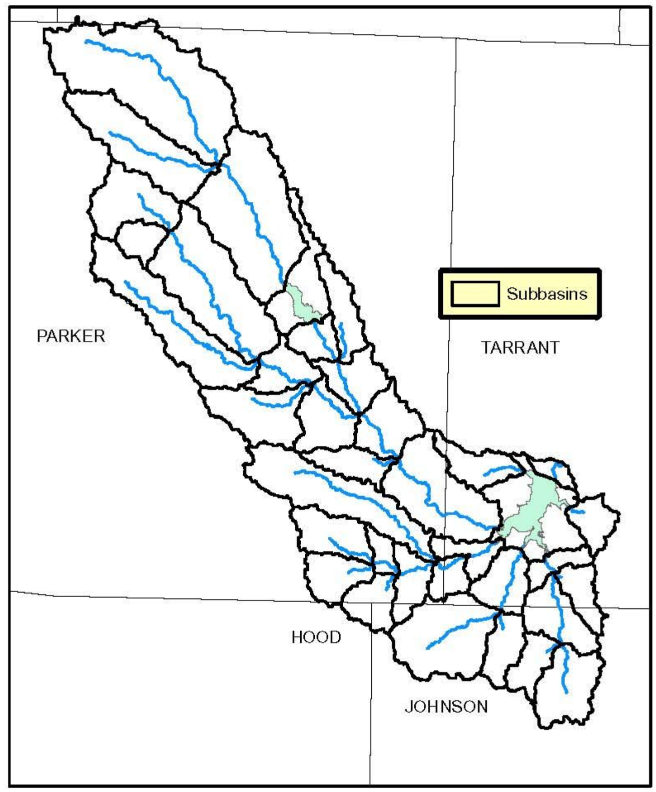


Figure 8.8 The Benbrook SWAT model used a stream threshold value of 2,471 acres to delineate subbasins, resulting in a total of 37 subbasins.

HRU Distribution

SWAT split each of the 37 subbasins into HRUs based on a user-defined threshold that determines the minimum percentage of any land cover required to designate an HRU within a subbasin. For the Benbrook Basin, we reduced both the land use [%] over subbasin area threshold and the soil class [%] over subbasin area threshold to 0%. By reducing these thresholds to 0%, SWAT represented all land cover and soil class combinations. We also used slope to break HRUs into two categories, those with less than or greater than 5% slope. Our threshold selection resulted in 1,119 HRUs.

Soil Chemistry

SWAT applied a default value of 5 milligrams of labile phosphorus per kilogram of soil to rangeland and forest. This value corresponded to a Mehlich III Soil Test Phosphorus (STP) of approximately 20 pounds per acre, a consistent value for soils with no fertilization history and a phosphorus deficiency. We also set pasture and cultivated lands to 10 milligrams of labile phosphorus per kilogram of soil (50 lb/acre STP). Finally, we set urban areas to a value of 40 milligrams of labile phosphorus per kilogram of soil (200 lb/acre), a value derived from urban lawn and garden soil tests in Delaware County, Oklahoma (Storm et al., 2005).

Ponds

Rather than use the National Inventory of Dams, the National Land Cover Dataset defined a total of 1,309 acres of ponds with 30-meter (98-foot) data (figure 8.9). We estimated the drainage area of each pond assuming a fixed surface to drainage area ratio of 25:1. Whitis (2002) recommends drainage to surface area ratios between 30:1 and 5:1 depending on land use and soil type within the drainage area. Under these assumptions, approximately 12% of the entire basin drains into ponds. As introduced in chapter 2, we assumed that the ponds were 6.6 feet in depth and initially 75% full of water. SWAT assigned all other pond parameters based on default settings. For example, ponds are assumed to have only a primary spillway.

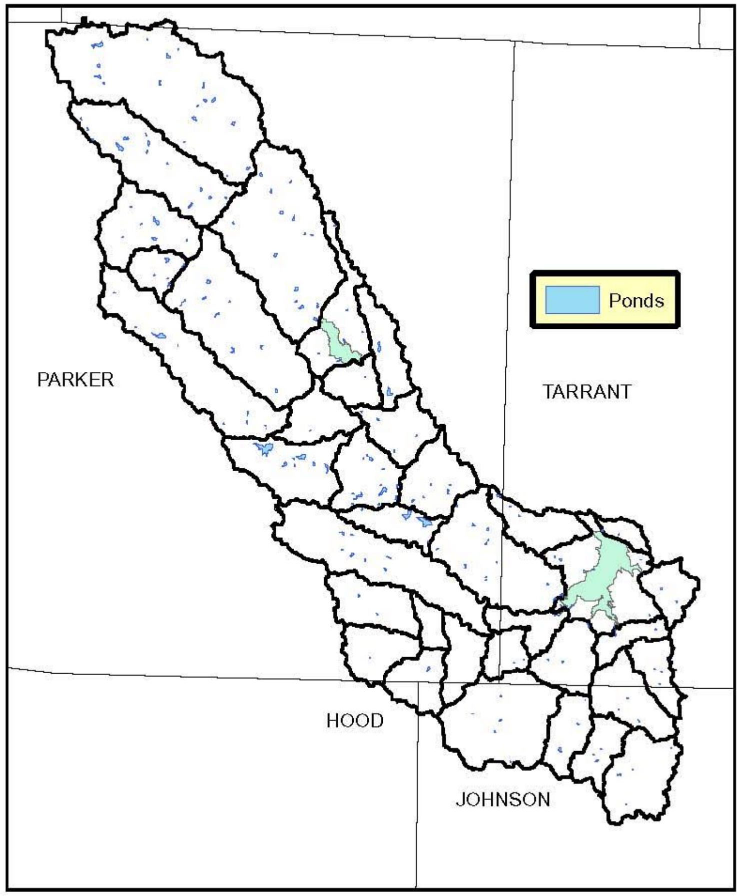


Figure 8.9 The 2001 National Land Cover Dataset defined ponds in the Benbrook Basin at a resolution of 30 meters (98 feet).

Model Calibration and Validation

Limited data were available in the Benbrook Basin with which to calibrate the SWAT model. Flow data were available at three USGS stream gauge sites (figure 8.10). However, all gauges were downstream of significant impoundments on the Clear Fork of the Trinity River. Unfortunately, data from these sites are not representative of the flow from overland areas because streamflow below reservoirs is more a function of reservoir management and releases than inflow from the drainage area. In addition, reservoir losses from evaporation, seepage and withdraw made the total water balance more difficult to define. The site at Aledo had a significant drainage area not subject to impoundments, but the period of record was too short (1970–1975) to serve as the primary calibration and validation site for streamflow. Therefore, this site was used for validation only.

Insufficient water quality data were available with which to estimate sediment and nutrient loads for calibrating the water quality components of SWAT. Several sites had some data, but those also containing USGS streamflow data were insufficient. Streamflow and water quality data are both required to estimate nutrient loads in order to calibrate the model. Sites downstream of major reservoirs such as Lake Weatherford could not be used. The SWAT model uses a relatively simple reservoir model, which may not reflect the true nutrient retention of these structures. For this reason, calibrating SWAT using measured data downstream of the structures is problematic. Due to the lack of suitable flow and water quality data for model calibration, we adopted SWAT-derived hydrologic and water quality parameters from the neighboring Bridgeport Basin SWAT model set-up, which did have measured data suitable for calibration in this basin. The parameter set for the Bridgeport model is given in table 8.2.

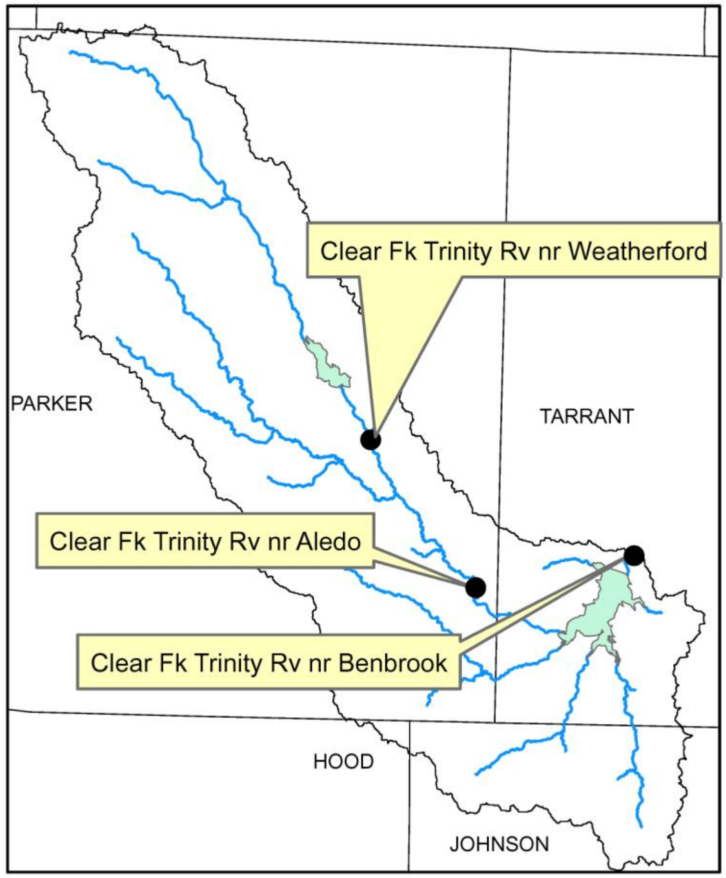


Figure 8.10 USGS gauges used for calibration and validation in the Benbrook Basin.

Observed and measured streamflow at both the Clear Fork of the Trinity River near Weatherford and near Benbrook are given in figures 8.11 and 8.12. Comparisons between observed and measured data at both stations yielded Nash-Sutcliff Efficiency (NSE) values that were acceptable for this application. Measured release data from Benbrook was incorporated into the SWAT model, creating an excellent model fit at the Clear Fork of the Trinity River near Benbrook gauge. Therefore, this site should not be used as an indicator of model quality. Monthly NSE values at the Clear Fork of the Trinity River near the Weatherford (figure 8.13) were better during the calibration period (0.50) than the validation period (0.17).

Table 8.2 Calibration parameters ported from the nearby Bridgeport model and used in the Benbrook model.

Component	Variable	Description	Values used in this study
Flow	CN2	Initial SCS runoff curve number for moisture condition II	Default-4
	ESCO	Soil evaporation compensation factor	0.5
	EPCO	Plant uptake compensation factor	0.5
	GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur	10
	REVAPMN	Threshold depth of water in the shallow aquifer required for "revap" to occur	10
	RCHRG_DP	Deep aquifer percolation fraction	0.25
	SOL_AWC1	Available water capacity of the soil layer	Default+0.04
	PHD_K	Hydraulic conductivity through bottom of ponds	0.5
	CH_K1	Effective hydraulic conductivity in tributary channel alluvium	0.5
Sediment	SPCON	Linear parameter for estimating the maximum amount of sediment that can be reentrained during channel sediment routing	0.0006
	CH_EROD	Channel erodibility factor	0.05
	CH_COV	Channel cover factor	0.95
Nutrients	IWQ	Stream nutrient transformations	0
Nitrogen	SDNCO	Denitrification threshold water content	0.85
Phosphorus	PHOSKD	Phosphorus soil partitioning coefficient	50
	PSP	Phosphorus sorption coefficient	8
	PPERCO	Phosphorus percolation coefficient	20
	IPET	PET method: 0=Priest-T,1=Pen-M, 2=Harg, 3=user input	1
	ISUBWQ	Instream water quality:1=model instream water quality	0

Table 8.3 Model calibration and validation statistics for streamflow (m^3/s)

Site	Period	Observed	Predicted	Relative Error, %
Trinity River near Weatherford	Calibration (1980–1995)	0.77	0.77	0
Trinity River near Weatherford	Validation (1996–2005)	0.51	0.86	-70
Trinity River near Aldeo	Validation (1970–1975)	1.62	1.66	-2
Trinity River near Benbrook	Validation (1970–2005)	3.36	2.93	13

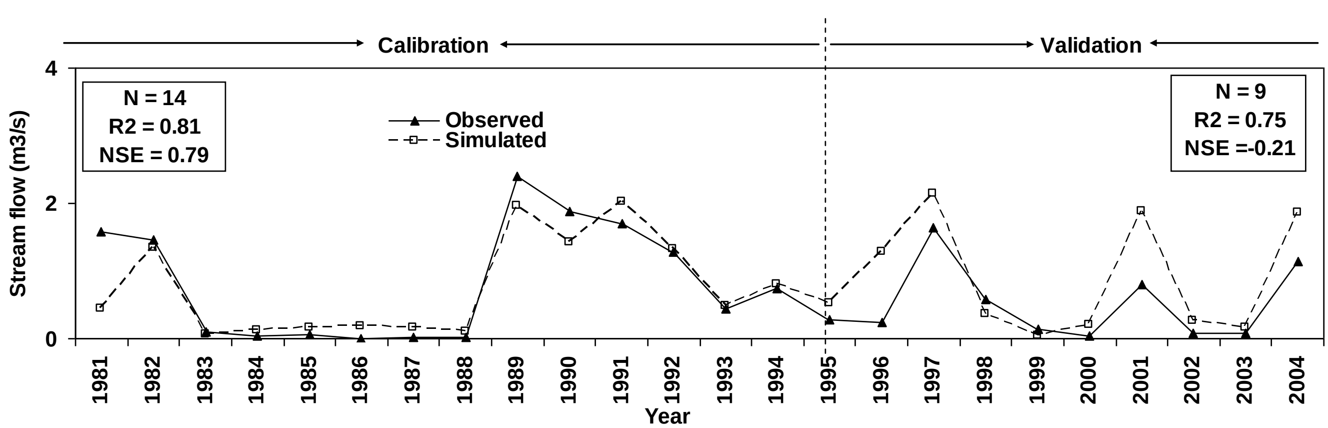


Figure 8.11 Observed and simulated annual streamflow at Clear Fork of the Trinity River near Weatherford.

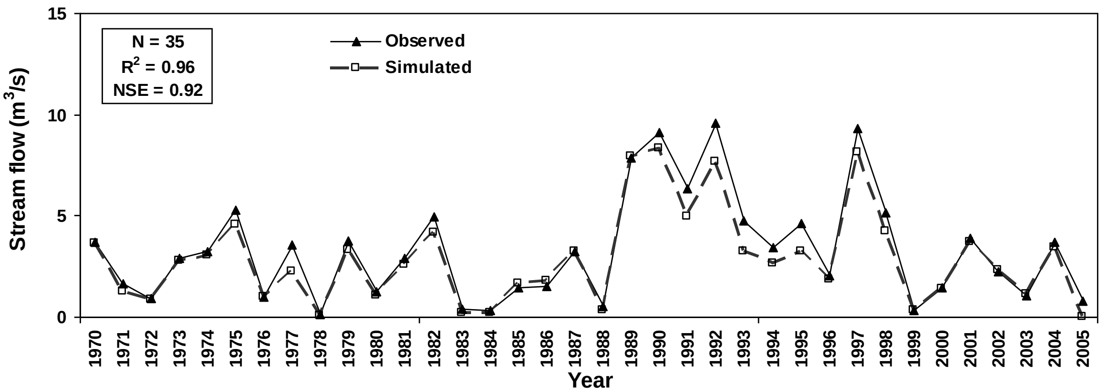


Figure 8.12 Observed and simulated annual streamflow at Clear Fork of the Trinity River near Benbrook. Note: This site is downstream from Benbrook Reservoir. In SWAT, reservoir release is based on measured data. Therefore, the high degree of correlation in this comparison is not an indicator of superior model performance.

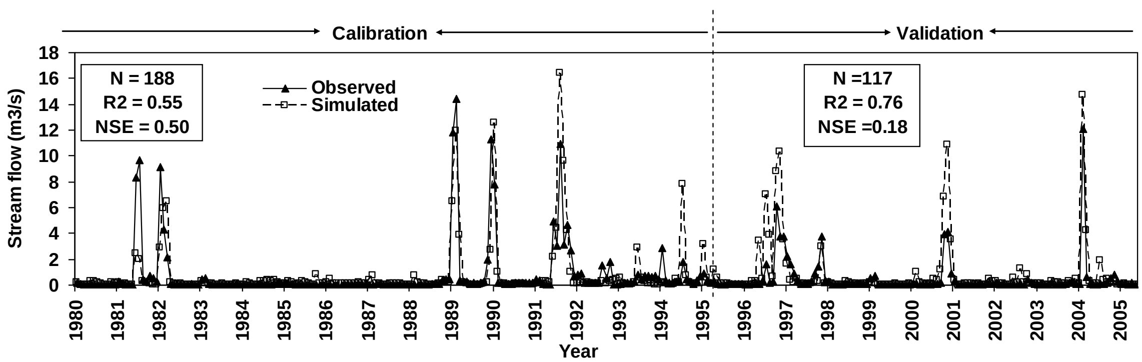


Figure 8.13 Observed and simulated monthly streamflow at Clear Fork of the Trinity River near Weatherford.

Model Predictions

The SWAT model can predict nutrient and sediment loading without calibration, but the results contain excessive uncertainty not suitable for many applications. There was no suitable data in the Benbrook Basin with which to develop sediment and nutrient loads for calibration of the SWAT model, so we used calibration parameters ported from the neighboring Bridgeport Basin. Only enough measured data were available to estimate loads and calibrate the Bridgeport model at a single location. The quantity and distribution of measured data available at this single site create a great deal of uncertainty in the model estimation of sediment and nutrient loads.

The model predicted that the Benbrook Basin generated 48,800 metric tons (53,793 tons) of sediment. A significant amount of sediment was deposited in streams and in Lake Weatherford. Therefore, according to SWAT, only 10,700 metric tons (11,795 tons) of sediment actually reached Benbrook Reservoir each year. The relative uncertainty in these predictions is very high due to the lack of data necessary to calibrate channel degradation and deposition of sediments. Sediment loads for Bridgeport were derived from total suspended solids data from grab samples. These data do not include bed load, thus the model was not calibrated to include it. While sediment is generated non-uniformly across the basin, overland range areas with erosive soils on steep slopes had higher sediment yields (figure 8.14). SWAT-predicted cumulative sediment yield from overland areas is given in figure 8.15.

The model predicted that 44,500 kilograms (98,106 pounds) of phosphorus and 145,000 kilograms (319,670 pounds) of nitrogen reach Benbrook Reservoir annually. Overland and point sources generated 50,000 kilograms (110,231 pounds) of phosphorus and 146,000 kilograms (321,875 pounds) of nitrogen. Nutrient loads were subject to far less attenuation than sediment loads in the routing portion of the model. Urban areas had the highest phosphorus yields, but both urban and overland range areas generated higher surface runoff than average. Total phosphorus loss is mapped in figure 8.16.

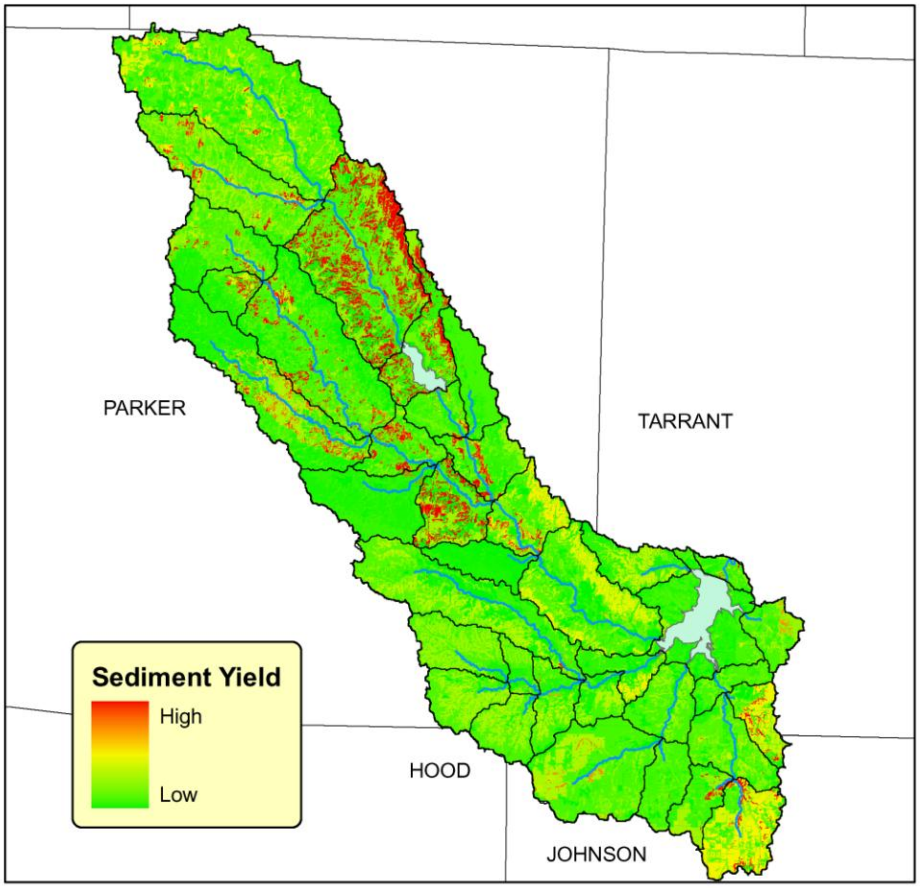


Figure 8.14 Sediment loss distribution as predicted by the Benbrook Basin SWAT model. Numeric values for sediment losses were intentionally excluded from this figure because the model was not validated enough to publish that level of detail.

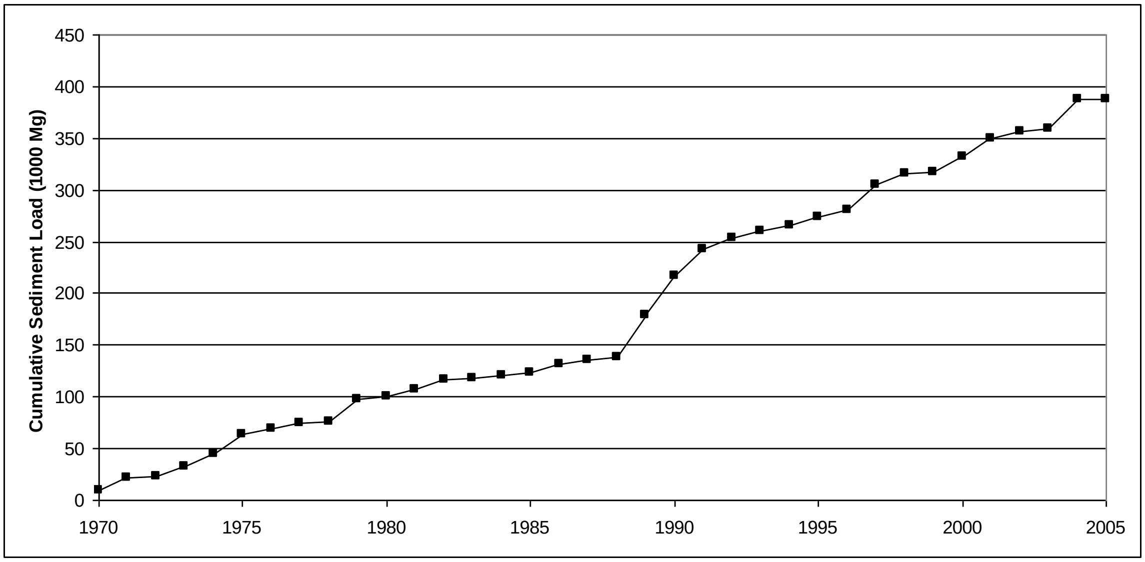


Figure 8.15 Cumulative sediment delivered to Benbrook Lake as predicted by SWAT (1970–2005).

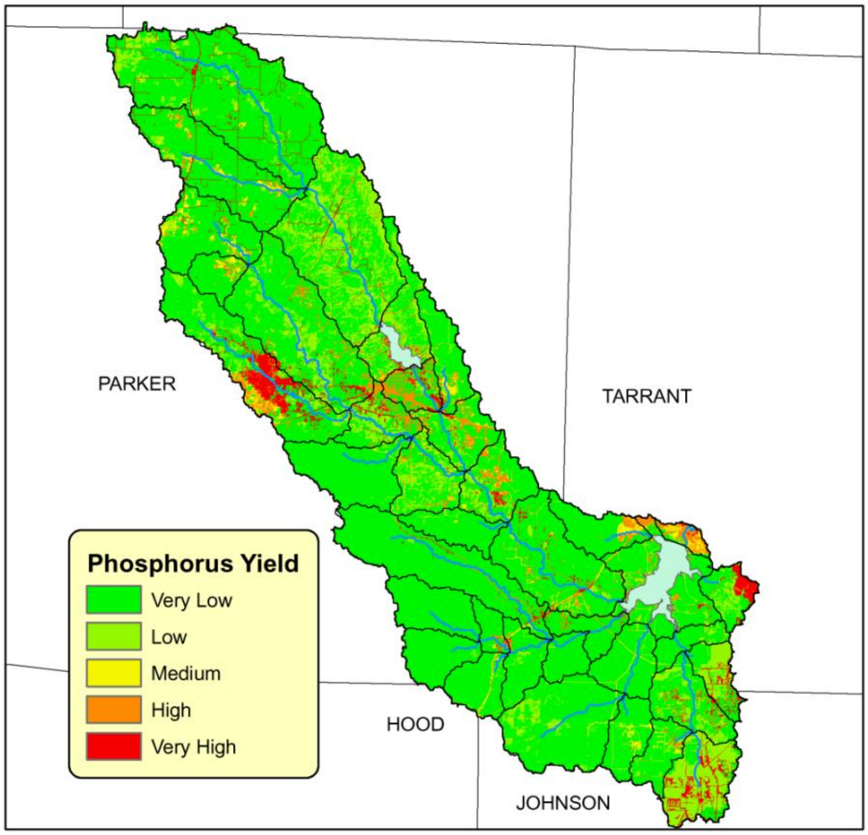


Figure 8.16 Phosphorus loss distribution as predicted by the SWAT model for the Benbrook Basin. Numeric values for phosphorus losses were intentionally excluded from this figure because the model was not validated enough to publish that level of detail.

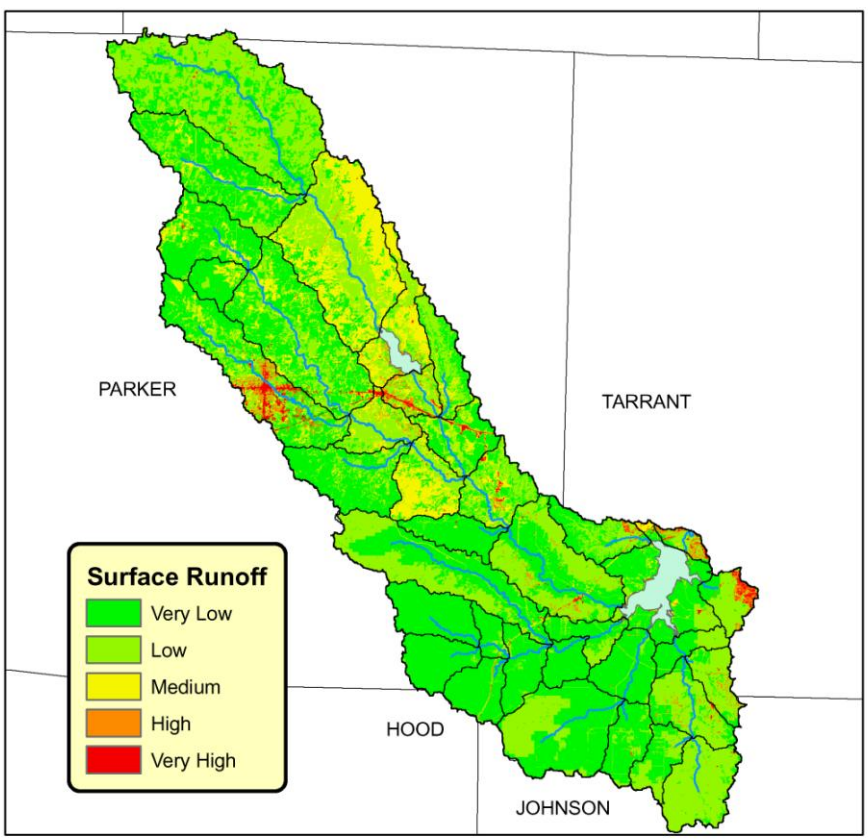


Figure 8.17 Surface runoff distribution as predicted by the SWAT model for the Benbrook Basin.

Scenarios

Conservation Practices

We simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 8.4.

Ponds

Removing ponds from the Benbrook model resulted in increased sediment and nutrient loads. According to SWAT, ponds in the Benbrook Basin cumulatively reduce sediments by 13%. Extrapolating their effects, the construction of 25% more ponds should reduce basin loads by 4.1%. A 25% increase in ponds was considered in this study due to its applicability to the Benbrook Basin; however, it was not considered for all basins.

Range Utilization

Removing grazing from rangelands resulted in a 16.7% reduction in sediment loads from overland areas.

Cropland Conversion

There is little cultivated cropland in the Benbrook Basin (1.9%). We assumed the cropland present was wheat, primarily for winter grazing. In the conservation scenario, we modified the model to convert all cultivated areas to hay. This resulted in a 4.3% reduction in sediment from overland areas.

Point Source Load Elimination

The basin contains only two point sources, accounting for 9.3% and 23% of the total phosphorus and nitrogen stream loads, respectively. The elimination of these WWTPs should reduce loads by a similar amount.

Urbanization

Urban area in Benbrook was projected to increase from 24,463 acres to 71,166 acres (figure 8.19). The model predicted the greatest change in nutrient loads, with nitrogen increasing 23% and phosphorus increasing 111%. The model indicated little change in sediment loads, with a decrease of 0.2% (figures 8.20-8.22).

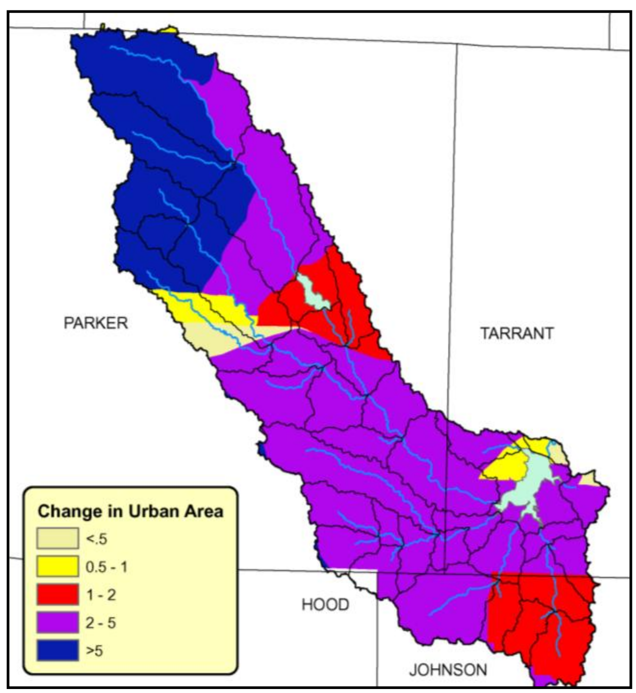


Figure 8.19 Projected change in urban area from 2000 to 2030.

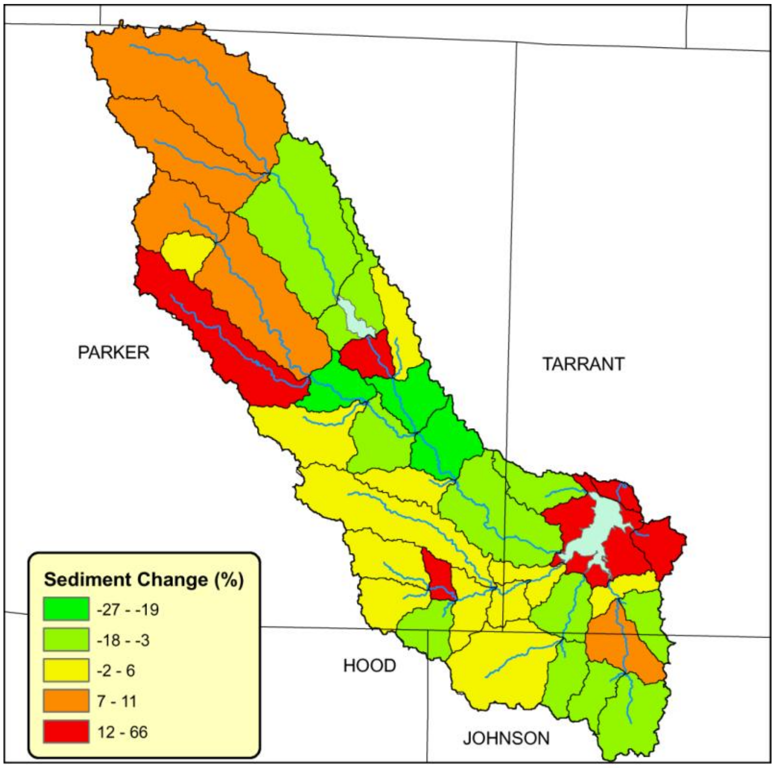


Figure 8.20 Changes in sediment loss from the Benbrook Basin (2000 to 2030) as predicted by SWAT.

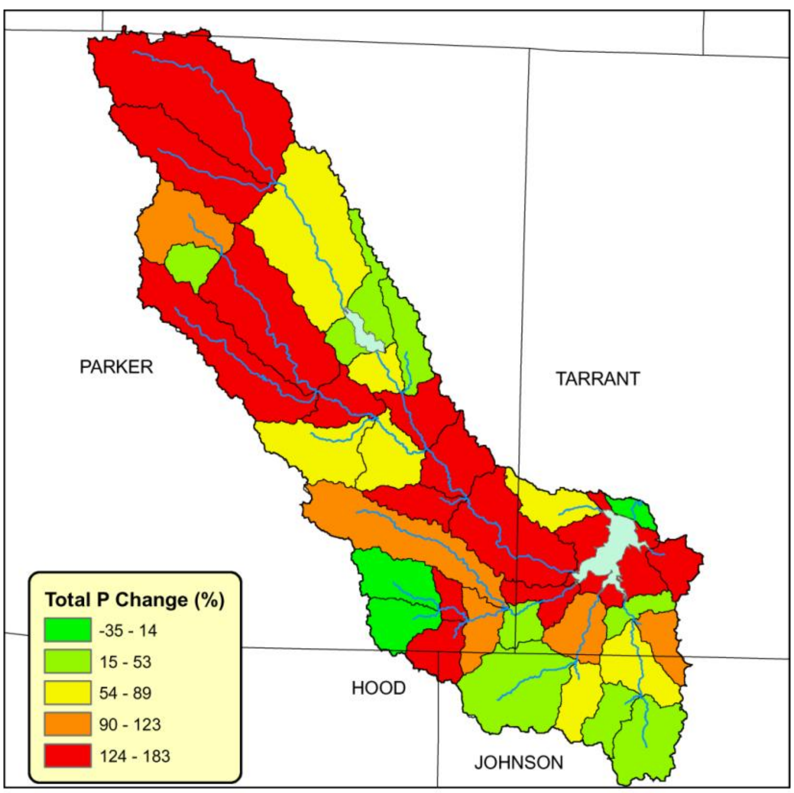


Figure 8.21 Changes in phosphorus losses from the Benbrook Basin (2000 to 2030) as predicted by SWAT.

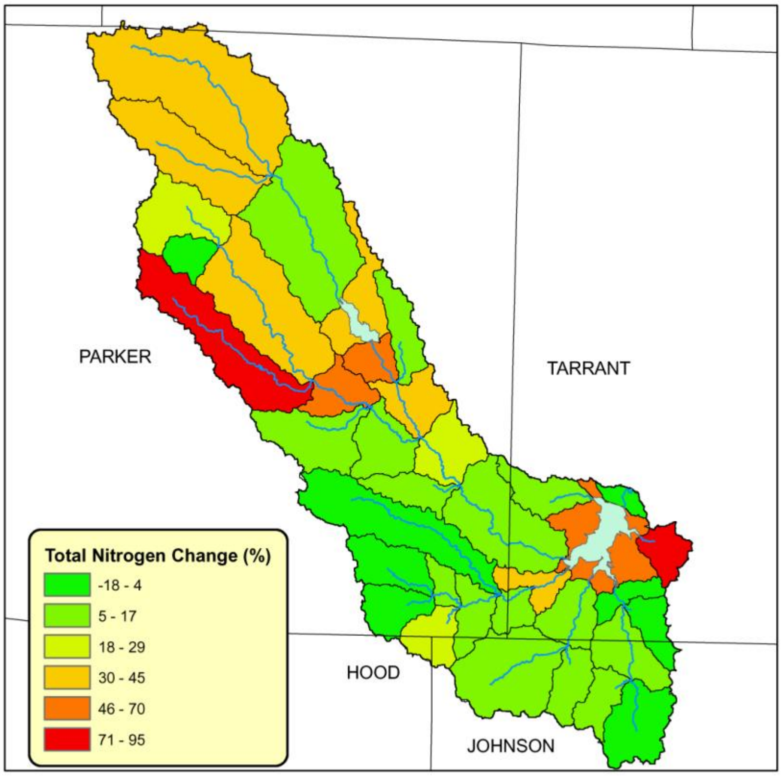


Figure 8.22 Changes in nitrogen losses from the Benbrook Basin (2000 to 2030) as predicted by SWAT.

Total Conservation Practice Impact

If all the proposed conservation practices were implemented, including the construction of 25% more ponds and the elimination of point-source contributions, nutrient loads could be significantly reduced (table 8.4). If all practices were implemented, sediment delivered to streams would be reduced by 24%. The cumulated reduction in overland sediment load is given in figure 8.23.

Table 8.4 SWAT-predicted sediment and nutrient load changes based on conservation scenarios.

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	10,700 metric tons/yr (11,795 tons/yr)	44,500 kg/yr (98,106 lb/yr)	145,000 kg/yr (319,670 lb/yr)
No Ponds	13.1%	5.8%	5.1%
No Range Grazing	-16.7%	-3.1%	-7.3%
Urban *	-0.2%	111%	23%
No Point Sources	-0.0%	-9.3%	-23.4%
Cropland to Hay Conversion	-4.3%	-2.0%	-6.4%
25% More Ponds	-4.1%	-2.0%	-1.7%

* load changes from overland only

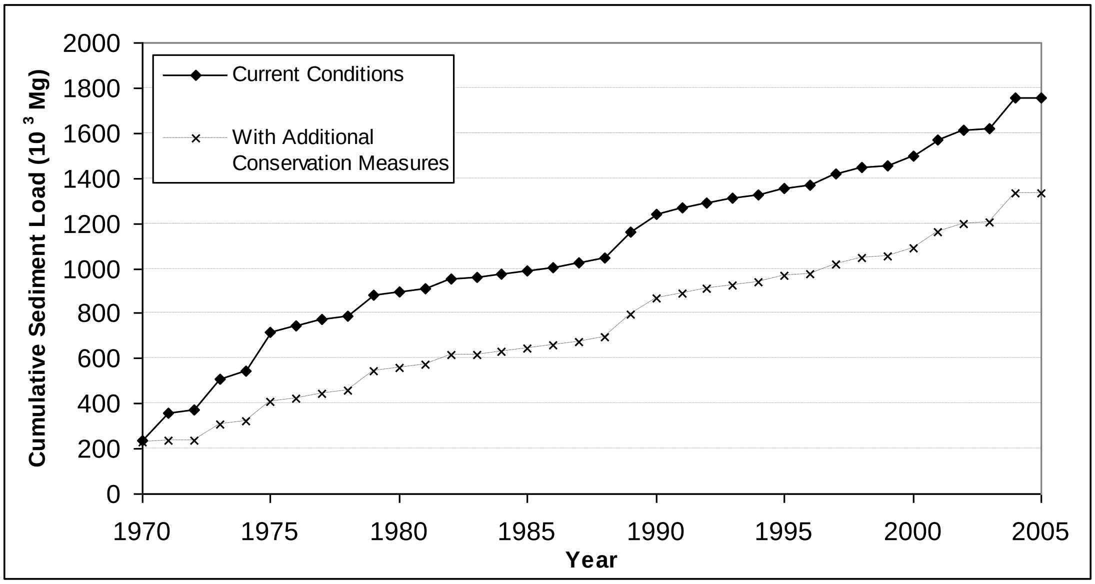


Figure 8.23 SWAT-predicted cumulative overland sediment losses with and without proposed conservation measures utilized in the Benbrook Basin (1970 to 2005). The load received by Benbrook reservoir is shown in figure 8.15

Conclusions

We evaluated the impacts of WWTPs, ponds, rangeland grazing, cropland conversion and urbanization on hydrologic and water quality variables using SWAT in the Bridgeport Basin. Due to the lack of suitable flow and water quality data for model calibration, we adopted SWAT-derived hydrologic and water quality parameters from the neighboring Bridgeport Basin SWAT model set-up. Flow records from the USGS gauging station on the Trinity River near Weatherford were used for calibration (1980–1995). For validation, we used streamflow records from the USGS gauging station on the Trinity River near Weatherford (1996–2005), Aldeo (1970–1975), and Benbrook (1970–2005).

Over the simulation period of 36 years (1970–2005), the model predicted that an average of 10,700 metric tons (11,794 tons) of sediment, 44,500 kilograms (98,106 pounds) of phosphorus and 145,000 kilograms (319,670 pounds) of nitrogen reach Benbrook Reservoir every year.

Scenario analyses included the removal of ponds, point sources, rangeland grazing and cropland conversion. The model also examined the effects of urban expansion predicted for 2030. Existing ponds, which are essentially PL-566 reservoirs, helped by settling about 13.1% of sediment, 5.8% of total phosphorus and 5.1% of total nitrogen generated and transported in the basin. Eliminating rangeland grazing and maintaining range grass reduced sediment, total phosphorus and total nitrogen by 16.7%, 3.1% and 7.3%, respectively. Point sources added 9.3% of total phosphorus and 23% of total nitrogen. The model also predicted that urban expansion in the basin would substantially increase total phosphorus loads (by 111%) and increase total nitrogen loads by 24%. One potential reason for the significant increase in phosphorus loss is the lack of cultivated agriculture in the basin. Rangeland grazing is the primary agricultural activity. With relatively light stocking rates and low soil phosphorus levels, these areas contribute relatively little phosphorus.

References

See the appendix.

Chapter 9: Bridgeport Basin

Introduction

In this project, we used SWAT to address the effects of urbanization and other land use changes on the delivery of sediment and nutrients to the Bridgeport Reservoir. Constructed on the West Fork of the Trinity River, Bridgeport Reservoir serves as a flood control structure and water supply reservoir (figure 9.1). The reservoir drains 704,003 acres and covers 12,000 acres in parts of Archer, Young, Clay, Jack and Wise counties. The basin is primary rangeland, with cattle production as the most common agricultural activity.

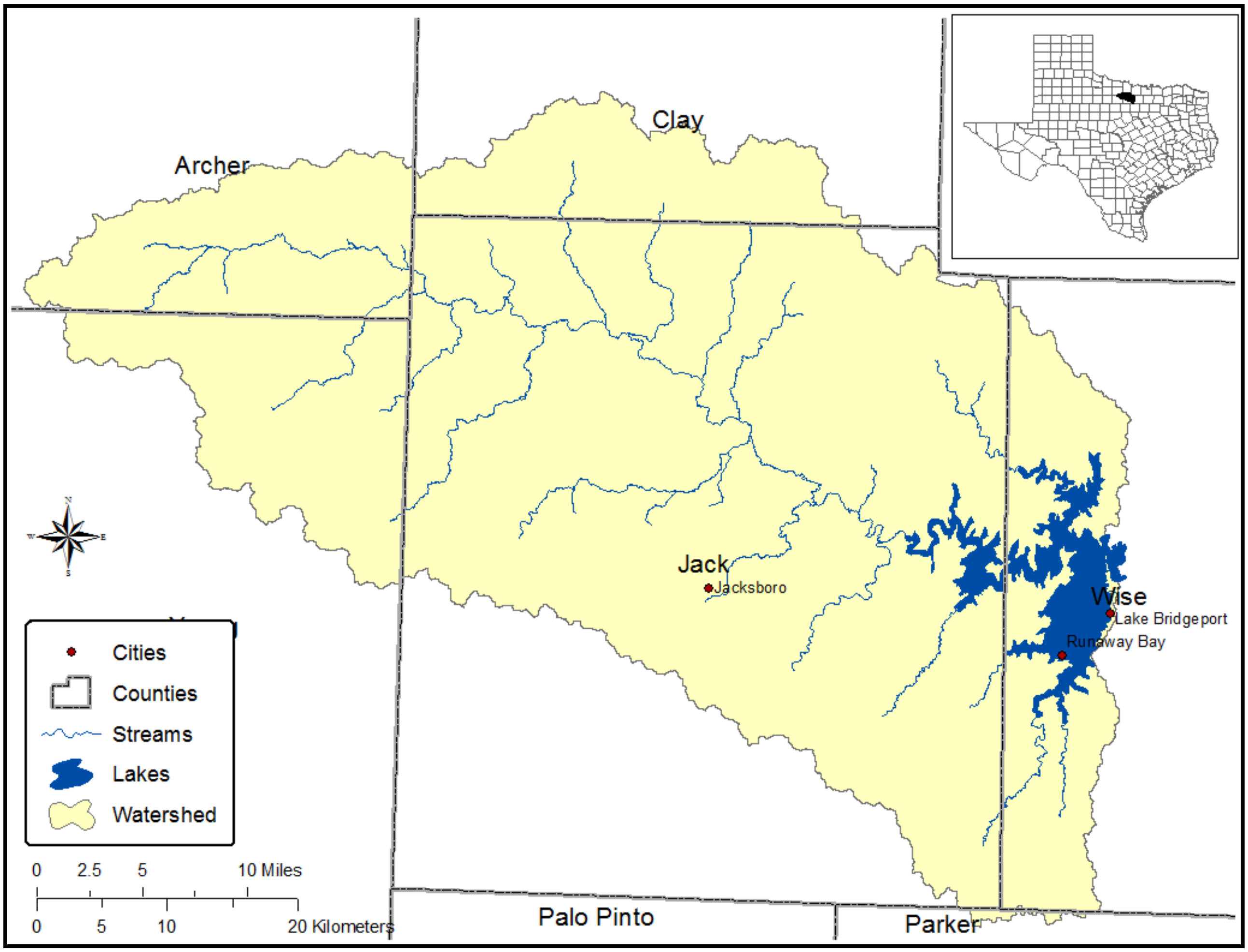


Figure 9.1 Location of Bridgeport Basin.

Topography

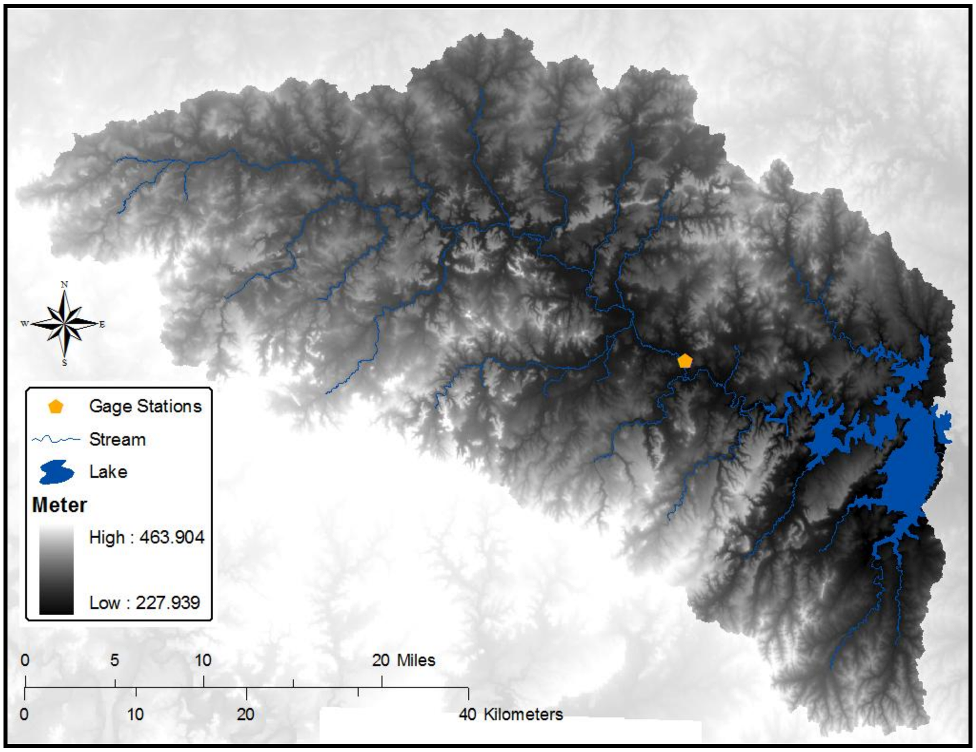


Figure 9.2 A 30-meter (98-foot) Digital Elevation Model defined the slope of the Bridgeport Basin SWAT model.

Soils

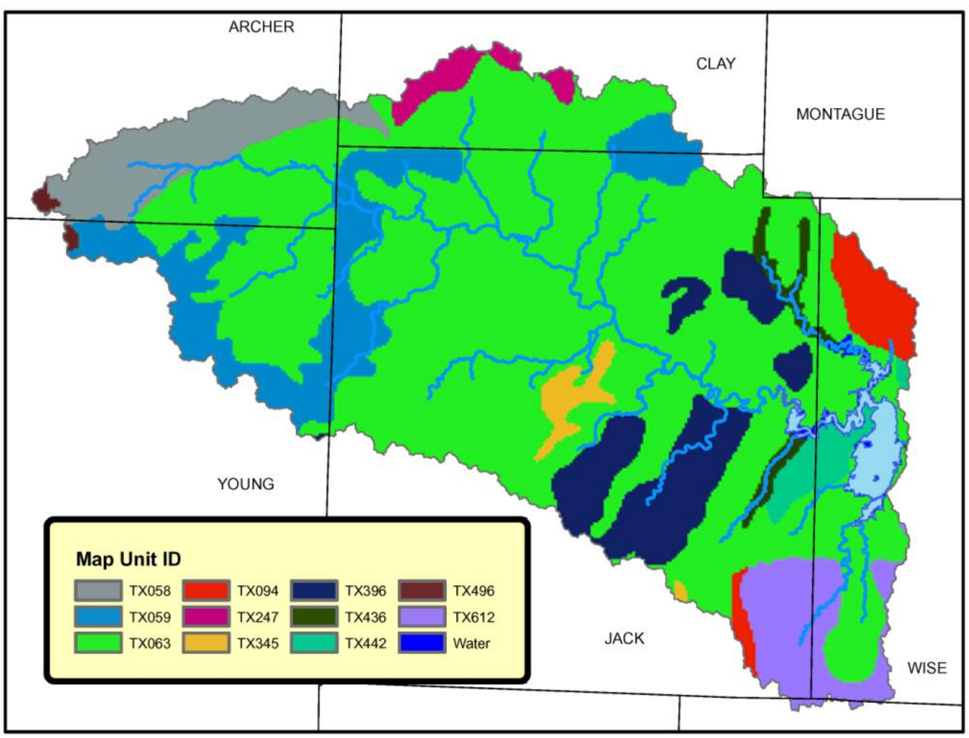


Figure 9.3 As in chapter 8, data from the State Soil Geographic Database (STATSGO) defined soil attributes in the Bridgeport Basin SWAT model.

Land use

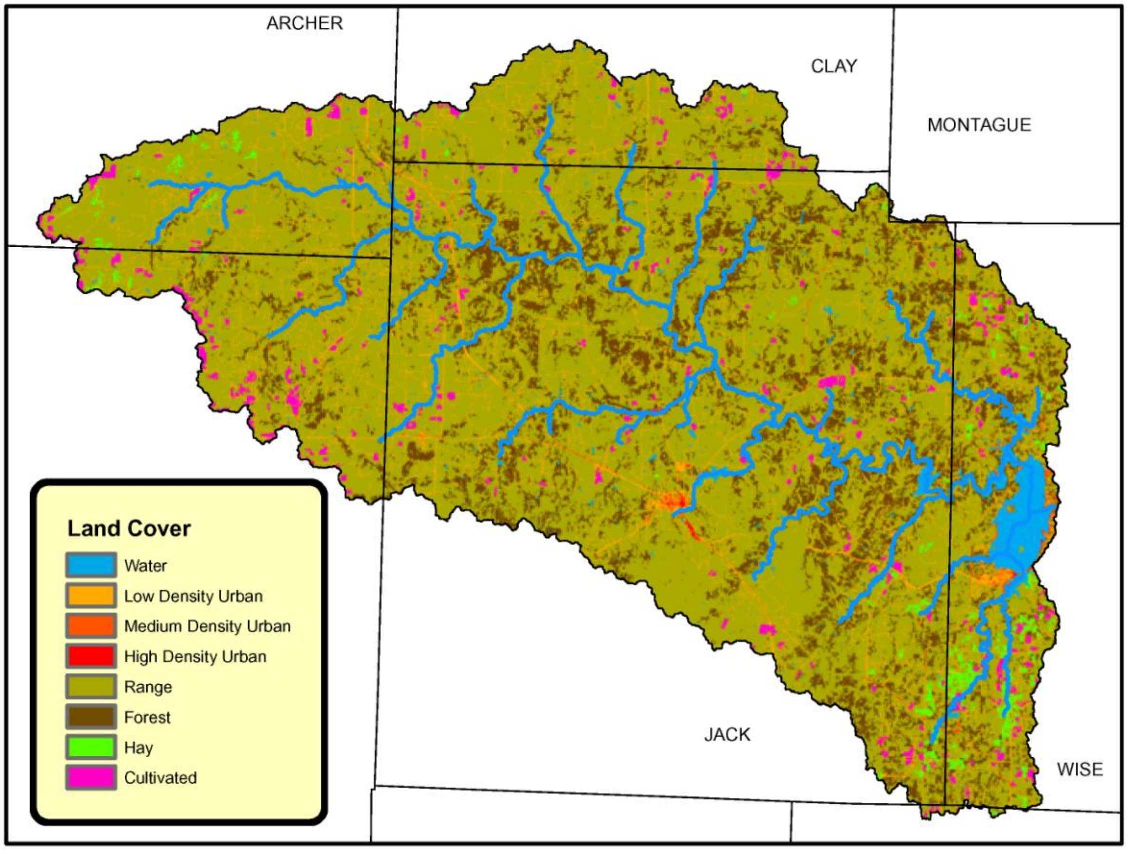


Figure 9.4 2001 National Land Cover Data defined the land cover of the Bridgeport Basin.

Weather

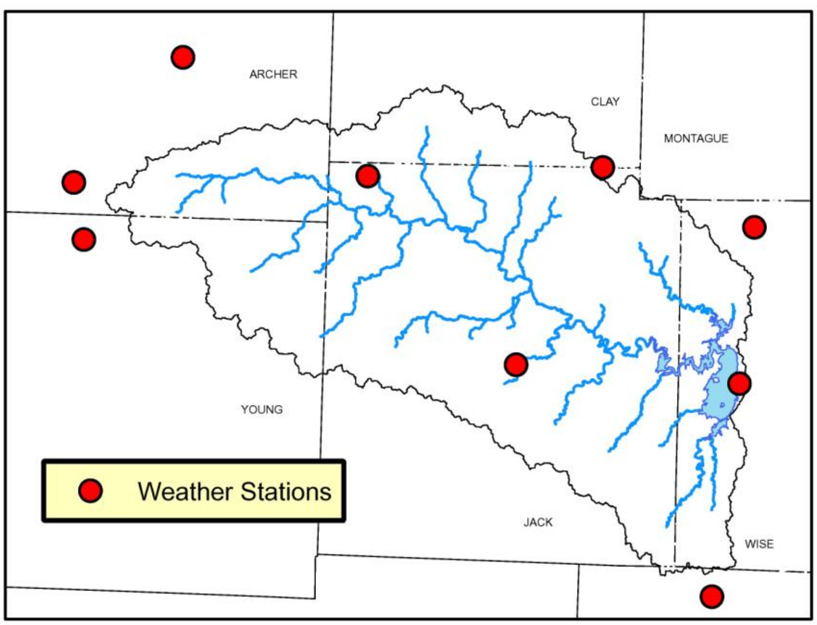


Figure 9.5 As described in the weather section of chapter 2, National Weather Service cooperative weather stations (shown here) provided minimum and maximum temperature and rainfall data from 1970 to 2005.

Point Sources

Instead of using the USGS Water Resource Database in this basin, the EPA's Permit Compliance System provided discharged data for six active wastewater treatment plants (figure 9.6). All six had measured discharge data but lacked useful nitrogen and phosphorus concentration data. We excluded point sources discharging less than 0.036 MGD (136 m^3 / day) from the model as insignificant, leaving only three point sources (table A-1). Nutrient load values for Texas dischargers were unavailable to us during the initial study. Therefore, we calculated nutrient loads for these point sources using concentrations derived from a comprehensive survey of municipal wastewater dischargers in the Virginia portion of the Chesapeake Bay Basin (Wiedeman and Cosgrove, 1998). Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

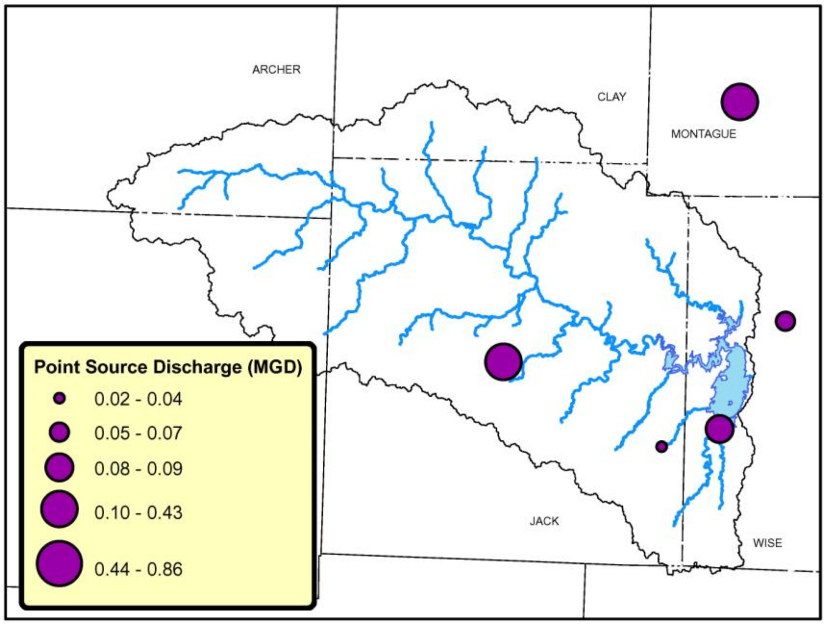


Figure 9.6 The location of point sources in the Bridgeport Basin by discharge in Millions of Gallons per Day (MGD).

Major Impoundments

Bridgeport Reservoir was constructed prior to the simulation period. Table 9.1 shows surface and storage characteristics provided by the National Inventory of Dams. Furthermore, Bridgeport Lake provided estimated daily discharge data for use in SWAT development. Approximately 43,779 acre-feet are removed from the reservoir annually for consumptive water use. SWAT utilized this data as average monthly consumption.

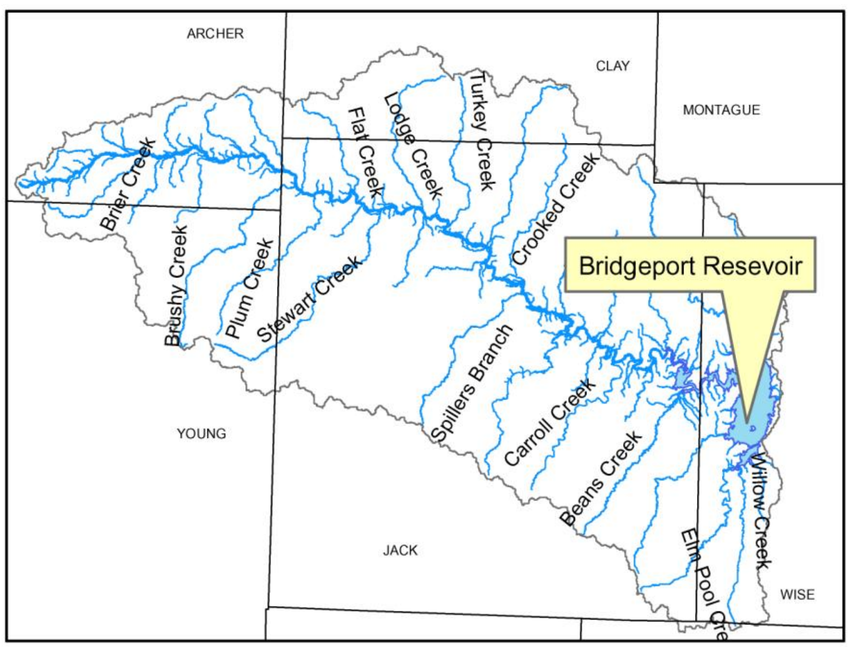


Figure 9.7 Major impoundments and streams in the Bridgeport Basin.

Table 9.1 Reservoir characteristics provided by the National Inventory of Dams.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (acre-feet)	Release
Bridgeport	50	13,318	386,386.0	19,966	923,451.0	Measured

Subbasin Delineation

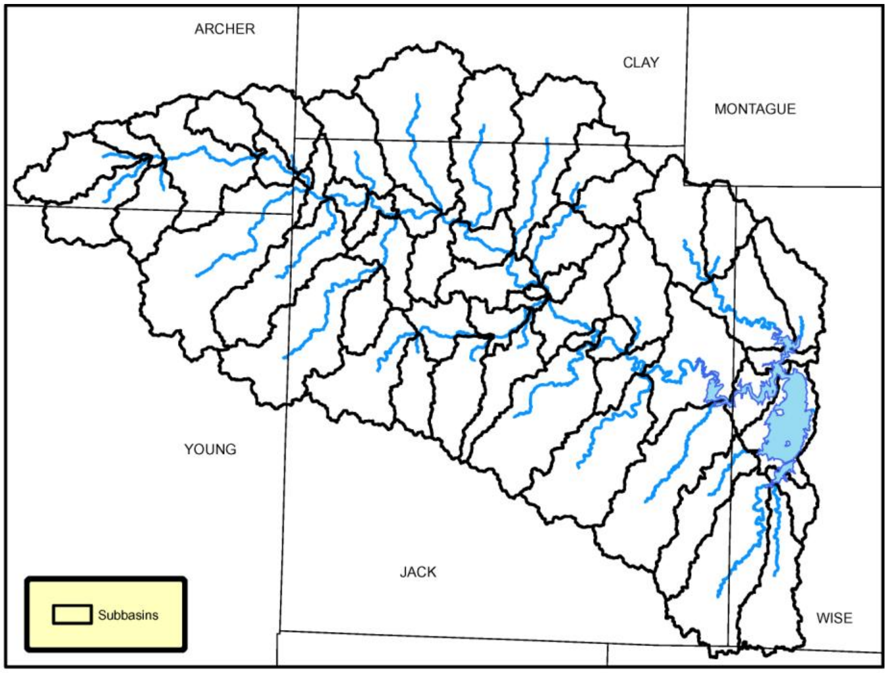


Figure 9.8 Using a stream threshold value of 2,471 acres, SWAT produced 57 subbasins within the Bridgeport Basin.

HRU Distribution

SWAT split each of the 57 subbasins into HRUs based on a user-defined threshold that determines the minimum percentage of any land cover required to designate an HRU within a subbasin. For the Bridgeport Basin, we reduced both the land use [%] over subbasin area threshold and the soil class [%] over subbasin area threshold to 0%. By reducing these thresholds to 0%, SWAT represented all land cover and soil class combinations. We also used slope to break HRUs into two categories, those with less than or greater than 5% slope. Our threshold selection resulted in 1,487 HRUs.

Soil Chemistry

SWAT applied a default value of five milligrams of labile phosphorus per kilogram of soil to rangeland and forest. This value corresponded to a Mehlich III Soil Test Phosphorus (STP) of approximately 20 pounds per acre, a consistent value for soils with no fertilization history and a phosphorus deficiency. We also set pasture and cultivated lands to 10 milligrams of labile phosphorus per kilogram of soil (50 lb/acre STP). Finally, we set urban areas to a value of 40 milligrams of labile phosphorus per kilogram of soil (200 lb/acre), a value derived from urban lawn and garden soil tests in Delaware County, Oklahoma (Storm et al., 2005).

Ponds

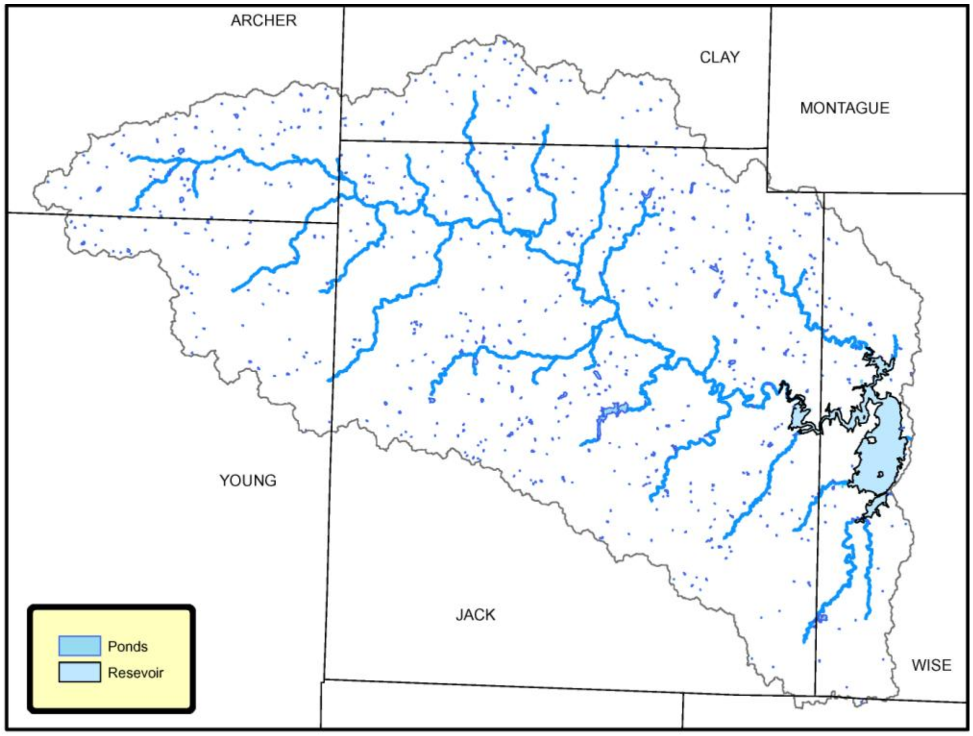


Figure 9.9 National Land Cover Data (2001) defined ponds in the Bridgeport Basin at a resolution of 30 meters (98 feet).

For this study, we used the National Land Cover Dataset rather than the National Inventory of Dams because NLCD data captures many more of the small but numerous ponds in this basin. We defined a total of 3,063 acres of ponds with 30-meter (98-foot) data (figure 9.9). As in chapter 8, we estimated the drainage area of each pond assuming a fixed surface to drainage area ratio of 25:1. Whitis (2002) recommends drainage to surface area ratios between 30:1 and 5:1 depending on land use and soil type within the drainage area. Under these assumptions, approximately 12% of the entire basin drains into ponds. As introduced in Chapter 2, we assumed that the ponds were 6.6 feet in depth and initially 75% full of water—a starting point that changes after the first rainfall of the simulation. SWAT assigned all other pond parameters based on its default settings. For example, ponds were assumed to have only a primary spillway.

Model Calibration and Validation

Sediment and Nutrient Load Development

For model calibration and validation, the use of loads is much preferred over discrete concentration grab sample data. We used observed water quality data to predict sediment, total phosphorus and nitrate loads at a single site in the basin (figure 9.10). These loads were estimated by station using the USGS DOS program LOADEST2. Developed by USGS supervisory hydrologist Charles Crawford (1996), this program estimates loading using the rating curve method.

A total of 65 total suspended solids samples were collected at the gauge between 1972 and 2007. The accuracy of the LOADEST2 predictions depends upon the quantity and distribution of the data provided by those samples. Using the sampling data, LOADEST2 predicted an average annual suspended sediment load of 25,000 metric tons (27,558 tons) per year from 1970 to 2008. Using 64 samples for a similar period, the model predicted a total phosphorus load of 38,000 kilograms (83,776 pounds) per year, and nitrate-nitrogen samples numbering 63 yielded a load of 30,000 kilograms (66,139 pounds) per year. The quantity of data derived from these loads was acceptable, but the distribution was less favorable. In each case, LOADEST2 warned that the measured flow record contained flow events more than two times larger than were present in the water quality data. This is due to insufficient high flow sampling in the measured water quality data. The extrapolation of concentration data outside the measured range increases the uncertainty in estimated loads. Although noteworthy, this is a fairly typical of most water quality sampling programs.

Calibration

Bridgeport Basin had limited available data with which to calibrate the SWAT model. One USGS gauging station on the Trinity River near Jacksboro (08042800) provided flow data and was the only gauge available for calibration and validation (figure 9.10). We calibrated the model for streamflow at this site from 1985 to 2007 and validated the model from 1970 to 1984 (figure 9.11–9.12). At the same gauge, the model was calibrated for water quality parameters including total suspended solids, total phosphorus and nitrate-nitrogen using estimates from Loadest2. Model calibration parameter adjustments are given in table 9.2. Relative errors for the calibration and validation period are given in table 9.3.

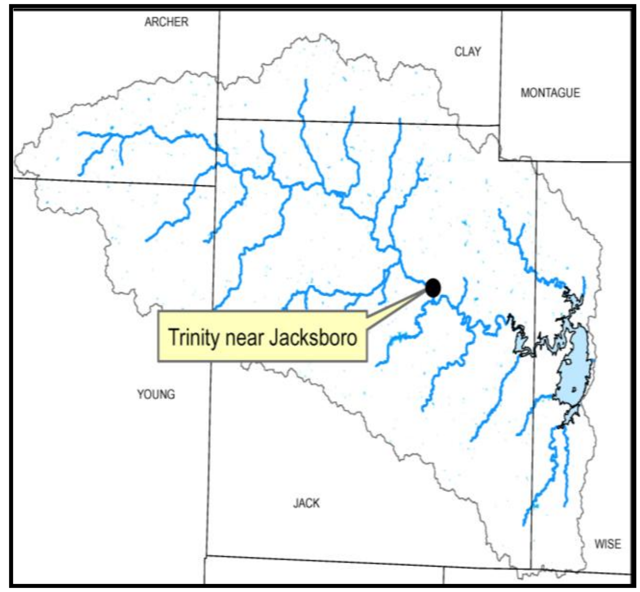


Figure 9.10 USGS gauges in the Bridgeport Basin.

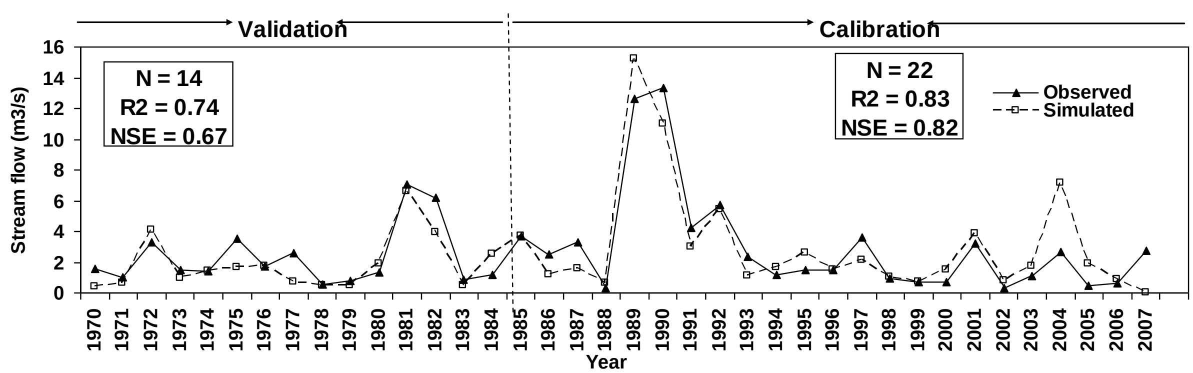


Figure 9.11 Annual streamflow at the West Fork of the Trinity River near Jacksboro for the calibration and validation periods.

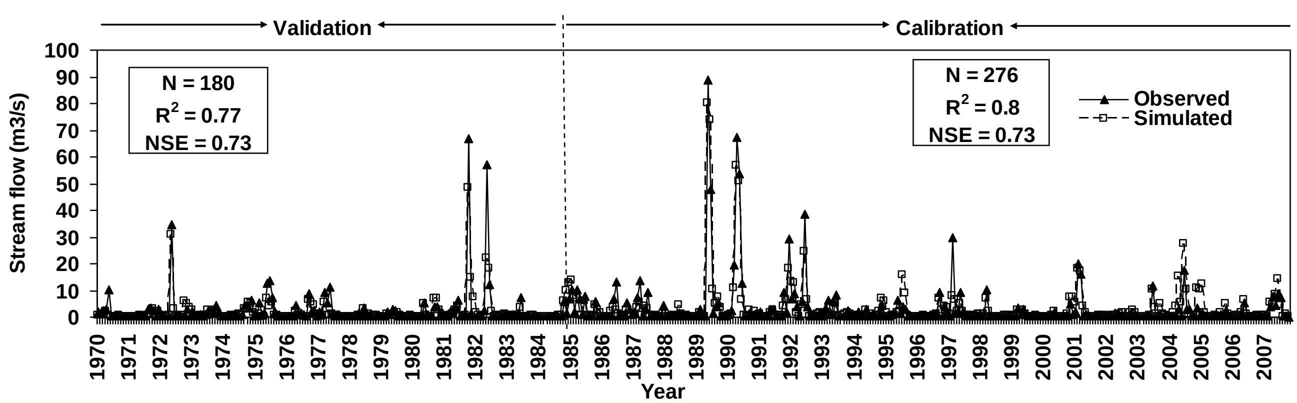


Figure 9.12 Monthly streamflow at the West Fork of the Trinity River near Jacksboro for the calibration and validation periods.

Table 9.2 Model calibration parameter adjustments.

Component	Variable	Description	Values used
Flow	**CN2**	Initial SCS runoff curve number for moisture condition II	Default-4
	ESCO	Soil evaporation compensation factor	0.5
	EPCO	Plant uptake compensation factor	0.5
	GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur	10
	REVAPMN	Threshold depth of water in the shallow aquifer required for “revap” to occur	10
	RCHRG_DP	Deep aquifer percolation fraction	0.25
	SOL_AWC1	Available water capacity of the soil layer	Default+0.04
	PHD_K	Hydraulic conductivity through bottom of ponds	0.5
	CH_K1	Effective hydraulic conductivity in tributary channel alluvium	0.5
Sediment	**SPCON**	Linear parameter for estimating the maximum amount of sediment that can be reentrained during channel sediment routing	0.0006
	CH_EROD	Channel erodibility factor	0.05
	CH_COV	Channel cover factor	0.95
Nutrients	**IWQ**	Stream nutrient transformations	0
Nitrogen	**SDNCO**	Denitrification threshold water content	0.85
Phosphorus	**PHOSKD**	Phosphorus soil partitioning coefficient	50
	PSP	Phosphorus sorption coefficient	8
	PPERCO	Phosphorus percolation coefficient	20
	IPET	PET method: 0=Priest-T,1=Pen-M, 2=Harg, 3=user input	1
	ISUBWQ	Instream water quality:1=model instream water quality	0

Table 9.3 Model calibration and validation statistics.

Constituent	Period	Observed	Predicted	Relative Error
Streamflow m^3/s	Calibration (1985-2007)	3.01	3.05	-1%
	Validation (1970-1984)	2.32	1.86	20%
Total Suspended Solids Metric tons (tons)	Calibration (1970-2007)	24,886 (27,432)	24,783 (27,319)	0.4%
Total Phosphorus kg/yr (lb/yr)		38,159 (84,126)	36,001 (79,369)	5.7%
Nitrate Nitrogen kg/yr (lb/yr)		29,864 (65,839)	32,352 (71,324)	-8.3%

Model Predictions

The SWAT model can predict nutrient and sediment loading without calibration, but the results contain excessive uncertainty unsuitable for many applications. Only enough measured data were available to estimate loads and calibrate the Bridgeport model at a single location. The quantity and distribution of measured data available at this single site create a great deal of uncertainty in the model estimation of sediment and nutrient loads.

The model predicted that overland areas of the Bridgeport Basin generated 45,000 metric tons (49,604 tons) of sediment per year under current management scenarios. Sediment generated from overland areas combined with additional streambank erosion caused reservoir loading of 74,000 metric tons (81,571 tons) per year. The relative uncertainty in these predictions is very high due to the lack of sediment data as referenced in Chapter 2. Furthermore, sediment loads for Bridgeport included only total suspended solids, so the model was not calibrated to include bed load. Sediment is generated non-uniformly across the basin (figure 9.13), but in general, overland range areas with steeper slopes had higher sediment yields. Cumulative sediment losses from overland areas are given in figure 9.14.

The model predicted that overland and point sources generated 75,000 kilograms (165,347 pounds) of phosphorus and 120,000 kilograms (264,555 pounds) of nitrogen. These nutrients were subject to little attenuation during the routing portion of the model. Therefore, 74,000 kilograms (163,142 pounds) of phosphorus and 113,000 kilograms (249,122 pounds) of nitrogen actually reached Bridgeport Reservoir annually during the simulation period. Urban areas had the highest nutrient yields. Total phosphorus and nitrate-nitrogen losses are mapped in figure 9.15.

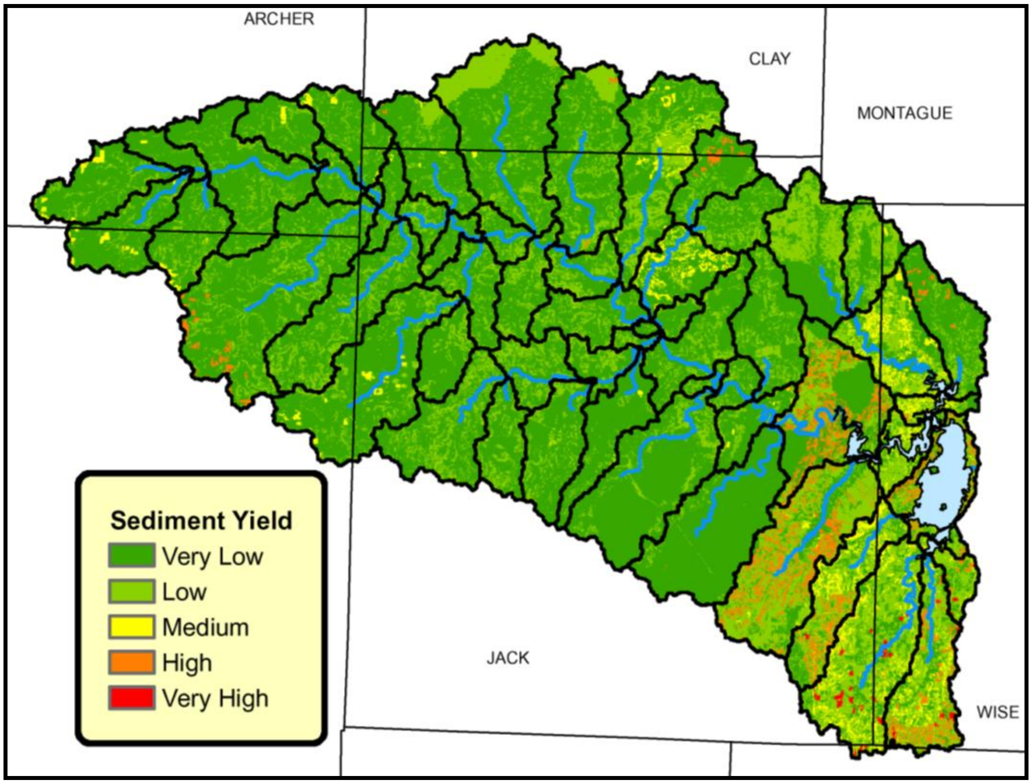


Figure 9.13 SWAT-predicted sediment yield distribution for the Bridgeport Basin.

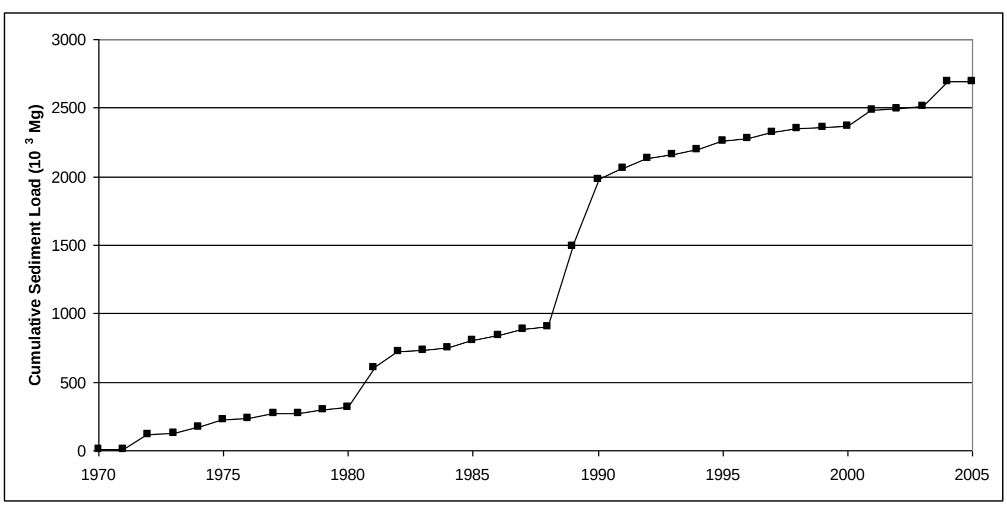


Figure 9.14 Cumulative Bridgeport Basin sediment losses as predicted by SWAT (1970 to 2005).

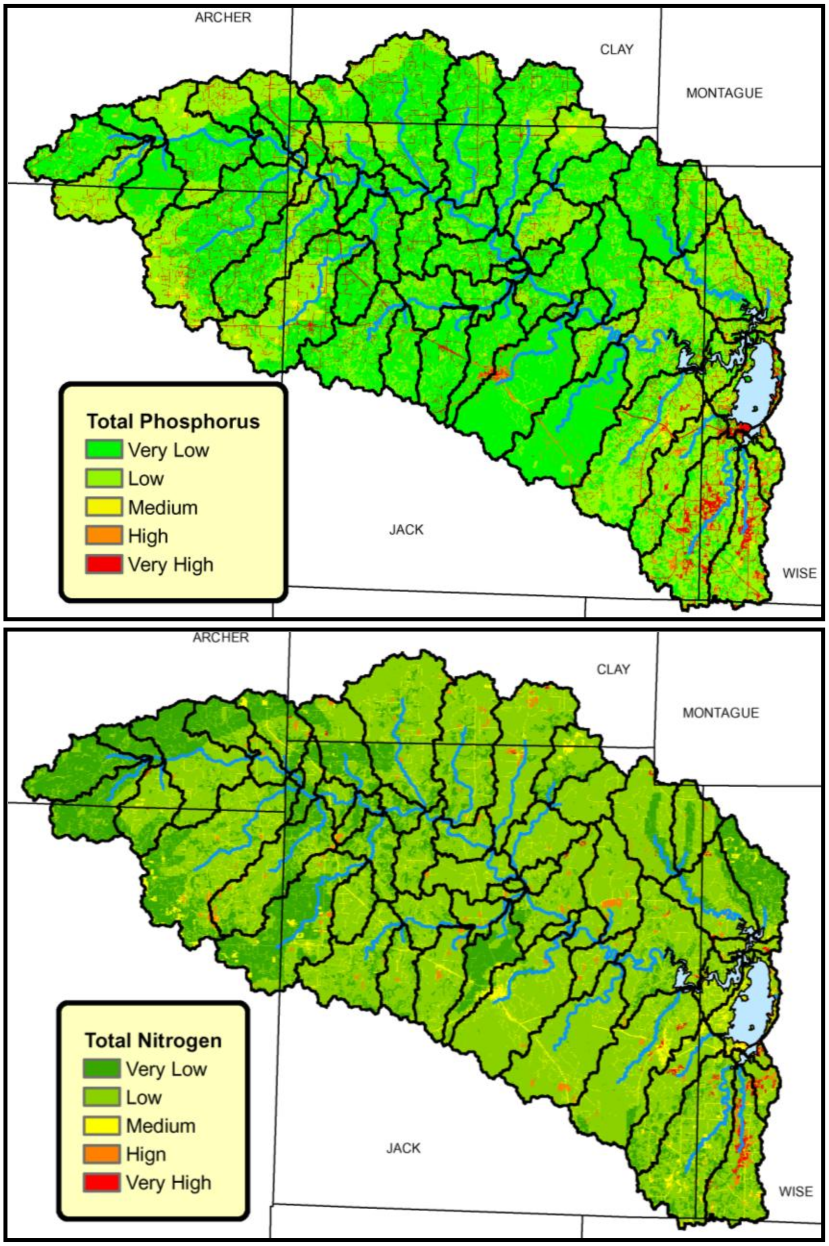


Figure 9.15 Distributions of Phosphorus (top) and nitrogen (bottom) losses as predicted by the SWAT model for the Bridgeport Basin.

Scenarios

Conservation Practices

We simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 9.4.

Ponds

By removing all ponds from the Bridgeport Basin SWAT model, we found that basin ponds reduced sediments by 9.6%, and the construction of 25% more ponds could potentially reduce sediment loads by an additional 2.5%. Placing ponds in the critical sediment source areas shown in figure 9.13 could significantly improve their effectiveness.

Range Utilization

Removing grazing from rangelands resulted in a 7.6% reduction in sediment loads from overland areas.

Cropland Conversion

There is little cultivated cropland in the Bridgeport Basin (2.1%), yet sediment losses from cultivated areas can be much greater than forest or grassland. We assumed cropland in the basin consisted of wheat, primarily for winter grazing. In this scenario, the model was modified to convert all cultivated areas to hay, resulting in a 19% reduction in sediment from overland areas.

Point Source Load Elimination

The basin contains only two point sources, which accounted for 14% of total phosphorus loads and 23% of total nitrogen loads in the stream. The elimination of these discharges should reduce loads by a similar amount.

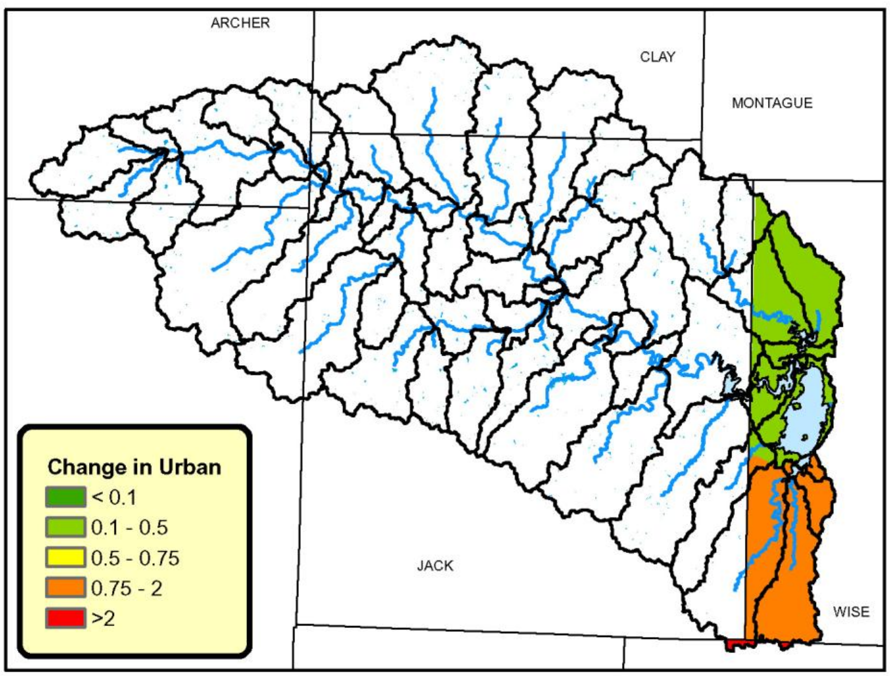


Figure 9.17 Projected change in urban area from 2000 to 2030. The average net increase of 59% in the Bridgeport Basin based on the partial coverage area was used for areas outside of the available projected area.

Urbanization

Using NCTCOG population projections, the model predicted the effects of urbanization on sediment, phosphorus and nitrogen in 2030 (figures 9.18–9.20). Urban area in the basin was predicted to increase by 59% (figure 9.17). This fractional increase was used for areas for which no population increase data were available. For the urbanization scenario, the model predicted the greatest change in nutrient loads, with nitrogen and phosphorus increasing by 10.3% and 30.7%, respectively, with a small reduction in sediment of only 1.7%.

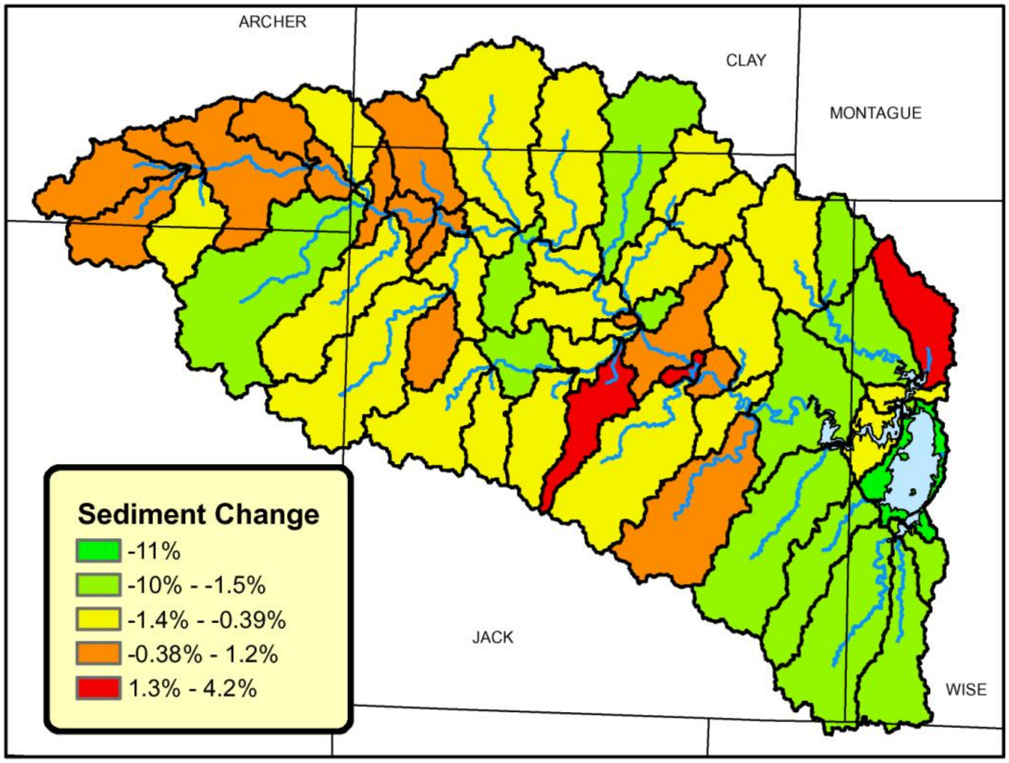


Figure 9.18 Change in overland sediment yields from 2000 to 2030 as predicted by SWAT.

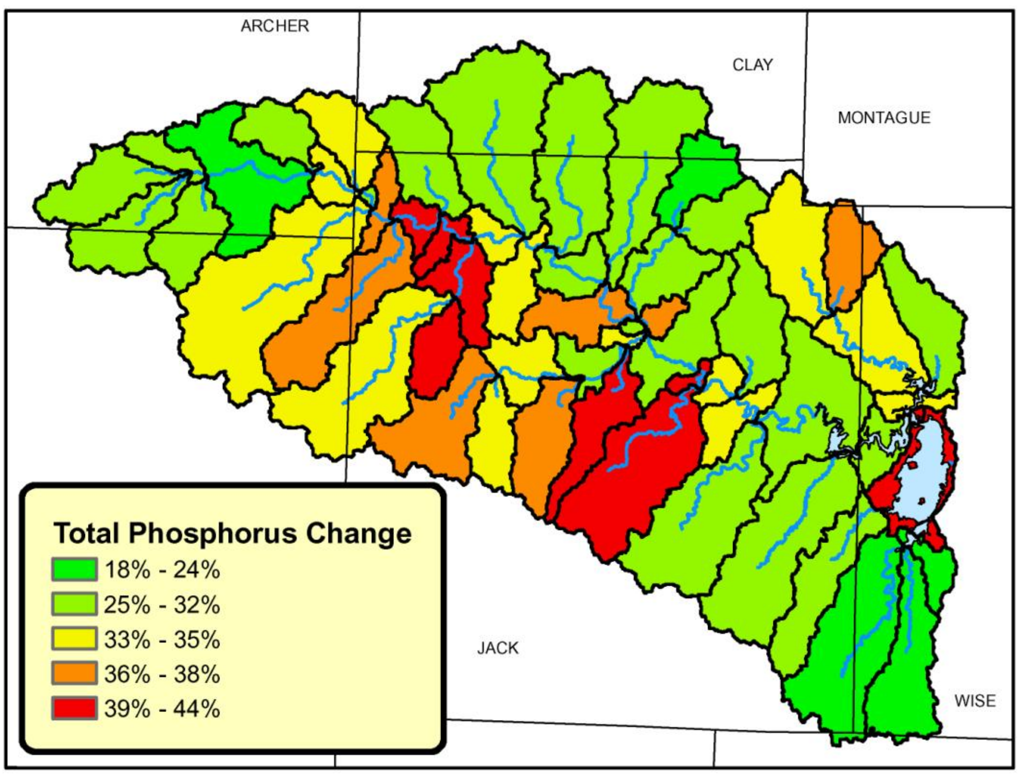


Figure 9.19 Change in overland phosphorus yields from 2000 to 2030 as predicted by SWAT.

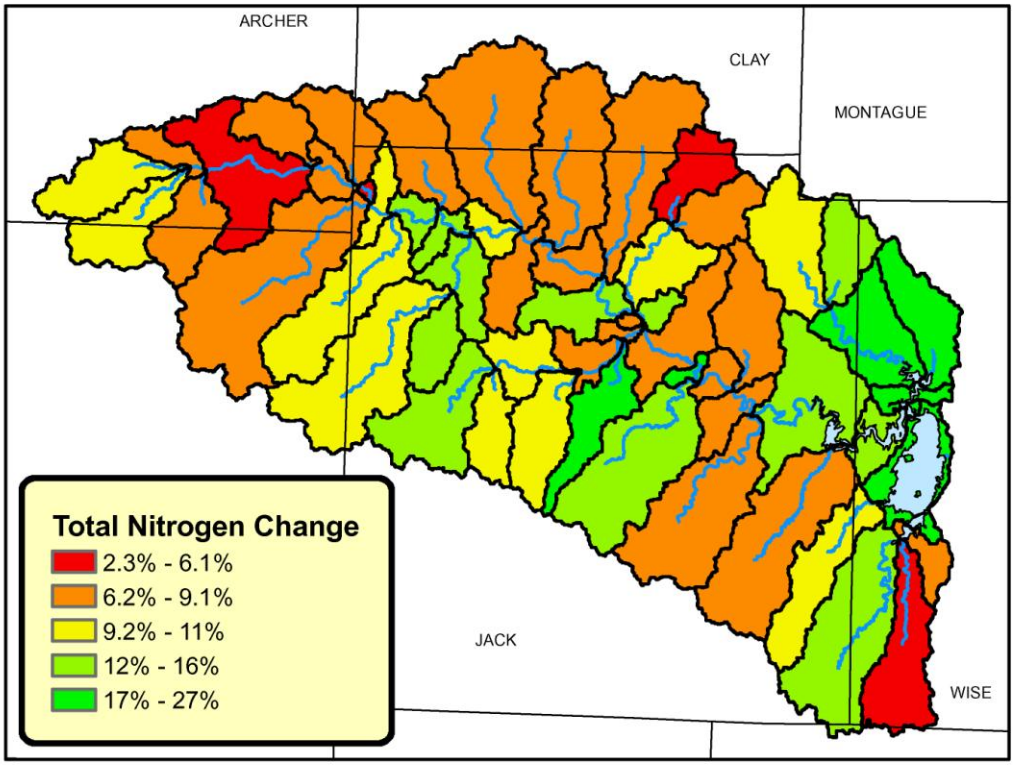


Figure 9.20 Change in overland nitrogen yields from 2000 to 2030 as predicted by SWAT.

Total Conservation Practice Impact

If all proposed conservation practices were implemented, including the construction of 25% more ponds and the elimination of point-source contributions, nutrient loads could be significantly reduced (table 9.4). If all practices were implemented, sediment delivered to streams could be reduced by 29%. Cumulative reductions in overland sediment loads are given in figure 9.21.

Table 9.4 Changes in sediment and nutrient loads under different scenarios as derived from SWAT model simulations.

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	74,000 metric tons/yr (81,571 tons/yr)	74,000 kg/yr (163,142 lb/yr)	113,000 kg/yr (249,122 lb/yr)
No Ponds	9.6%	8.6%	5.9%
No Range Grazing	-7.6%	-9.7%	-19%
Urban *	-1.7%	30.7%	10.3%
No Point Sources	-0.0%	-2.9%	-13%
Cropland to Hay Conversion	-19%	-1.1%	-5.4%
25% More Ponds	-2.5%	-1.9%	-1.5%

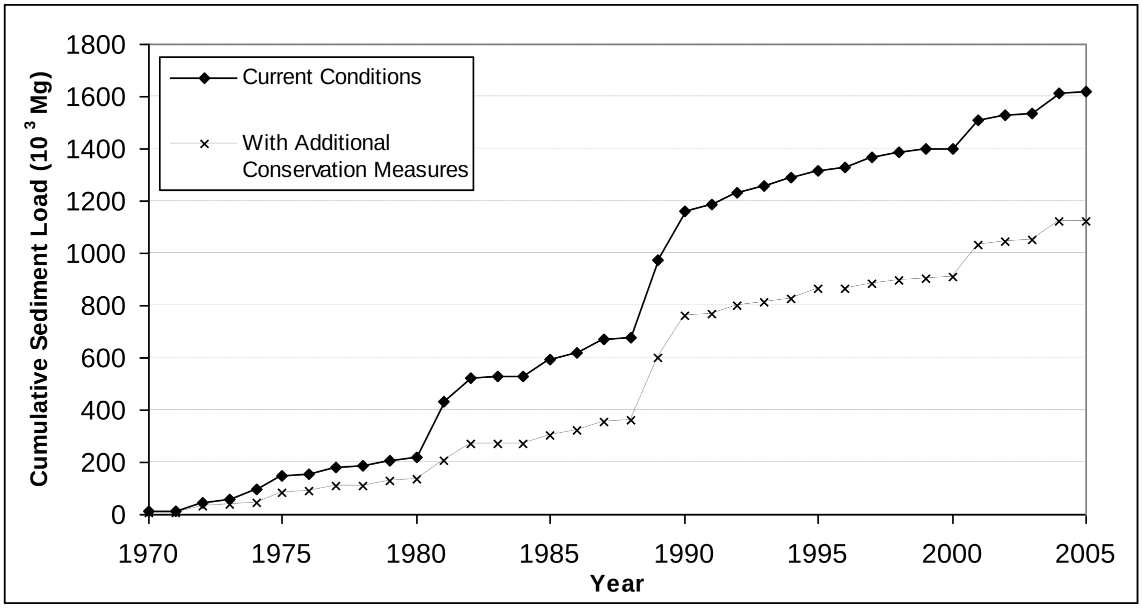


Figure 9.21 SWAT-predicted cumulative overland sediment yields with and without proposed conservation measures utilized in the Bridgeport Basin (1970 to 2005).

Conclusions

Using SWAT, we evaluated the impacts of point sources, ponds, rangeland grazing, cropland conversion and urbanization on hydrologic and water quality variables in the Bridgeport Basin. The USGS gauging station on the Trinity River near Jacksboro provided flow and water quality records (estimated with LOADEST2) for calibration (1985–2007) and validation (1970–1984).

Over 38 years (1970–2007), the model predicted that an average of 74,000 metric tons (81,571 tons) of sediment, 74,000 kilograms (163,142 pounds) of phosphorus and 113,000 kilograms (249,122 pounds) of nitrogen reach Bridgeport Reservoir every year.

Scenario analyses included the removal of ponds, point sources, rangeland grazing and conversion of cropland into hay. The model also examined the effects of urban expansion predicted for 2030. Existing ponds, which are essentially PL-566 reservoirs, helped by settling about 9.6% of sediment, 8.6% of total phosphorus and 5.9% of total nitrogen generated and transported in the basin. Eliminating rangeland grazing and maintaining range grass reduced sediment, total phosphorus and total nitrogen by 7.6%, 9.7% and 19%, respectively. Point sources add 2.9% of total phosphorus and 13% of total nitrogen. Converting cropland to hay, as expected, resulted in a substantial reduction (19%) in sediment. The model also predicted that urban expansion in the basin would increase total phosphorus and total nitrogen loads by 31% and 10%, respectively.

References

See the appendix.

Chapter 10: Richland-Chambers Basin

Introduction

The watershed modeling objective of this project was to assess the effects of urbanization and other land use changes on sediment and nutrient delivery to Richland-Chambers Reservoir using the Soil and Water Assessment Tool. Richland-Chambers Reservoir, covering an area of 45,003 acres, is the third largest lake contained entirely within the state of Texas. Construction of the lake began in 1982 and ended in 1987. The reservoir has a drainage area of 1,274,322 acres. This drainage area will hereafter be referred to as Richland-Chambers Basin. The reservoir serves as a major drinking water supply reservoir for over 1.6 million people in the north Texas area, and covers parts of Navarro, Ellis, Hill, Johnson, Freestone and Limestone counties (figure 10.1). The reservoir has two main arms: Richland Creek and Chambers Creek.

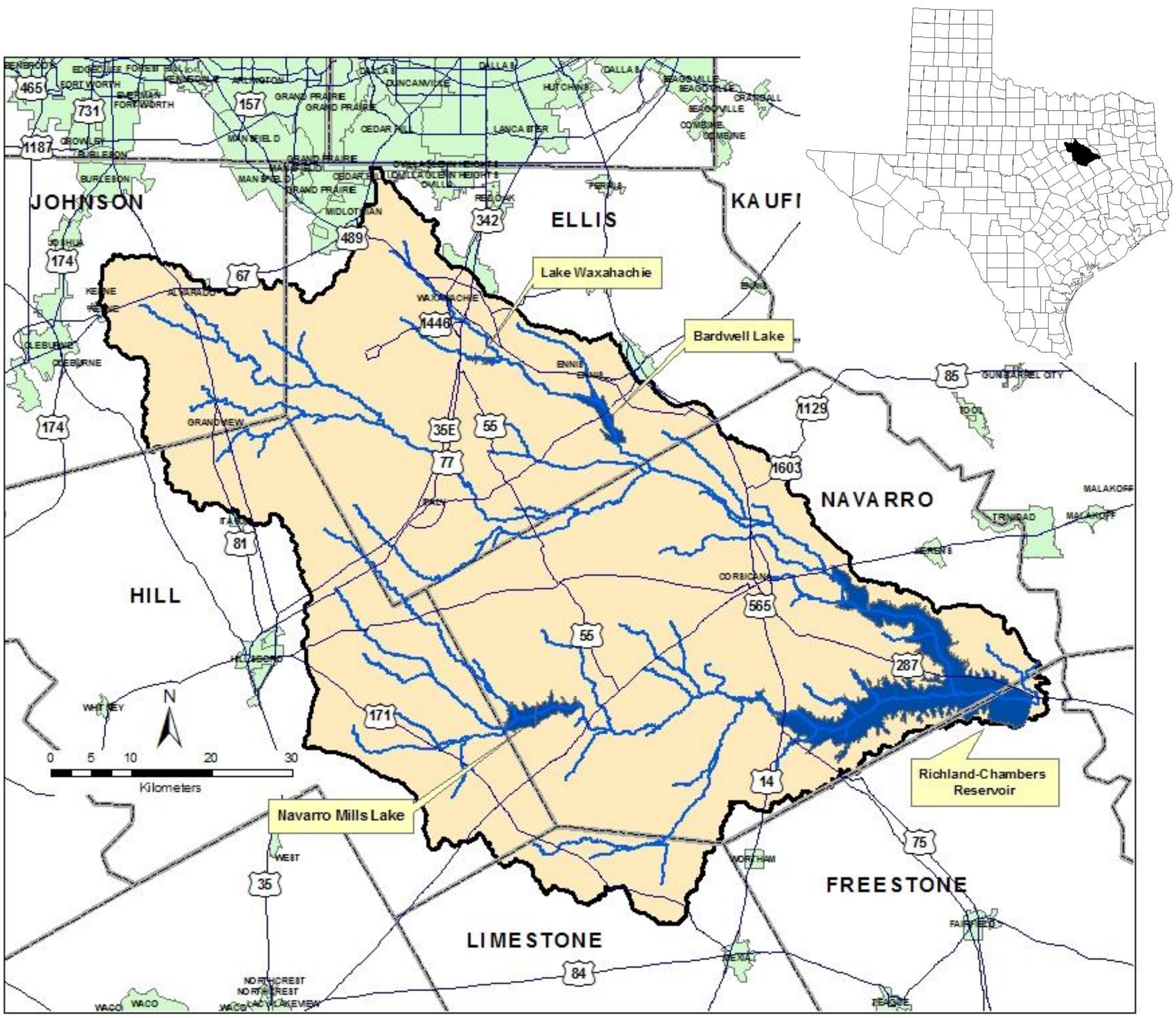


Figure 10.1 The location of Richland-Chambers Basin.

Model Input Data Tables and Figures

Topography

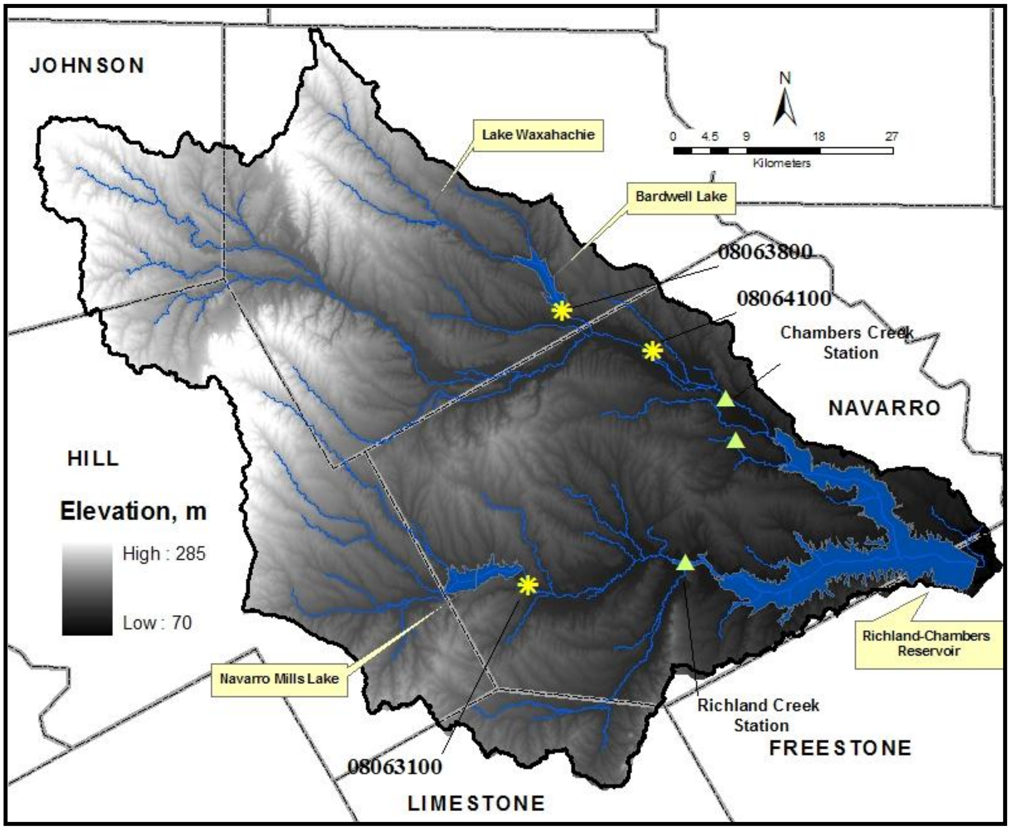


Figure 10.2 A 30-meter (98-foot) Digital Elevation Model (DEM) defined the topography of the Richland-Chambers Basin.

Soils

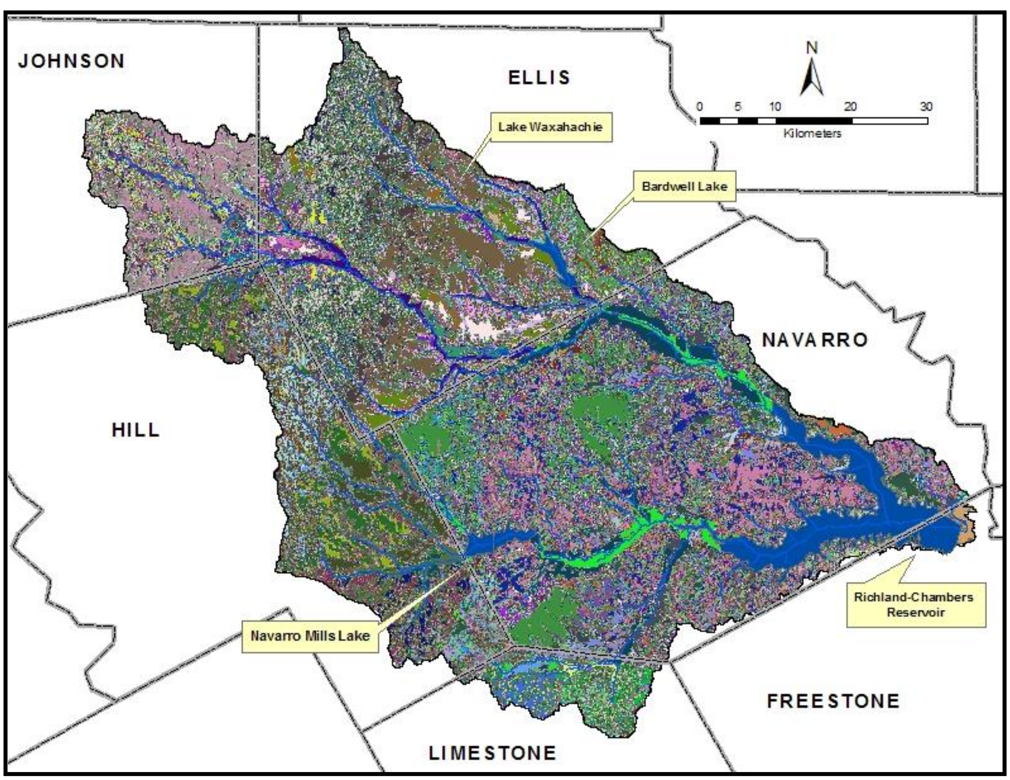


Figure 10.3 The Soil Survey Geographic (SSURGO) database defined soil attributes in the Richland-Chambers SWAT model.

Land Use

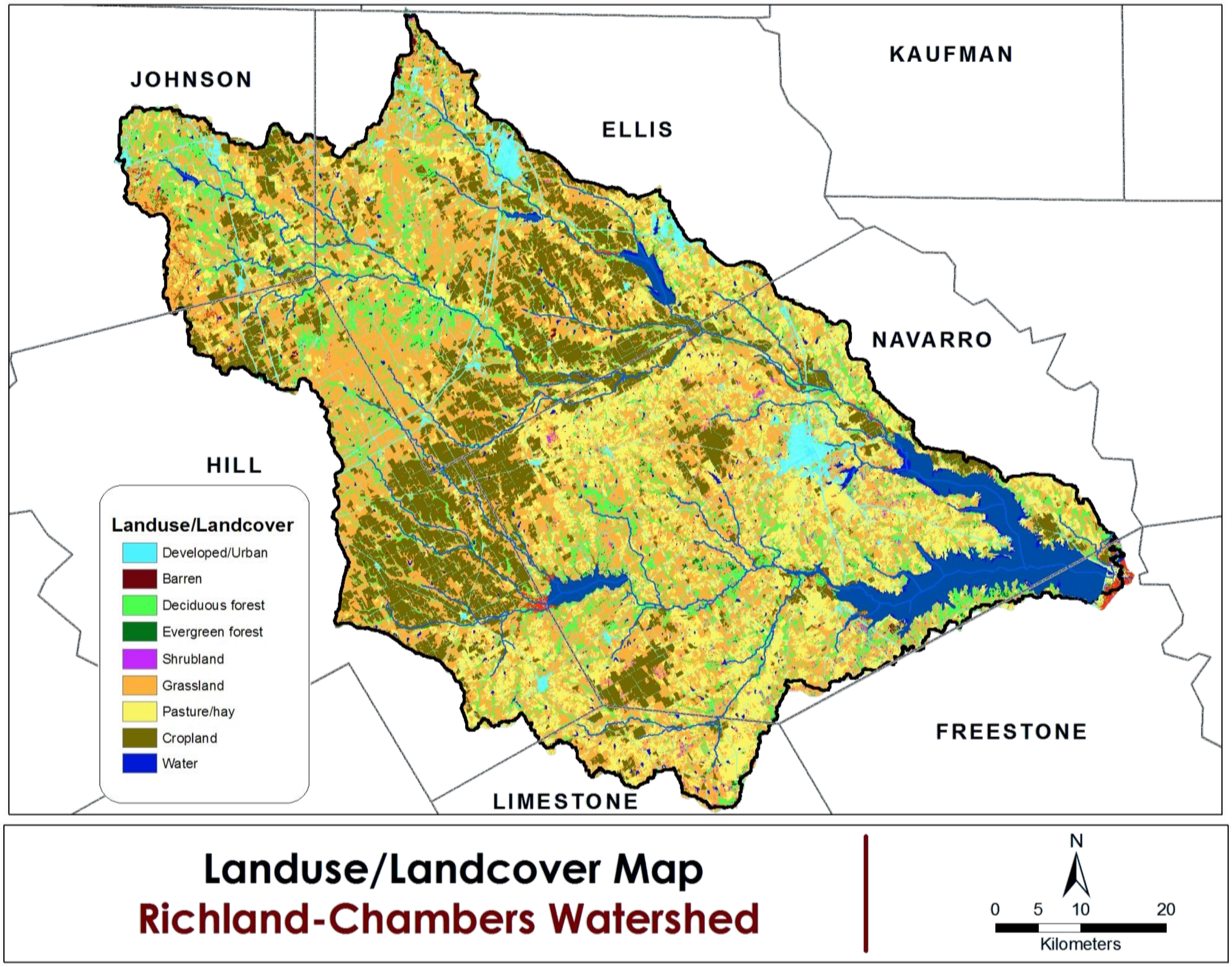


Figure 10.4 2001 National Land Cover Data defined the land cover of the Richland-Chambers Basin.

Pasture

As in previous simulations, pasturelands were composed of bermudagrass. However, in the Richland-Chambers SWAT model, we simulated grazing on over 75% of pastureland while the rest grew hay with three cuttings per year. The stocking rate considered for grazing on pastureland was one animal unit per three acres (Personnel communication, Navarro County Soil and Water Conservation district).

Cultivated

In the Richland-Chambers Basin, SWAT simulated winterwheat (32%) as the dominant crop followed by corn (30%), sorghum (22%) and cotton (16%). SWAT used typical management inputs related to the type and dates of tillage and the type, rates and dates of fertilizer applications.

Weather

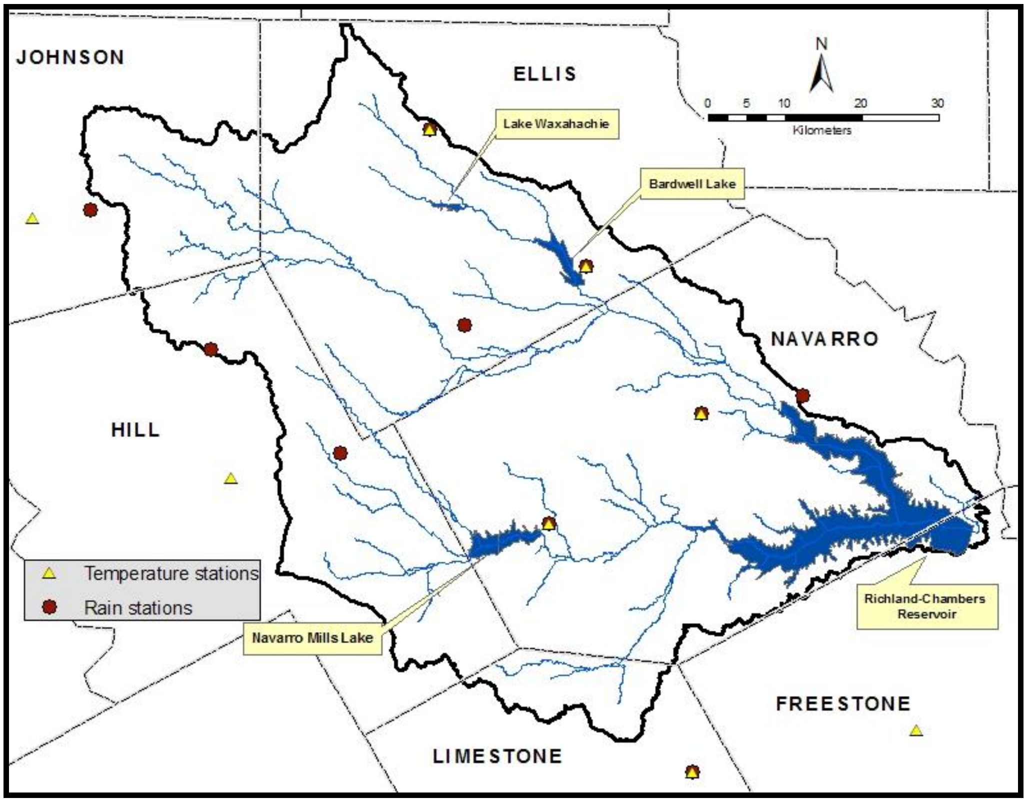


Figure 10.5 As described in the weather section of chapter 2, National Weather Service cooperative weather stations provided minimum and maximum temperature and rainfall data from 1970 to 2006.

Point Sources

The EPA's Permit Compliance System listed 20 active WWTPs, all with measured discharge data (figure 10.6). Most of them had ammonia-nitrogen, CBOD and dissolved oxygen data, but none had any nitrogen or phosphorus concentration data. Regardless, all 20 WWTPs were included in the model. Point source input data, including discharges and permitted limits, for each watershed are given in Appendix table A-1.

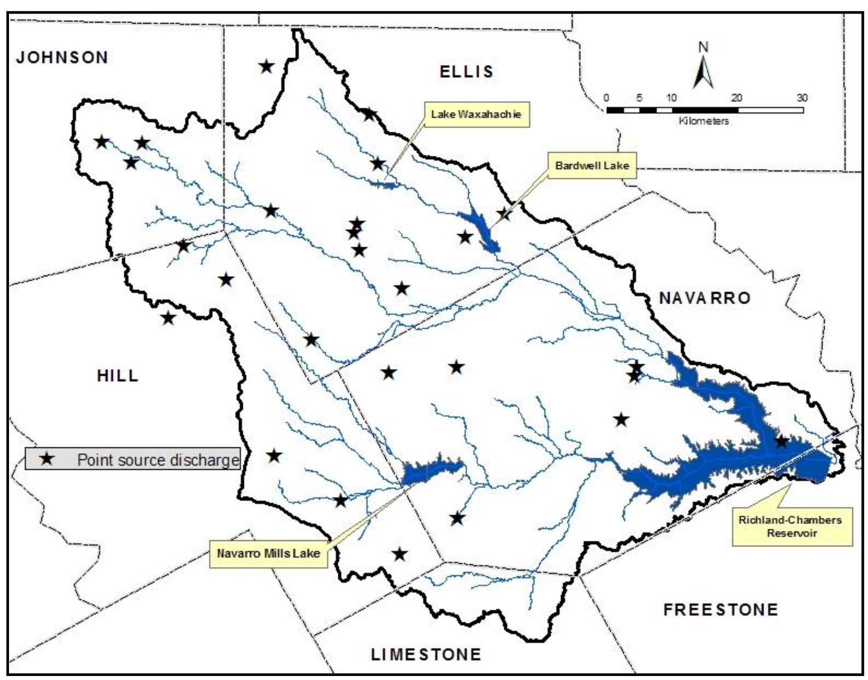


Figure 10.6 WWTPs in the Richland-Chambers Basin.

Ponds

The U.S. Army Corps of Engineers' National Inventory of Dams (USACE, 1982) supplied reservoir and pond data including dam locations and dimensions. A total of 307 PL-566 reservoirs, including Bardwell, Waxahachie and Navarro Mills Lakes, (figure 10.7) were included in the simulation. These three lakes are the largest reservoirs within Richland-Chambers Basin (figure 10.1). We used lake outflow data available from 1972 for both Bardwell and Navarro Mills in the development of the SWAT model. Aside from these large lakes, all PL-566 reservoirs were modeled as ponds.

Table 10.1 The National Inventory of Dams provided the reservoir characteristics for Lake Bardwell, Lake Navarro Mills and Richland-Chambers Reservoir

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Bardwell	27	3138	4.65	9350	13.80	Measured
Navarro Mills	123	5070	5.58	18520	20.66	Measured
Richland-Chambers	117	41356	113.66	136104	375.09	Measured

Subbasin Delineation and HRU distribution

We used a stream threshold value of 6,301 acres to delineate subbasins and manually added several outlet points, resulting in 156 subbasins (figure 10.7). Using SSURGO soils (figure 10.3) and 2001 NLCD land use/land cover data (figure 10.4), the 156 subwatersheds were further divided into 3,687 HRUs.

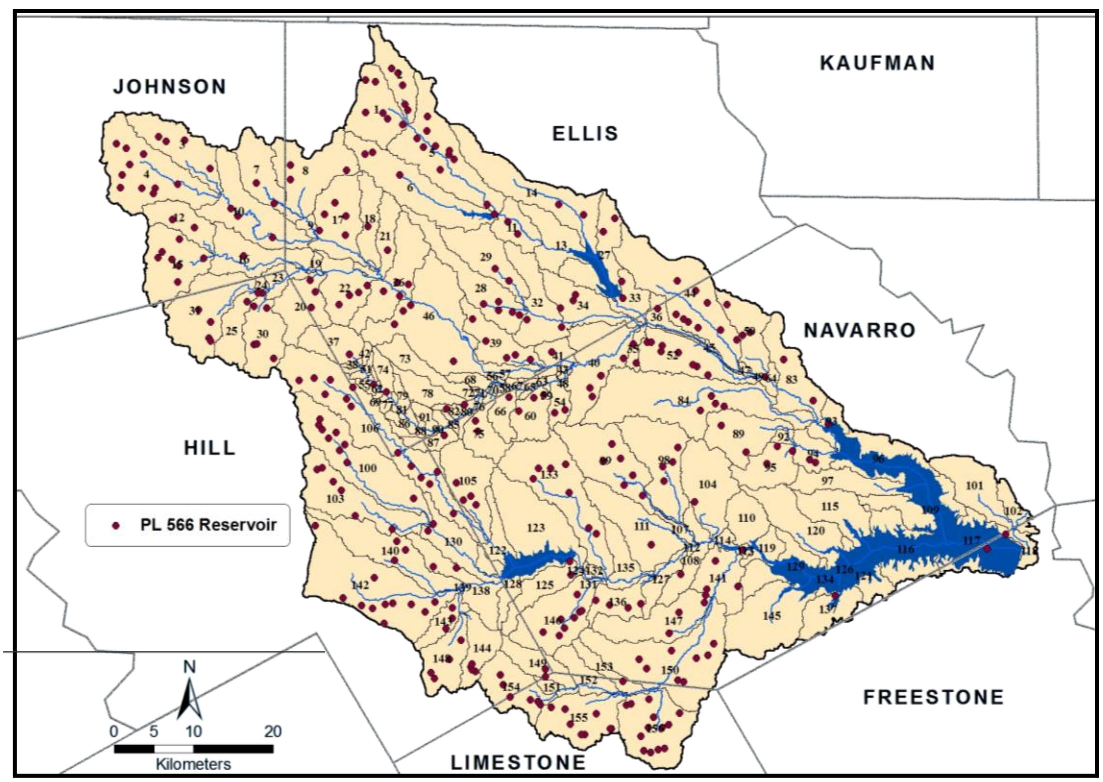


Figure 10.7 Subbasins used in the Richland-Chambers SWAT Model.

Model Calibration and Validation

Calibration

To calibrate SWAT for flow, sediment and nutrients, we used data from USGS gauging stations and monitoring stations managed by Tarrant Regional Water District (TRWD). All three USGS gauging stations, shown in figure 10.2, had long-term continuous records of observed streamflow data. However, no continuous monitoring records for sediment and nutrient data were available within this watershed. However, USGS gauge 08604100 and all three TRWD monitoring stations had grab sample data for the calibration period (usually 2–5 samples per year, with a few years missing in some cases). The model was calibrated at all three USGS gauging stations on a monthly and annual basis for the period 1982–1995, the first two years serving as a model warm-up period. During calibration, we carefully matched the proportion of surface flow and baseflow that contributed to streamflow, analyzing baseflow with the baseflow filter program (Arnold and Allen, 1999; Arnold et al., 1995; Nathan and McMahon, 1990).

Due to limited sampling data, we could not perform a rigorous calibration of sediment and nutrients. However, we adjusted certain model parameters, giving careful consideration to key overland and channel processes influencing the model-simulated pollutant loads.

To evaluate model-predicted streamflow during calibration and validation, we used mean, standard deviation, coefficient of determination (R^2) and Nash-Sutcliffe modeling efficiency (NSE) (Nash and Sutcliffe, 1970). We also compared mean observed data with mean simulated flow as well as sediment and nutrient loadings from days with available grab sample data.

For validation, the Army Corps of Engineers' hydrologic data Website (USACE, 2007) provided inflow estimates for Bardwell Lake and Navarro Mills Lake. TRWD provided observed data on Richland-Chambers Reservoir. We compared these observed data with simulated streamflow values. This is known as spatial validation, where model simulations are validated for the same period as calibration but at a different location(s).

Results and Discussion

Model Calibration and Validation

Table 10.2 shows model parameters used in calibration, and table 10.3 presents calibration results for measured and simulated annual and monthly flow data at all three USGS gauging stations. The absolute percent difference between measured and simulated flows at both annual and monthly time steps was no greater than 4%. The model performed very well, with both R^2 and NSE ≥0.90 at USGS gauging stations 08064100 and 08063100. Model performance was satisfactory at the annual time step at USGS gauging station 08063800, based on ratings by Moriasi et al. (2007) (see Chapter 2). Figure 10.8 illustrates the time series of measured and simulated monthly streamflow at USGS gauging station 08064100.

Table 10.2 Calibration parameters used in the Richland-Chambers SWAT model.

Model component	Variable	Description	Range	values used in this study
Flow	CN2	Initial SCS runoff curve number for moisture condition II	-5 – 5	-4
	ESCO	Soil evaporation compensation factor	0.01 – 1.00	0.55
	EPCO	Plant uptake compensation factor	0.01 – 1.00	1.0
	GW_REVAP	Groundwater revap coefficient	0.02 – 0.40	0.02
	GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur	0.0 – 300.0	250
Sediment	SPCON	Linear parameter for estimating the maximum amount of sediment that can be reentrained during channel sediment routing	0.0001 – 0.01	0.01
	CH_COV	Channel cover factor	0.0 – 1.0	0.8
	CH_EROD	Channel erodibility factor	0.0 – 1.0	0.056 – 0.075
	C-factor	Land surface cover factor	0.003 –0.45	Corn: 0.2 Cotton: 0.2 Sorghum: 0.2 Wheat: 0.03 Range: 0.007 Pasture: 0.007
	SPEXP	Exponent parameter for estimating the maximum amount of sediment that can be reentrained during channel sediment routing	1.0 – 2.0	1.0
	CH_N(2)	Channel Manning's roughness coefficient	0.014	0.02
Sediment and nutrients	RSDCO	Residue decomposition coefficient	0.01 – 0.05	0.05
Mineral nitrogen	CDN	Denitrification exponential rate coefficient	0.0 – 3.0	0.3
Nitrogen in reach	NPERCO	Nitrate percolation coefficient	0.01 – 1.0	0.9
	AI1	Fraction of algal biomass that is nitrogen	0.07 – 0.09	0.09
Mineral phosphorus	RS4	Rate coefficient for organic N settling in the reach at 20ºC (day-1)	0.001 – 0.1	0.001
	BC3	Rate constant for hydrolysis of organic N to NH4 in the reach at 20ºC (day-1)	0.2 – 0.4	0.3
	BC2	Rate constant for biological oxidation of NO2 to NO3 in the reach at 20ºC (day-1)	0.2 – 2.0	2.0
	PPERCO	Phosphorus percolation coefficient	10.0 – 17.5	10
Phosphorus in reach	PHOSKD	Phosphorus soil partitioning coefficient	100 - 400	350
	AI2	Fraction of algal biomass that is phosphorus	0.01 – 0.02	0.01
	BC4	Rate constant for mineralization of organic P to dissolved P in the reach at 20ºC (day-1)	0.01 – 0.70	0.01
	RS5	Organic phosphorus settling rate in the reach at 20ºC (day-1)	0.001 – 0.1	0.1
Nitrogen and phosphorus	CMN	Rate factor for humus mineralization of active organic nutrients (N and P)	0.0001 – 0.0003	0.0003
Nitrogen/ phosphorus in reach	MUMAX	Maximum specific algal growth rate (day-1)	1.0 – 3.0	1.0

Table 10.3 Summary of model performance statistics analyzing flow measured at USGS gauging stations during the calibration period (1984-1995).

Station	Time-step	Mean		Std. Dev		R^2	NSE
		Measured	Simulated	Measured	Simulated		
08064100	Annual	14.66	14.3	5.87	5.95	0.94	0.93
	Monthly	14.69	14.33	19.85	16.66	0.91	0.90
08063100	Annual	5.73	5.79	2.96	3.22	0.99	0.98
	Monthly	5.74	5.83	7.97	8.14	0.98	0.98
08063800	Annual	3.39	3.54	1.70	1.81	0.63	0.55
	Monthly	3.40	3.54	5.21	4.07	0.67	0.44

Because USGS stations 08063100 and 08063800 lacked water quality data, only data from station 08064100 was used to calibrate the model for sediment and nutrients. Additionally, the TRWD monitoring stations on Richland Creek and Chambers Creek had limited grab sample data on sediment and nutrients, which we compared with SWAT-predicted values. At USGS station 08064100, model simulated sediment, organic nitrogen and mineral nitrogen were close to observed values (within 4%) whereas simulated average mineral and total phosphorus were higher due to a large overprediction by the model on a few days (table 10.4). SWAT over-predicted all constituents, except organic nitrogen and total phosphorus, measured at TRWD monitoring stations (figure 10.9). Due to limited sampling data, matching daily simulated values with observed values using only those days of observation is tedious. Additional monitoring data would be very helpful in adequately calibrating and validating model-predicted loadings.

Table 10.4 Summary of model performance statistics used to evaluate water quality predictions at USGS gauging station 08064100 during the calibration period (1984–1995).

Component (unit)	# of samples	Mean		Std. dev.	
		Measured	Predicted	Measured	Predicted
Sediment in metric tons (t)	37	1541.50 (1699.21)	1487.00 (1639.14)	3249.40 (3581.41)	1865.38 (2056.23)
Organic N in kg (lbs)	91	1762.30 (3885.21)	1735.00 (3825.02)	5354.30 (11804.21)	14276.00 (31473.19)
Mineral N in kg (lbs)	41	3367.00 (7422.96)	3256.00 (7178.25)	7488.00 (16508.21)	3.38 (7.45)
Mineral P in kg (lbs)	41	50.00 (110.23)	64.31 (141.78)	104.70 (230.82)	135.45 (298.62)
Total P in kg (lbs)	91	443.00 (976.65)	800.00 (1763.70)	2041.00 (4499.63)	4482.00 (9881.11)

Table 10.5 shows the model performance statistics used to compare SWAT-simulated inflow with measured inflow to Lake Bardwell, Navarro Mills and Richland-Chambers Reservoir during validation. Figure 10.10 illustrates a time series of measured versus simulated monthly inflows into the Richland-Chambers Reservoir. Model-simulated cumulative inflow to Richland-Chambers was less than TRWD estimates by 1.3% (figure 10.11). The simulated sediment load received by Richland-Chambers Reservoir was 2,818,221 metric tons (3,106,557 tons) per year, which was less than the estimated value of 3,222,394 metric tons (3,552,081 tons) per year by 14% (1987–1995). The TRWD provided this estimated value based on the “volumetric and sedimentation survey of Richland-Chambers reservoir” by TWDB. The model predicted that the Richland-Chambers Basin generated 488,623 metric tons (538,615 tons) of sediment per year, considering no conservation management practices. The relative uncertainty in these

predictions is very high due to a lack of data with which to calibrate channel degradation and deposition of sediments.

Table 10.5 Summary of SWAT model performance statistics used to compare simulated and measured inflow (m^3/s) to reservoirs during validation (1984–1995).

Time-step	Mean		SD		R^2	NSE
	Measured	Simulated	Measured	Simulated		
Richland-Chambers Reservoir (1987-1995)						
Annual	40.82	39.42	13.30	15.14	0.80	0.73
Monthly	41.73	39.53	43.56	42.22	0.87	0.85
Bardwell Reservoir (1991-1995)						
Annual	5.00	4.98	0.55	1.00	0.98	0.94
Monthly	4.91	4.91	5.52	4.54	0.76	0.76
Navarro Mills Reservoir (1984-1995)						
Annual	6.74	5.10	2.52	2.05	0.78	0.59
Monthly	6.79	5.25	9.21	5.56	0.74	0.65

Over a 30-year period (1977–2006), the model predicted that overland flow within the Richland-Chambers Basin generated an average of 370,383 metric tons (408,277 tons) of sediment annually. As the simulation study showed, Richland-Chambers Basin is characterized by substantial channel erosion, as nearly 2,302,469 metric tons (2,538,037 tons) of sediment reach the reservoir every year. This value is much higher than the sediment load produced by runoff alone. For nutrients, runoff from overland areas combined with point sources generated 142,999 kilograms (315,259 pounds) of phosphorus and 4,047,437 kilograms (8,923,071 pounds) of nitrogen. The predicted annual delivery of total phosphorus and total nitrogen to the Richland-Chambers Reservoir amounted to 285,104 kilograms (628,547 pounds) and 4,011,580 kilograms (8,844,020 pounds), respectively. The spatial distribution of sediment, total phosphorus and total nitrogen losses from overland processes are illustrated in figures 10.12–10.14.

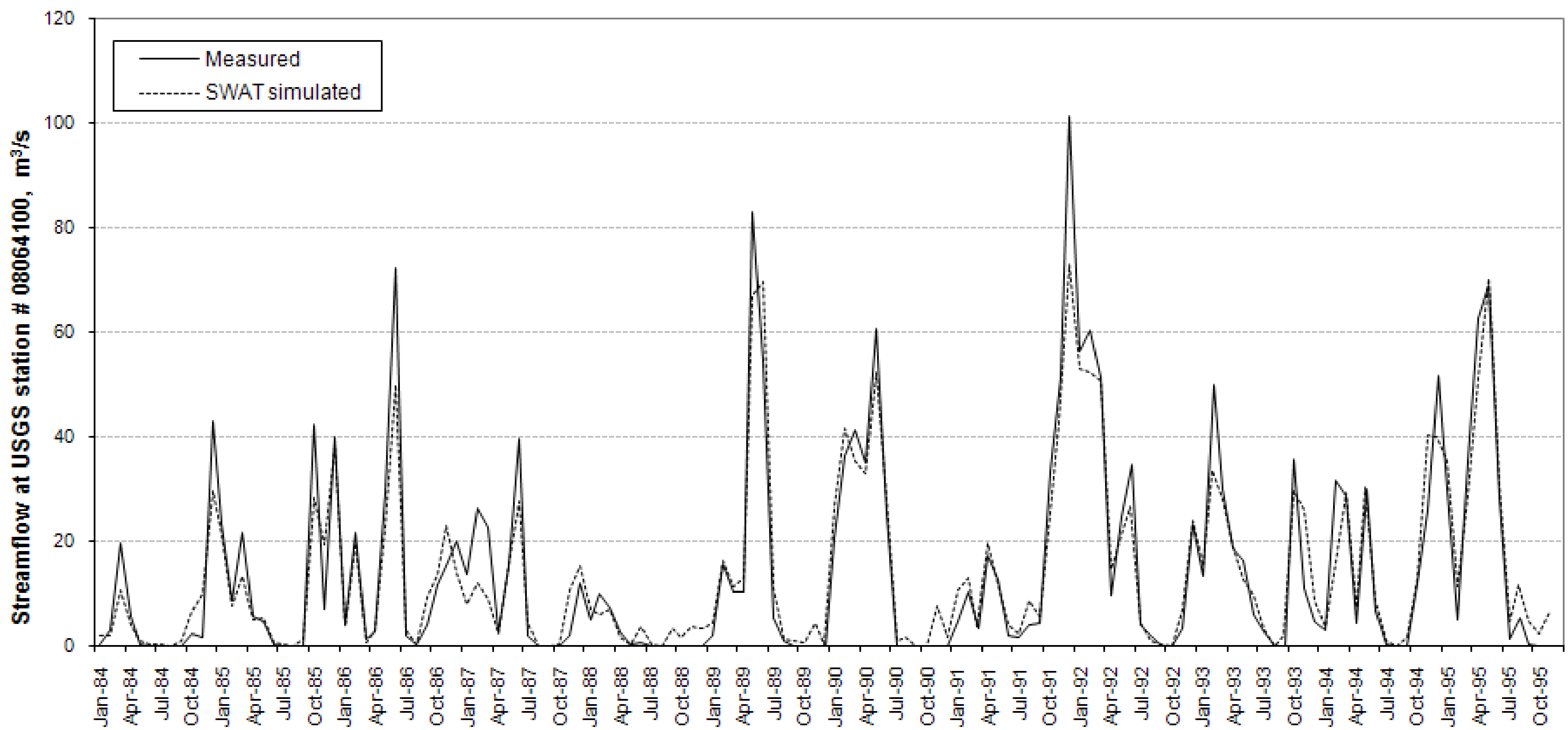


Figure 10.8 Flow calibration at USGS gauging station 08064100 during the calibration period (1984–1995).

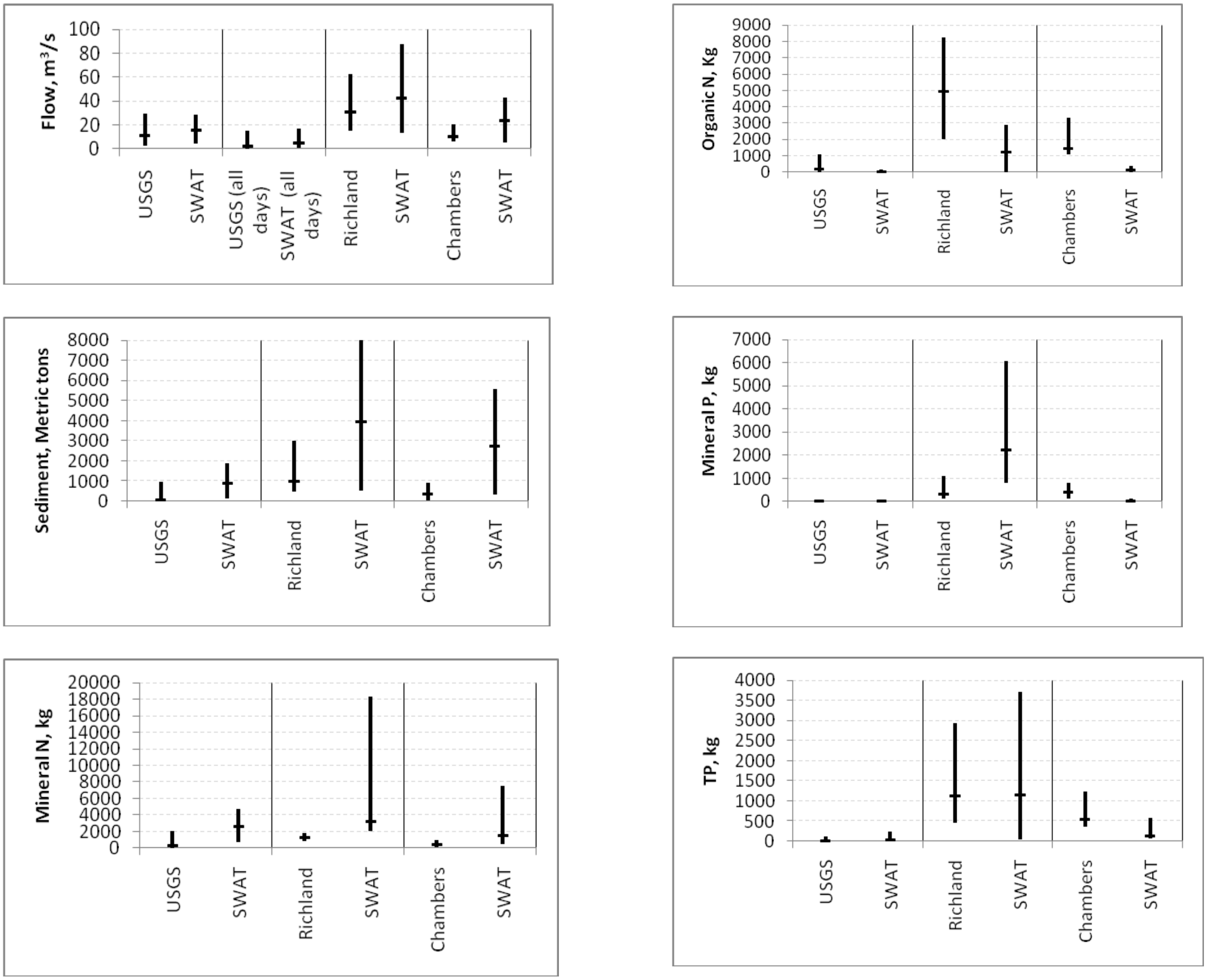


Figure 10.9 The median, 25th percentile and 75th percentile of measured and simulated streamflow, sediment, mineral nitrogen (mineral N), organic nitrogen (organic N), mineral phosphorus (mineral P) and total phosphorus (TP) at USGS gauge 08064100, Richland Creek and Chambers Creek monitoring stations during calibration (1984–1995).

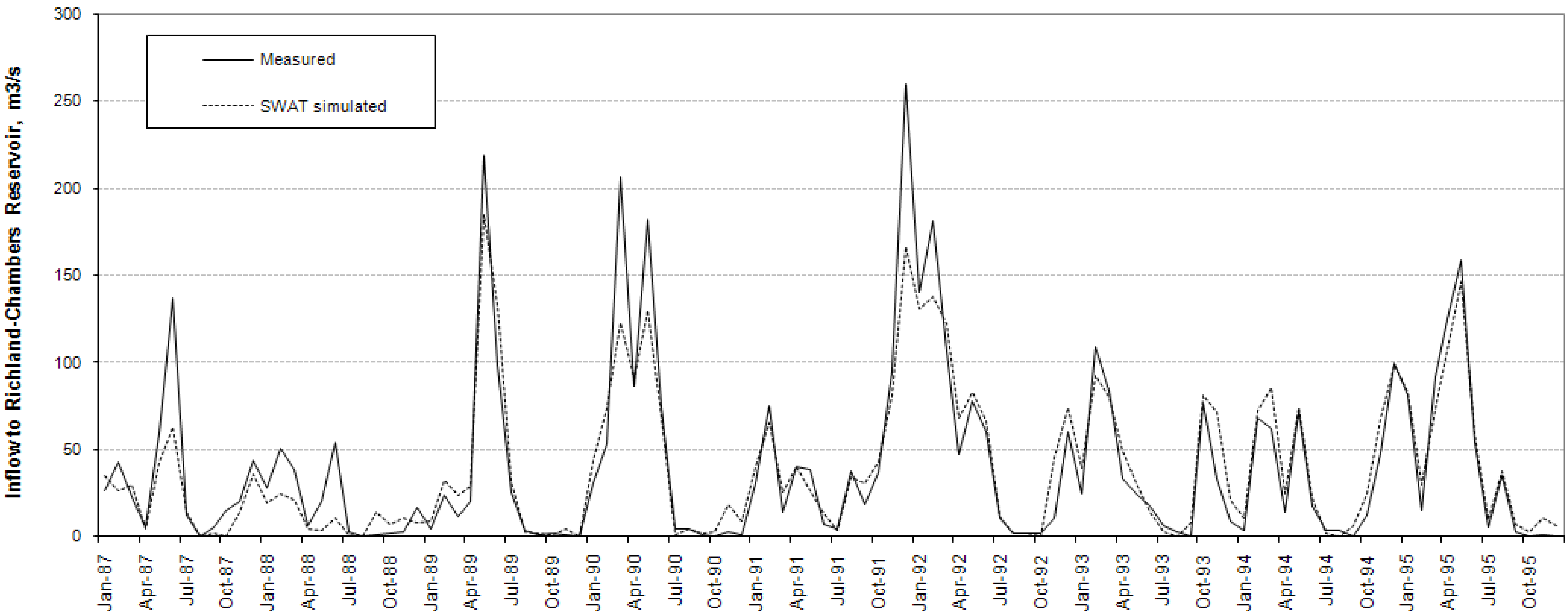


Figure 10.10 A monthly time series of measured versus SWAT-simulated flow into the Richland-Chambers Reservoir during validation (1984–1995).

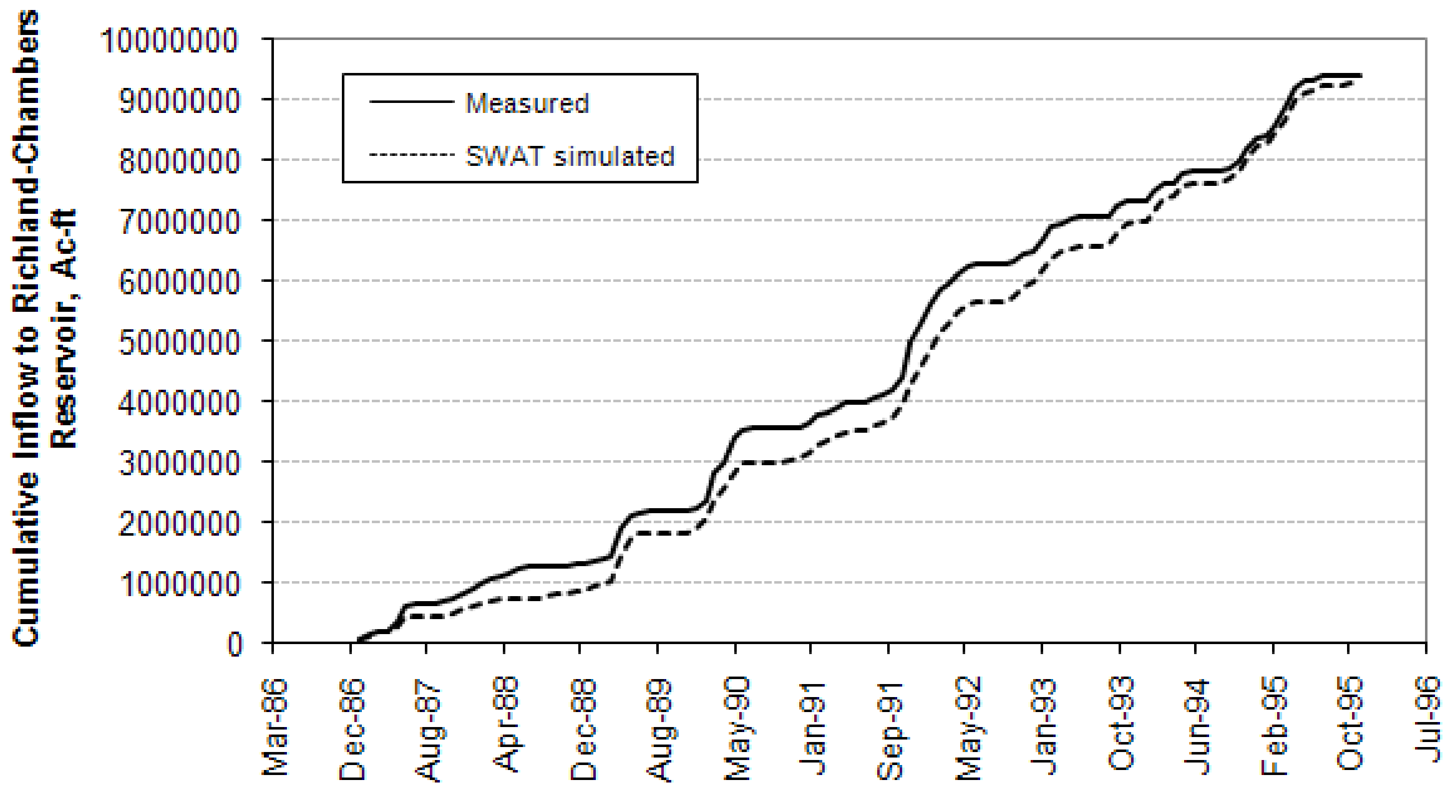


Figure 10.11 Measured versus SWAT-simulated cumulative monthly flow into the Richland-Chambers Reservoir during validation (1984–1995).

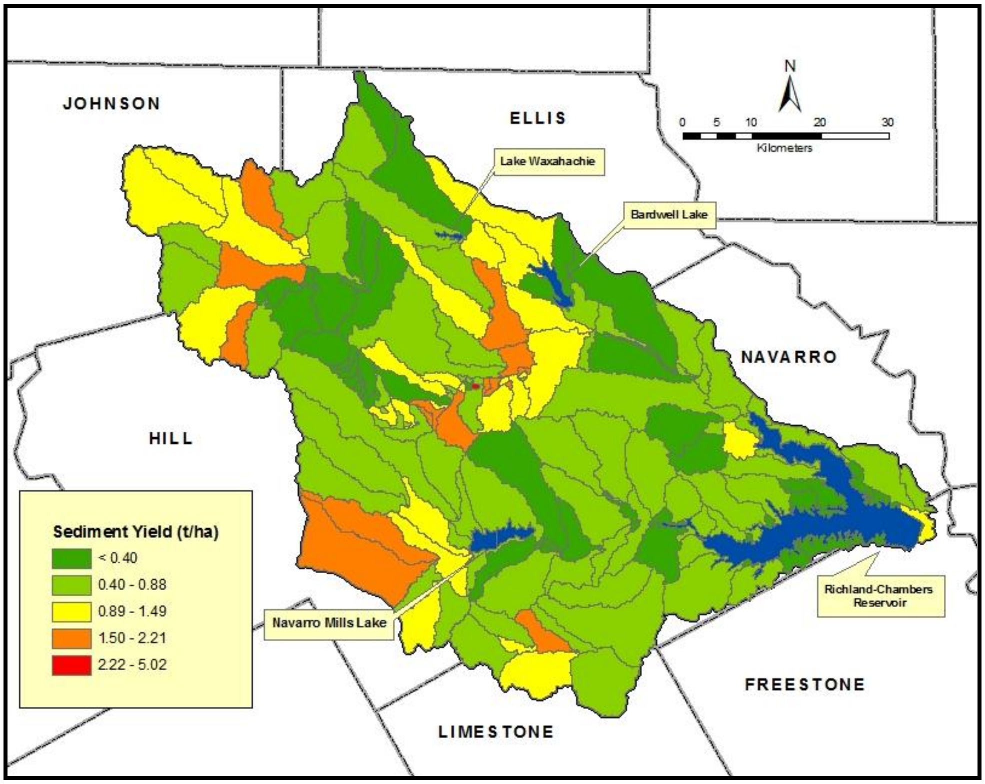


Figure 10.12 Sediment yield distribution as predicted by the Richland-Chambers SWAT model.

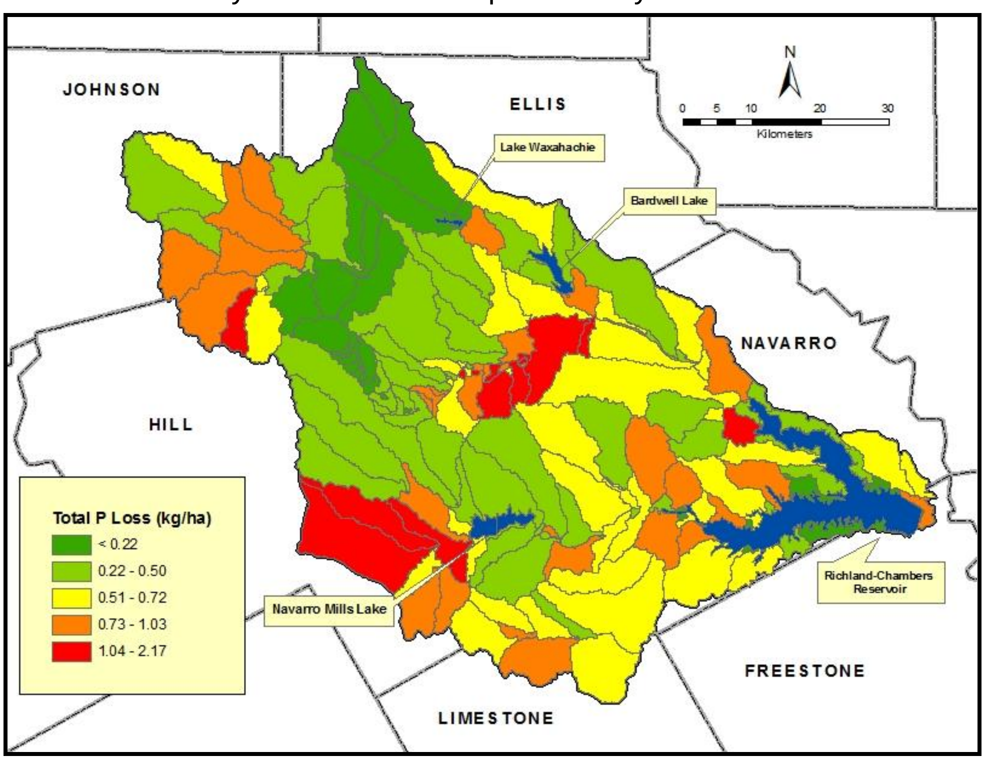


Figure 10.13 Phosphorus yield distribution as predicted by the Richland-Chambers SWAT model.

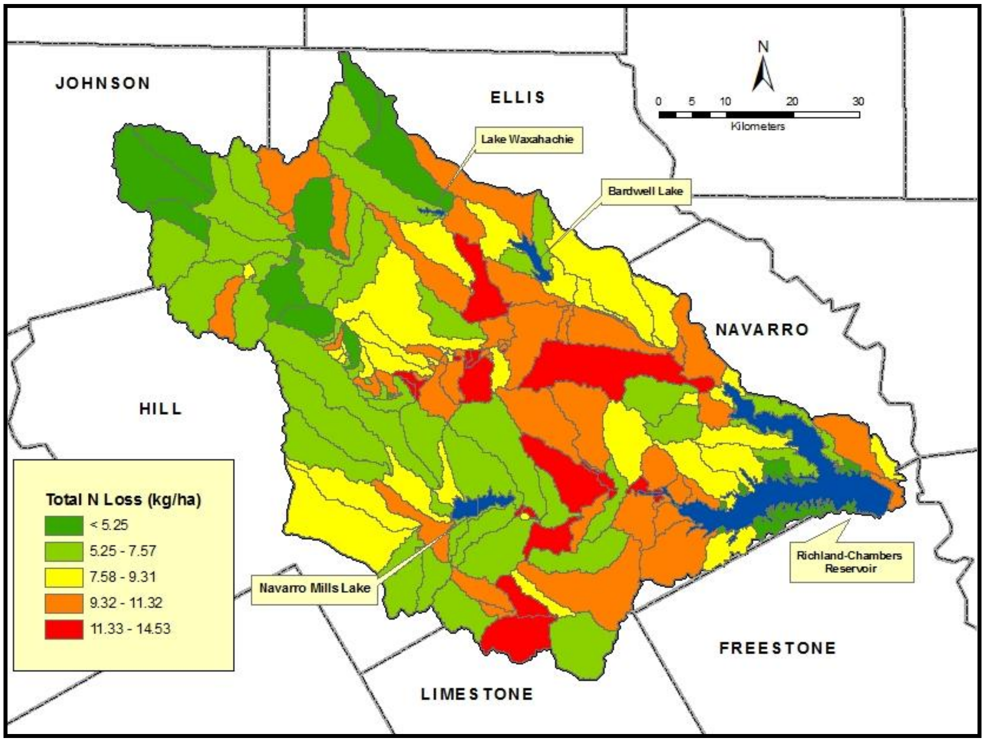


Figure 10.14 Total nitrogen load distribution as predicted by the Richland-Chambers SWAT model.

Scenarios

Conservation Practices

We simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions for each scenario are given in table 10.7.

Ponds

Removing all simulated ponds within the Richland-Chambers Basin revealed that the 304 PL-566 reservoirs reduced sediment by 7.5%.

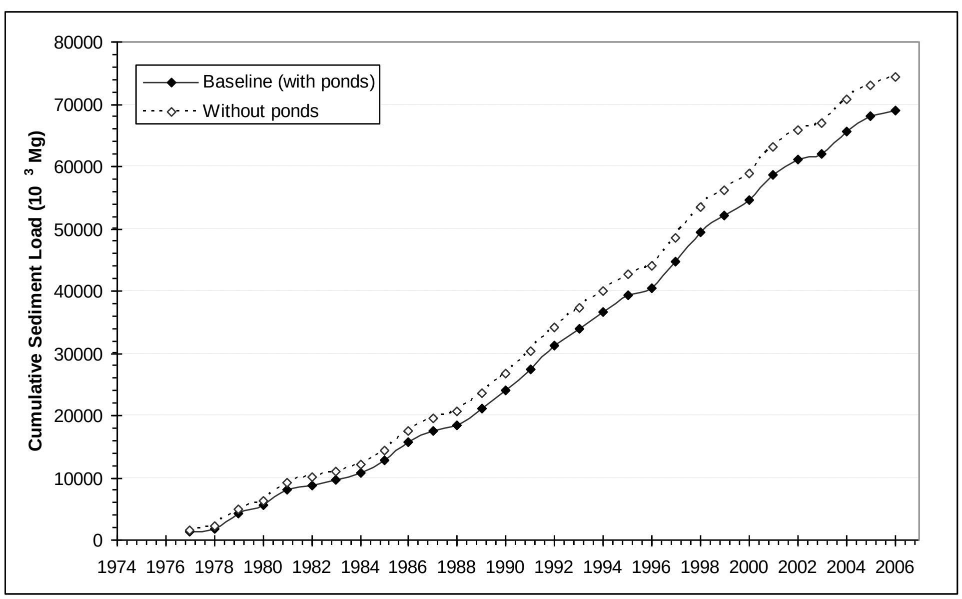


Figure 10.15 Cumulative sediment loads received by the Richland-Chambers Reservoir as predicted by the SWAT model.

Range Utilization

Removing rangeland grazing did not result in considerable erosion reduction (sediment load). However, it did reduce total phosphorus and total nitrogen by 5.7% and 9.1%, respectively.

Point Source Load Elimination

In some cases, we only input flow, sediment and ammonia-nitrogen concentration values from the 20 WWTPs in the basin. During the calibration process, we only used available data as model input. However, to evaluate the impact of point sources, we estimated a constant loading rate for each point source, assuming nutrient concentrations presented in table 10.6, then used this data as SWAT input.

Table 10.6 Assumed nutrient concentrations for WWTPs in the Richland-Chambers model.

Constituent	Organic P	Soluble P	Organic N	Nitrate N	Ammonia N
Assumed concentration (mg/l)	0.38	2.1	2.99	1.68	14.0

Urbanization

In the 2030 urbanization projections, the model predicted a sediment increase of 4% due to urban expansion. In Richland-Chambers Basin, pastureland and cropland dominate the landscape, having profound impacts on nutrient loading as compared to urban areas. Therefore, nutrient loads actually decreased with urbanization due to the proportional decrease in pastureland and cropland.

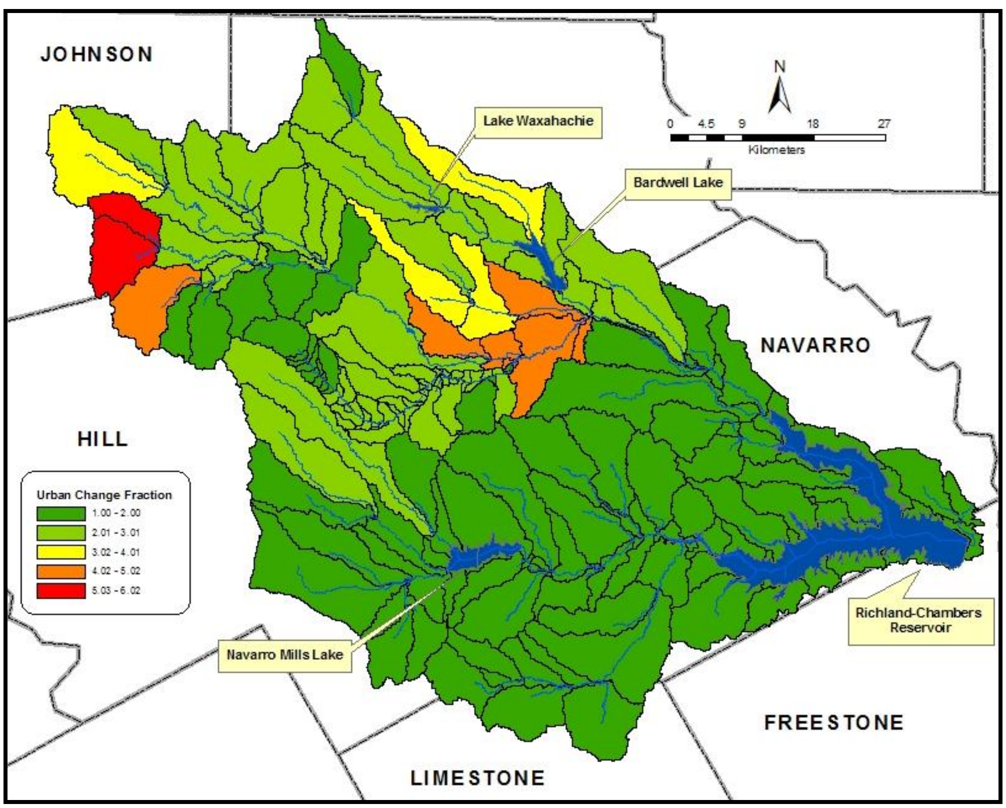


Figure 10.16 Projected change in urban areas within the Richland-Chambers Watershed from 2000 to 2030.

Table 10.7 SWAT-predicted changes in overland sediment and nutrient loads based on different conservation scenarios.

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	2,302,469 metric tons/yr (2,538,038 tons/yr)	285,104 kg/yr (628,547 lbs/yr)	4,011,580 kg/yr (8,844,020 lbs/yr)
No Ponds	7.5%	5.8%	5.3%
No Range Grazing	-1.0%	-5.7%	-9.1%
Urban	3.8%	-1.05%	-2.6%
No Point Sources	-0.35%	-0.25%	-4.0%

Conclusions

SWAT evaluated the impacts of point sources, ponds, rangeland grazing and urbanization on hydrologic and water quality variables within the Richland-Chambers Basin. USGS gauging station 08064100 had long-term streamflow records. We used data from this station to evaluate model-simulated streamflow by calculating monthly and annual NSE statistics, which resulted in values of 0.9 and 0.93, respectively. To calibrate for water quality, we compared total loads from days with available data with simulated values. At gauging site 08064100, model-simulated sediment, organic nitrogen and mineral nitrogen were close to observed values (within 4%) whereas simulated mineral and total phosphorus means were higher because of a large overprediction by the model on few days.

Over 30 years (1977–2006), the model predicted that an average of about 2,302,469 metric tons (2,538,038 tons) of sediment, 285,104 kilograms (628,547 pounds) of phosphorus and 4,011,580 kilograms (8,844,020 pounds) of nitrogen reach the reservoir every year.

Scenario analyses included the removal of ponds, point sources and rangeland grazing. The model also examined the effects of urban expansion predicted for 2030. Existing ponds, which are essentially PL-566 reservoirs, helped by settling about 7.5% of sediment, 5.8% of total phosphorus and 5.3% of total nitrogen generated and transported in the basin. Eliminating rangeland grazing and maintaining range grass reduced sediment, total phosphorus and total nitrogen by 1%, 6% and 9%, respectively. The proportion of urban land in Richland-Chambers Basin is less than pasture, rangeland, brushland, cropland and forest. Therefore, while urbanization increased sediment loads by about 4%, it had relatively small impacts on total nitrogen and total phosphorus losses. The basin does not contain many large cities, so point source loading from the municipal wastewater treatments plants was not considerable.

References

See the appendix.

Chapter 11: Cedar Creek Basin

Introduction

The watershed modeling objective of this project was to use the Soil and Water Assessment Tool (SWAT) to assess the effects of urbanization and other land use changes on sediment and nutrient delivery to the Cedar Creek Reservoir. Constructed in 1964, Cedar Creek Reservoir provides municipal water for Tarrant County (figure 11.1). The total area of the reservoir is about 32,124 acres, and it is the fourth largest lake in Texas. The watershed containing the reservoir, referred to hereafter as Cedar Creek Basin, has a total drainage area of 642,474 acres. The basin has two main channels: Kings Creek and Cedar Creek, as shown in figure 11.1.

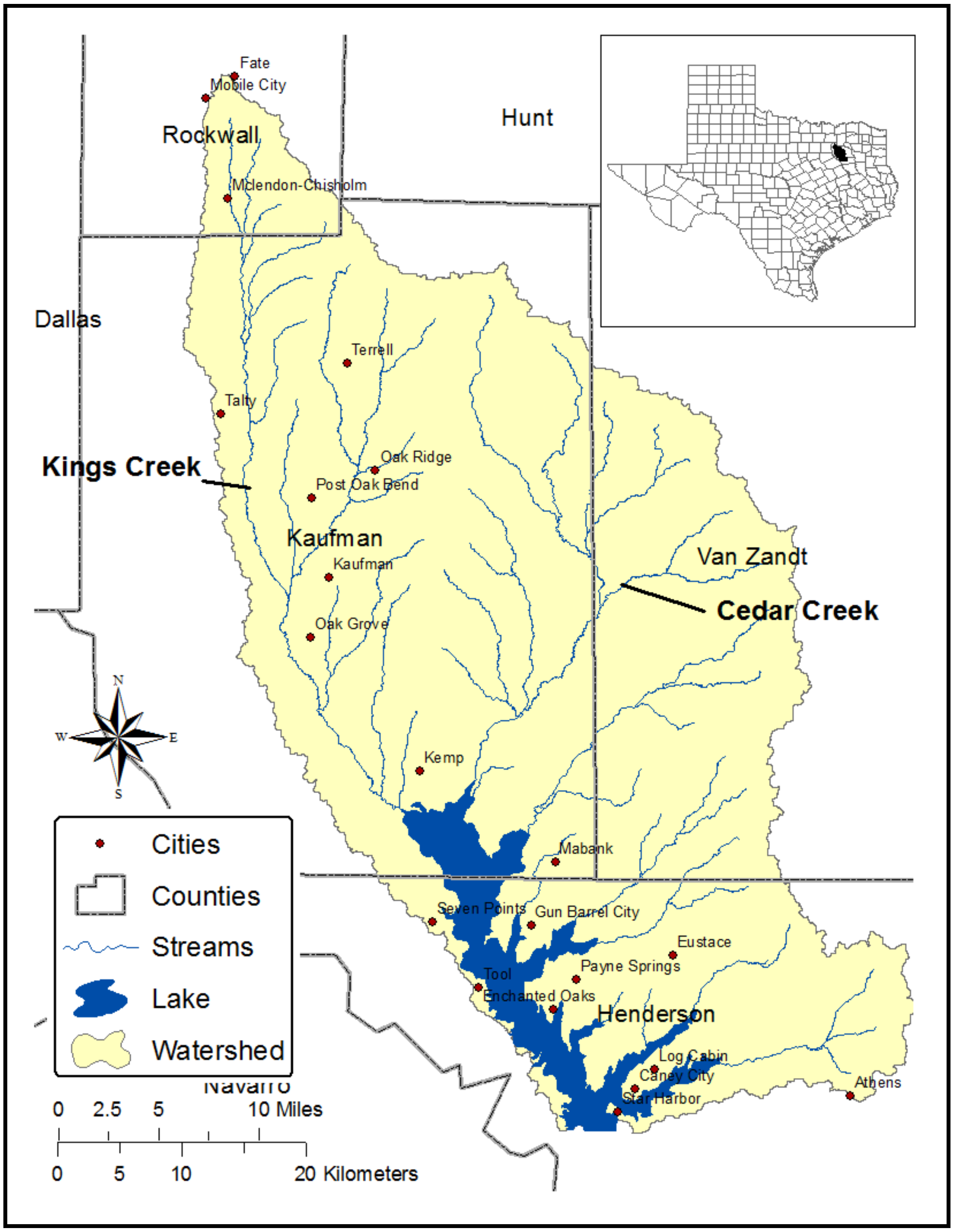


Figure 11.1 Location of Cedar Creek Basin.

Model Input Data Tables and Figures

Topography

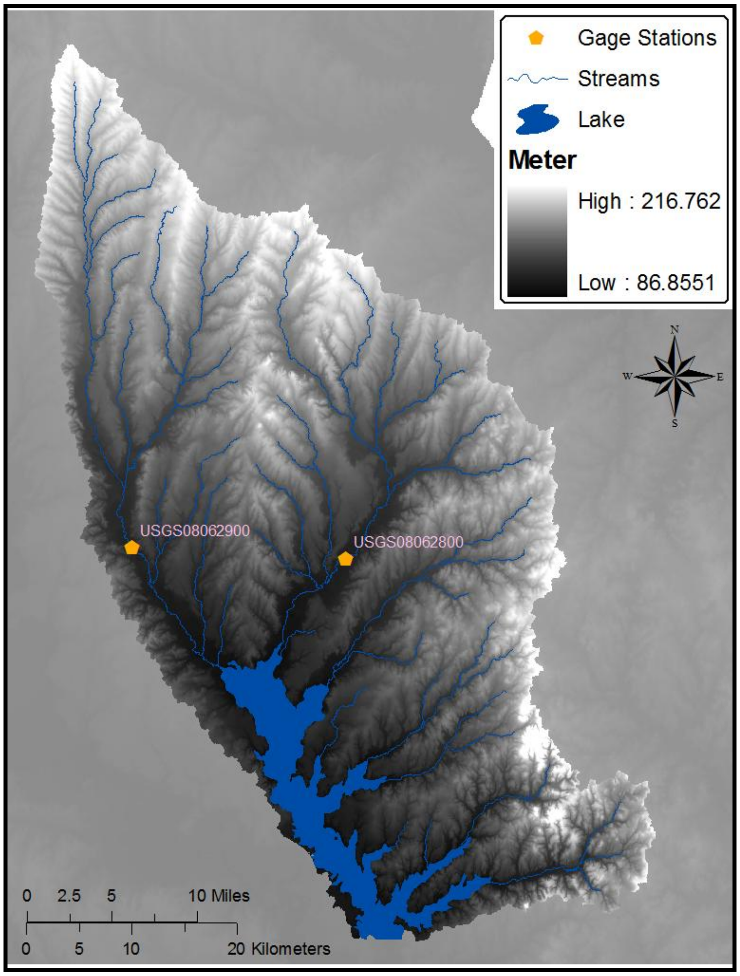


Figure 11.2 Topography of Cedar Creek Basin as defined by a 30-meter (98-foot) DEM.

Soils

We used Soil Survey Geographic (SSURGO) data for the majority of the Trinity River Basin Environmental Restoration Initiative because it is the most detailed soil database available. This 1:24,000-scale soils database is available as printed county soil surveys for over 90% of Texas counties. However, not all mapped counties are available in GIS format (vector or high resolution cell data). In the SSURGO database, each soil delineation (mapping unit) is described as a single soil series. The SSURGO soils data for Hunt, Rockwall, Kaufman and Van Zandt counties are complete, but digitizing of Henderson County is not finished. Therefore, we used a combination of SSURGO and Computer Based Mapping System (CBMS) soils data for simulations of Cedar Creek Basin, as shown in figure 11.3.

The Natural Resources Conservation Service (NRCS) produces the CBMS, also known as Map Information Assembly Display System (MIADS) (Nichols, 1975). It is a grid cell digital map created from 1:24,000 scale soil sheets with a cell resolution of 820 feet. The CBMS database differs from some grid GIS databases in that attributes of each cell are determined by the soil under the center point of the cell instead of soil comprising the largest percentage of the cell.

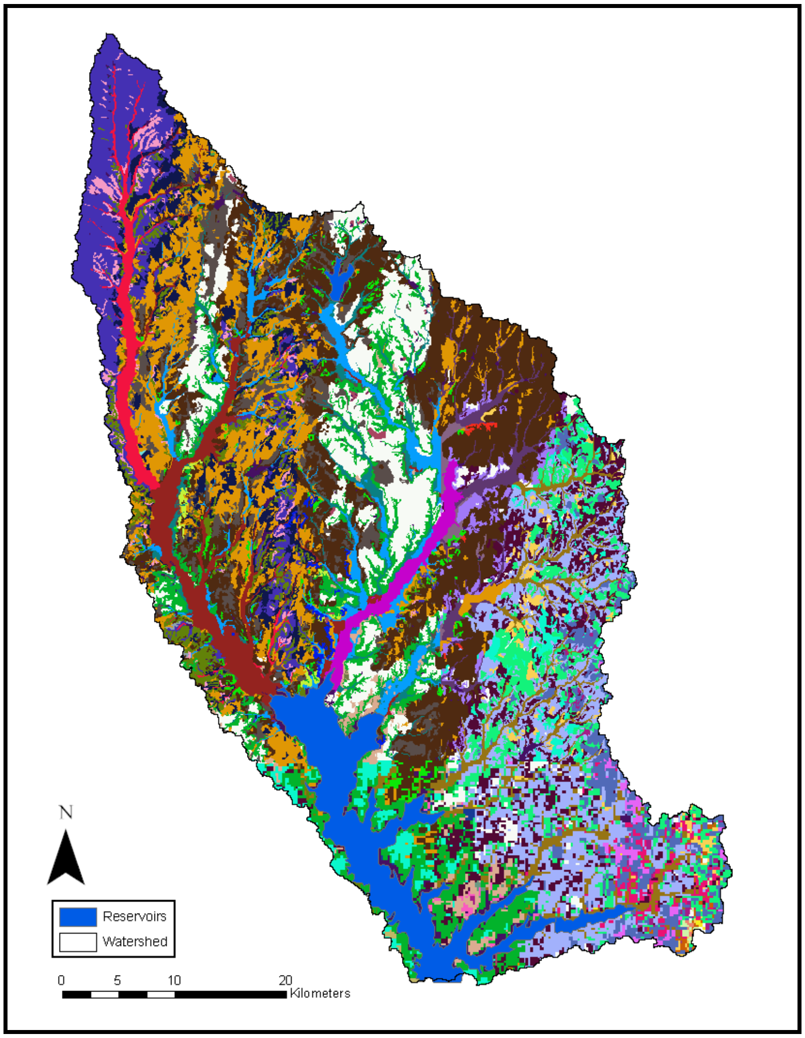


Figure 11.3 The soil distribution of Cedar Creek Basin as defined by two soil databases, SSURGO and CBMS.

Weather

Precipitation data from nearby stations (shown in figure 11.4) substituted for any missing data in each station's record, and SWAT generated missing temperature data.

During flow calibration, we noted that predicted flow was much higher than measured flow from 1999 through 2002, but the remainder of the simulation (1980–1998) matched well. Five climate stations had no data from 1999–2002, so data from nearby stations were used to represent the missing data. We suspected this as the cause of model over-prediction. To correct the problem, we used NEXRAD data to "enhance" the climate stations' missing data from 1999 to 2002. This was done by averaging NEXRAD grid data for all subbasins near an individual climate station and using those values. This enhancement resulted in a much better match between simulated and measured flow.

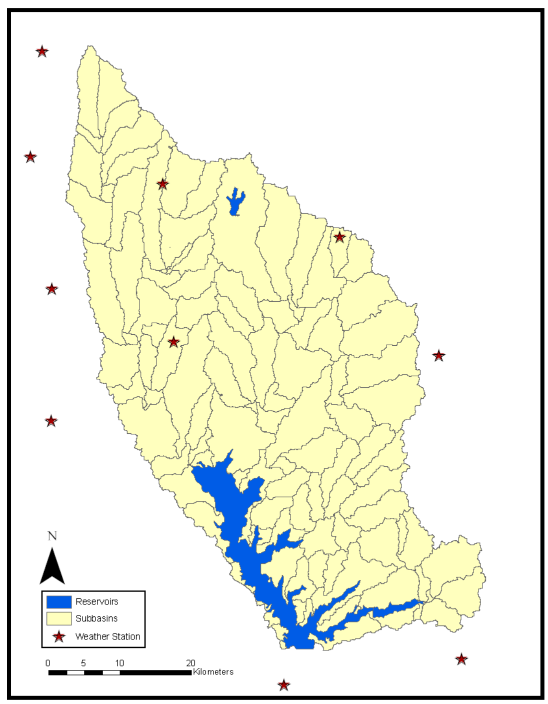


Figure 11.4 The location of National Weather Service stations that provided temperature and precipitation data for the Cedar Creek SWAT model from 1950–2002.

Land Use

In addition to National Land Cover data, the Texas A&M Spatial Sciences Lab (SSL) developed a land use/land cover map (30-meter) from 2001 Landsat 7 data using ground control points collected by Tarrant Regional Water District (TRWD) (figure 11.5Figure). Due to rapid urban development in the basin, urban land use categories from the SSL 2001 map were superimposed onto the 1992 NLCD map to provide a more current representation of urban areas in this study.

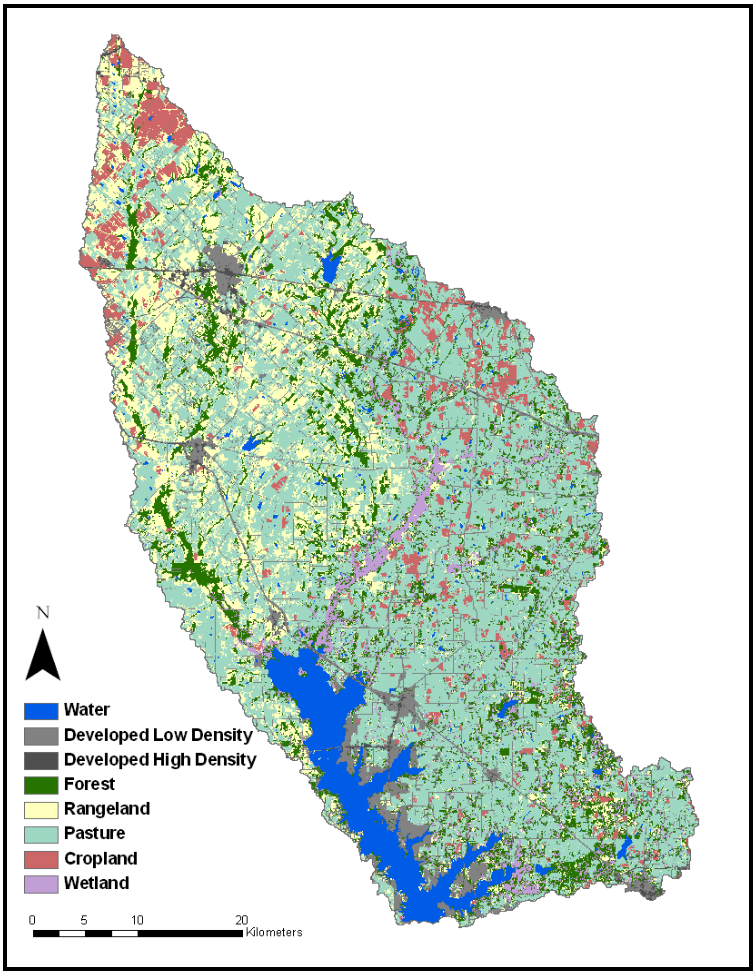


Figure 11.5 Land cover data (2001 NLCD) enhanced for urban area.

Point Sources

Wastewater treatment plant loading was based on one year of weekly nutrient and flow data voluntarily collected and measured by the WWTPs themselves. We combined the weekly data into monthly loadings for each WWTP and routed them through the creeks. Cedar Creek Basin contains nine wastewater treatment plants (WWTPs) distributed across the basin, and two of these WWTPs discharge directly into the Cedar

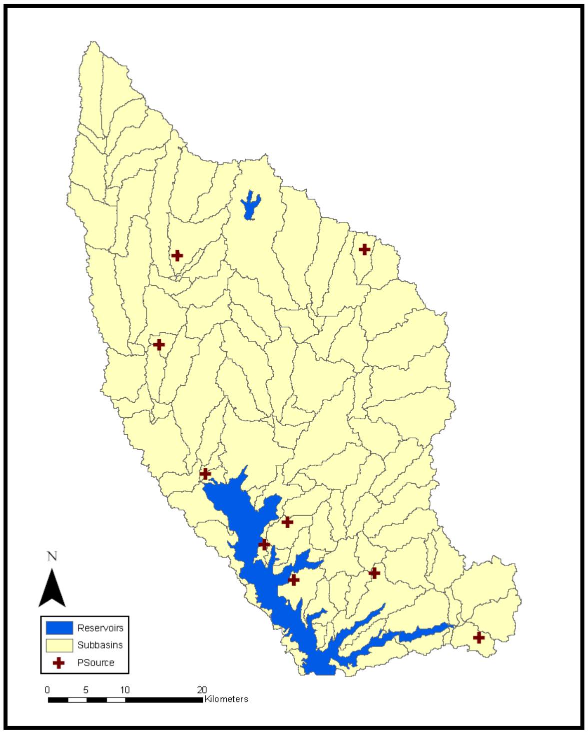


Figure 11.6 The location of wastewater treatment plants used in Cedar Creek SWAT model.

Creek Reservoir (figure 11.6Figure). Point source input data are given in Appendix table A-1.

Monitoring Stations

Unlike most of the chapters that use USGS gauging stations, here water quality data were provided by 24-hour samples collected at various points along Kings Creek, downstream of a major wastewater treatment plant (City of Terrell). To set up the baseline model for calibration and validation, we collected the following input data: concentration of dissolved oxygen, biological oxygen demand, ammonia, phosphorus, Chlorophyll a, organic nitrogen and nitrate-nitrite concentrations. Furthermore, we set up an independent QUAL-2E model based on measured channel geometry and hydraulics developed during the dye study. The calibrated QUAL-2E kinetic terms and coefficients were used as initial estimates to set up SWAT's instream water quality parameters. In addition, TRWD periodically collected water quality grab samples from 1989 to 2002 at ten monitoring stations (figure 11.7). These were used to modify and calibrate instream SWAT model parameters.

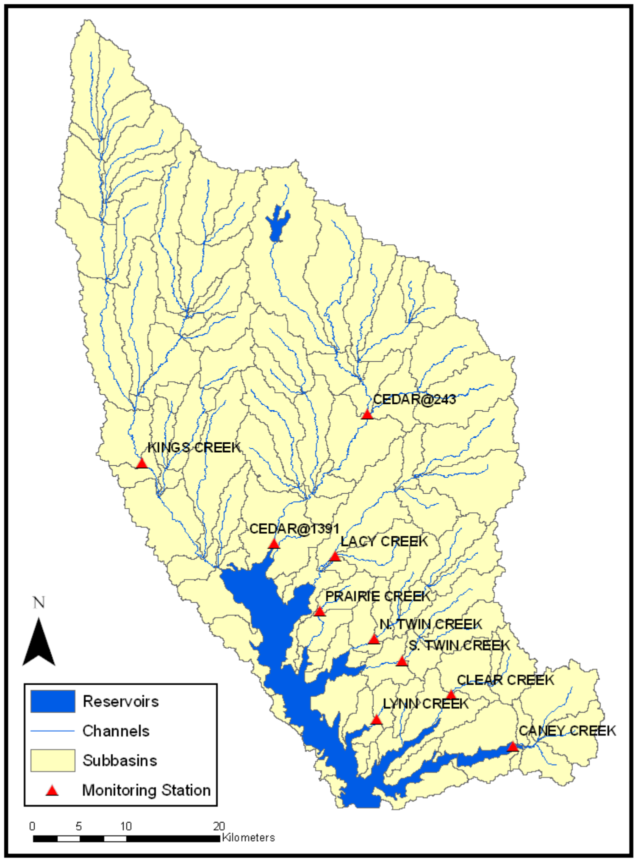


Figure 11.7 Monitoring stations that provided nutrient data for model calibration.

Ponds

The Cedar Creek Basin contains about 120 inventory-sized dams (as defined by the Texas Commission on Environmental Quality). These include NRCS flood prevention dams, farm ponds and other privately owned dams. We used physical data (e.g., surface area, storage, drainage area and discharge rates) for these dams as SWAT inputs to allow routing of runoff through the structures. Four impoundments were big enough to be simulated as reservoirs while the rest were simulated as small ponds (figure 11.8).

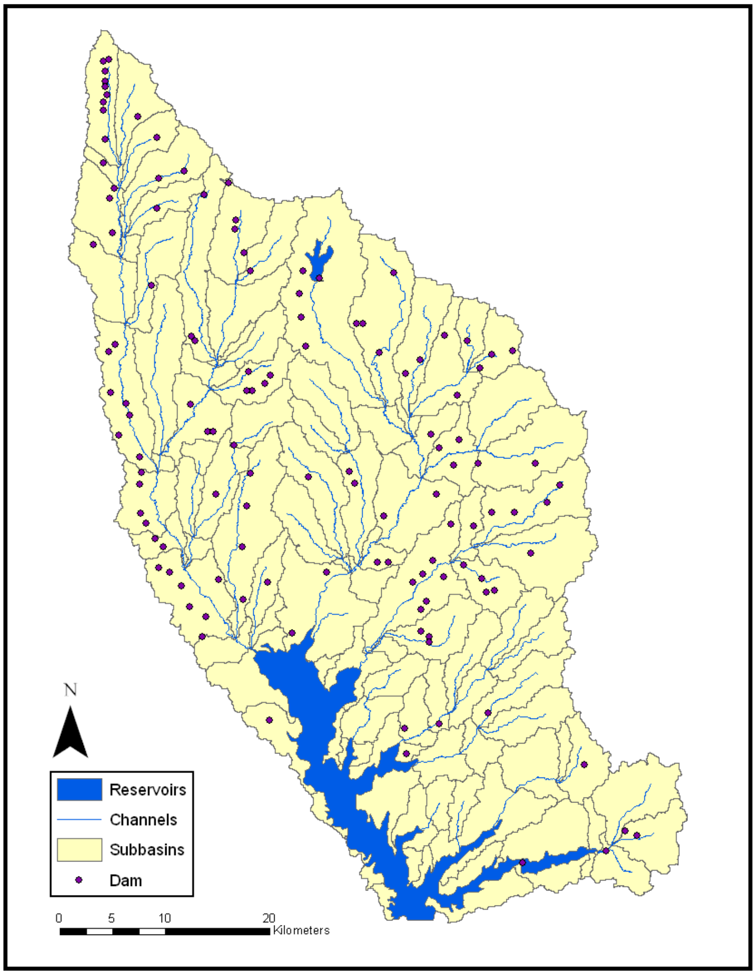


Figure 11.8 Distribution of NRCS flood prevention dams and other dams in Cedar Creek.

Table 11.1 The National Inventory of Dams provided reservoir characteristics for Cedar Creek Reservoir.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Cedar Creek	102	25,850	63.7	73,248	180.5	Measured

Subbasin Delineation

BASINS 3.0/AVSWAT automatically delineated subbasins within Cedar Creek Basin using a stream definition threshold of 1,236 acres. Additional subbasin outlets were inserted at USGS stream gauge stations, TRWD tributary sampling points, municipal wastewater discharge points, WASP model input locations (Cedar Creek Reservoir boundaries) and at four of the larger lakes within the basin (Terrell City Lake, Lake Kaufman, Forest Grove Dam and Valley View Lake). The resulting map contained 106 subbasins (figure 11.6Figure).

Because Cedar Creek Reservoir partially submerged several subbasins, it was not included as a reservoir in SWAT simulations. Instead, we simulated the land cover for these submerged areas as "WATER". We accounted for the effects of submergence in main channel inputs (channel erodibility and channel cover were set to "0.0") and turned off QUAL2E in SWAT for affected subbasins.

HRU distribution

SWAT's input interface divided each subbasin into HRU's with unique soil and land use combinations. SWAT determined the number of HRU's within a subbasin by: (1) creating an HRU for each land use that equaled or exceeded two percent of the subbasin area, and (2) creating an HRU for each soil type that equaled or exceeded 10 percent of any land uses selected in (1). Using these inputs, the interface created 1,516 HRUs within the basin. The thresholds used in this study are slightly different than those listed in the methodology.

Management

Based on data derived from NRCS field office personnel, we assumed that farmers grew grain sorghum on all cropland and employed no conservation practices (Universal Soil Loss Equation (USLE) "P" = 1.0). We assumed that farmers applied fertilizer to cropland at a rate of 60.1 pounds of nitrogen and 30.3 pounds of phosphorus per acre and used conventional tillage.

Pasture

According to Homer Sanchez, an NRCS State Range Conservationist, the basin's pastureland was in fair hydrologic condition. Based on conversations with county extension agents, we simulated two hay cuttings per year with annual fertilization on 50% of crop fields. SWAT applied fertilizer to pasture at a rate of 60 pounds of nitrogen per acre.

Urban

SWAT simulated urban pervious surfaces with Bermuda grass and applied fertilizer automatically, with rates and amounts based on a nitrogen stress level of 0.9.

Model Calibration and Validation

Flow

The available period of record for streamflow at two USGS gauging stations within the basin determined the calibration period (1963–1987). To determine the appropriate fraction of baseflow and surface flow at selected gauging stations, we used a baseflow filter program (Arnold et al., 1995a).

For each model simulation, we input appropriate plant growth parameters for brush, native grasses and other land covers. Initial inputs were based on known or estimated watershed characteristics. To calibrate for flow, we adjusted appropriate inputs affecting surface runoff and baseflow including the runoff curve number, soil evaporation compensation factor, shallow aquifer storage, shallow aquifer re-evaporation and channel transmission loss. Parameters were adjusted until the simulated total flow and baseflow fraction were approximately equal to the measured total flow and baseflow, respectively.

We validated the model by comparing simulated flow to calculated inflow at the Cedar Creek Reservoir. TRWD's mass balance of the Cedar Creek Reservoir (1980–2002) provided data for inflow calculations (figures 11.9 and 11.10). The analysis was performed using measured daily reservoir volume, water surface evaporation, withdrawals, discharges and rainfall. Again, NEXRAD provided data used to "enhance" the climate stations' missing data from 1999–2002.

Mean, standard deviation, coefficient of determination (R^2) and Nash-Sutcliffe modeling efficiency (NSE) (Nash and Sutcliffe, 1970) were used to evaluate model predicted streamflow during calibration and validation. Statistical evaluations for this study were based on a comparison between mean observed data and mean simulated flow as well as sediment and nutrient loadings for days with available grab sample data.

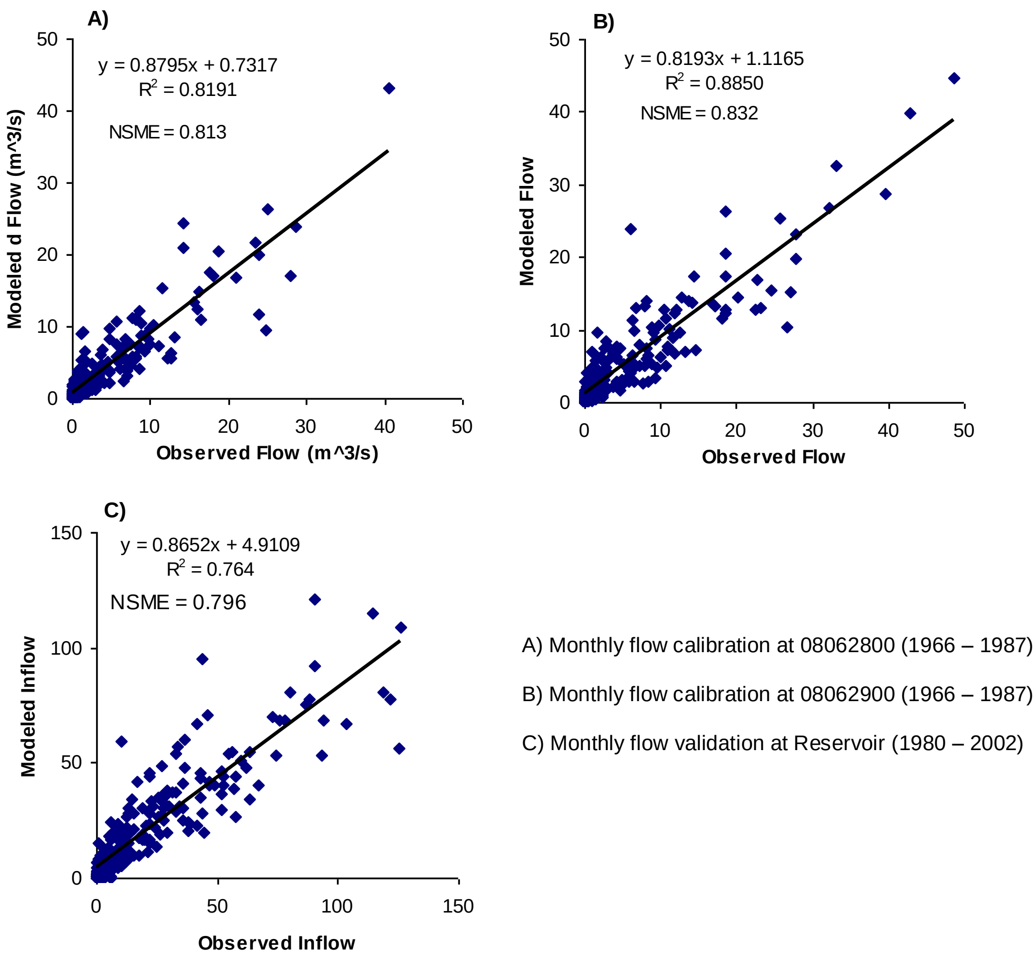


Figure 11.9 Flow calibration and validation at two USGS gauging stations and reservoir inflow.

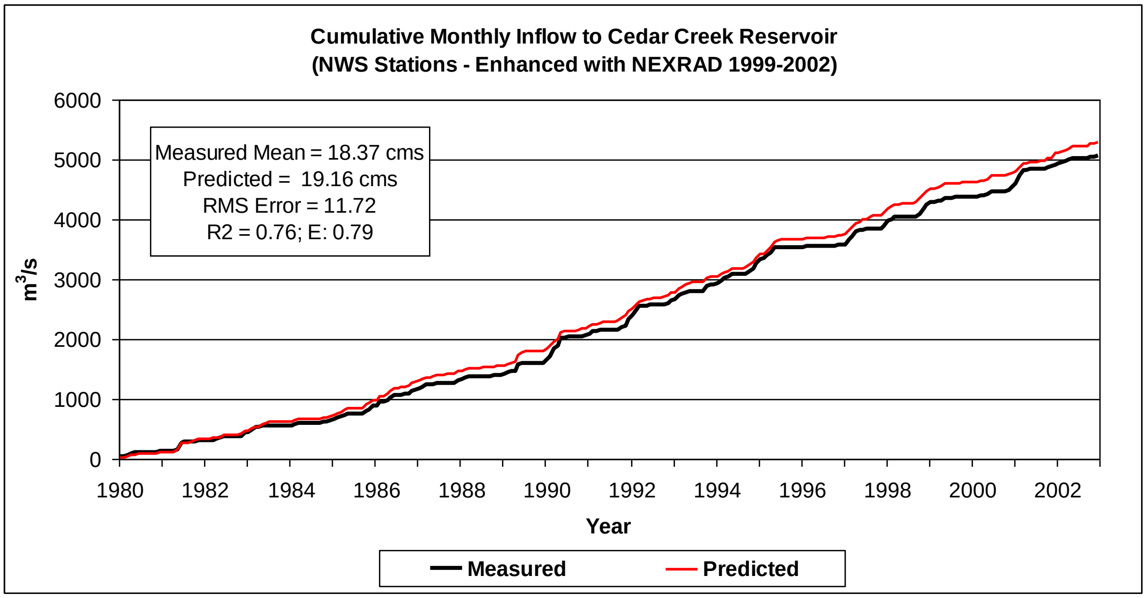


Figure 11.10 Cumulative monthly reservoir inflow.

Sediment

The Texas Water Development Board performed hydrographic surveys of Cedar Creek Reservoir in February 1995 (TWDB, 1995) and May 2005 (TWDB, 2006). We compared the measurements from these two surveys with the original reservoir design information to determine the volume of sediment deposited from 1966 (year of dam completion) to 1994 and 2005. The sedimentation rate based on just the 1995 and 2005 data is about 32.5 acre-feet per year.

In early 2006, Baylor University undertook a lake sediment survey, collecting sediment cores to estimate average density and thickness of sediment at the lake bottom (Allen et al., 2006). In addition, Allen et al. (2006) conducted a watershed survey to identify stream segments with channel erosion problems and to quantify channel erosion using NRCS field assessment techniques, such as RAP-M.

In contrast to the amount estimated by the 1995 survey, the original design volume indicated a sedimentation rate of 1,032 acre-feet per year. This rate was also consistent with a sediment thickness of 1.2 to 1.5 feet observed in cores from Baylor's study. Hence, we did not use the 1995 lake survey data for model calibration.

The average dry-weight density of the post-impoundment sediment was about 21.5 pounds per cubic foot. Based on the lake sediment survey and the watershed survey, the erosion rate within the Cedar Creek Basin is estimated at about 446,558 metric tons (492,246 tons) per year. Of the overall erosion rate, channel erosion contributions about 152,572 metric tons (168,182 tons) per year (34%). The rest of the sediment (293,986 metric tons (324,064 tons) per year) comes from overland erosion (Allen et al., 2006).

We compared SWAT-simulated sediment to measured sediment for a 37-year period from 1966 to 2002. Appropriate input parameters were adjusted until the predicted annual sediment load from overland and channel erosion were approximately equal to measured data (figure 11.11). Final values for SWAT input coefficients used in flow and sediment calibration are given in table 11.2.

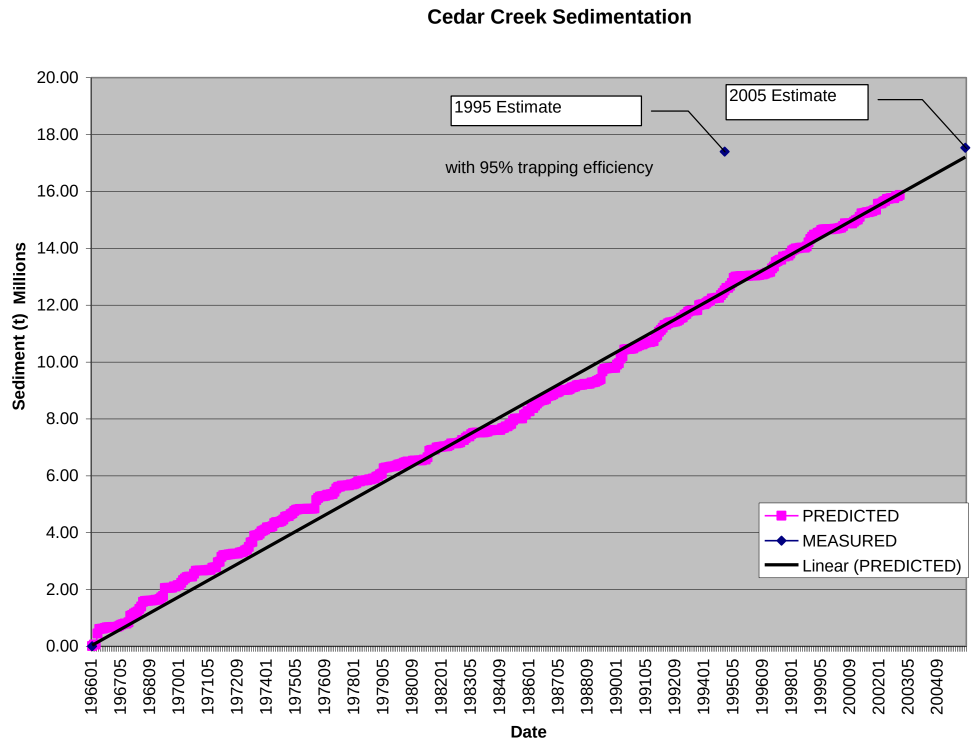


Figure 11.11 Measured and predicted sediment accumulation in Cedar Creek Reservoir from 1966 through 1994.

Table 11.2 SWAT input coefficients adjusted for calibration of flow and sediment.

Component	Parameter (file)	Description	Input Value
Flow	CN2 (*.mgt)	SCS runoff curve number (adjustment range)	+3 to -3
	ESCO (*.hru)	Soil evaporation factor	0.85
	GW_REVAP (*.gw)	groundwater re-evaporation coefficient	0.1
	GW_DELAY (*.gw)	Groundwater delay time (days)	135
	GWQMN (*.gw)	Groundwater storage required for return flow (mm)	1.00
	REVAPMN (*gw)	Groundwater storage required for revap (mm)	1.6000
	ALPHA_BF (*gw)	Baseflow alpha factor (days^{-1})	0.0420 to 0.2006
	CH_N2 (*.rte)	Mannings "n" roughness for channel flow	0.075
	CH_K2 (*.rte)	Hydraulic conductivity of channel alluvium (mm/hr)	0.1 to 4.0
Sediment	RSDIN (*.hru)	Initial soil residue cover (kg/ha)	1000
	USLE_C (crop.dat)	Minimum "C" value for pastureland in fair condition	0.007
	SPCON (basins.bsn)	Linear parameter for calculating the maximum amount of sediment that can be reentrained during channel sediment routing	0.01
	SPEXP (basins.bsn)	Exponent parameter for calculating sediment reentrained in channel sediment routing	1.4
	CH_COV (*.rte)	Channel cover factor	0.1 to 1.0
	CH_EROD (*.rte)	Channel erodibility factor	0.3 to 0.8

Nutrients

To calibrate SWAT for nutrients, we first selected subbasins in the Kings Creek Basin that correlated with the Qual2E reaches set up by Espey Consultants, Inc. Final coefficients from the Espey model provided a starting point for SWAT calibration. WWTP data measured during the TRWD King's Creek study were used as point-source loads.

We calibrated SWAT for King's Creek by comparing daily output on September 16, 2002 with data from TRWD's King's Creek study measured on September 17 and 18, 2002, in selected subbasins of the King's Creek tributary. The 16th was chosen for comparison of SWAT output because SWAT predicted rainfall and runoff on the 17th, 18th and 19th even though no rainfall actually occurred in the Kings Creek tributary during that time. The model error was due to rainfall variability not reflected in measured rainfall data used by SWAT. The main goal was to compare measured and predicted values for low flow periods (no runoff = municipal wastewater discharge only), and this was accomplished by using SWAT output from the 16th.

The next step was to use SWAT defaults as a starting point in calibrating the remainder of the subbasins for the simulation period (1989–2002). One year's worth of weekly self-reporting data collected by TRWD in 2001 and 2002 defined WWTP loads. The following rules were used to generate the data:

Data Issues

1. We used 12 months of data for the analysis, but if more than 12 months of data were reported, a subset of 12 months was used instead.
2. If a value under the detection limit was reported, it was rounded up to the detection limit.
3. If a value was reported as non-detectable with no reported detection limit, the value was estimated to be approximately 0.01 mg/L less than the minimum value in the dataset.
4. If an NH_3 value was not reported, it was assigned the same value as the individual plant's permit limit, if applicable.
5. If flow was not reported one week, it was estimated as the average of the preceding and following week's flows.
6. If multiple flows were missing for any one-month period, we used the average flow reported in the DMR for that month.
7. There is no weekly data available for the Athens WWTP. The DMR average flows for the Athens WWTP and weekly concentration data from the Kaufman WWTP were used to estimate the plant's monthly loads.
8. According to the Terrell WWTP operator, the Terrell dataset lacks reliable data for a 20-week period (11/14/01–3/27/02). This data was not used for analysis. Instead, we calculated an average value for each parameter using the remaining data. This average value was used each week during the 20-week period of bad data. However, all reported flows were used.
9. If a calculated organic nitrogen or organic phosphorus value resulted in a negative value, this value was estimated as zero.

We compared this simulation to water quality data collected by TRWD (1989–2000) in each major tributary (Kings, Cedar, Lacy, North Twin, South Twin, Lynn, Clear, Caney and Prairie) (figure 11.7). To account for daily variability in SWAT, simulated output was averaged for the three day period surrounding the measured grab sample. Median, 25th percentile and 75th percentile of the 3-day averages were compared to the medians, 25th and 75th percentiles of the measured grab samples (figures 11.12–11.14). The coefficients for all subbasins, except those correlated with the TRWD study of Kings Creek Basin, were adjusted to match observed data (table 11.3). We made only one change to subbasins of the Kings Creek study: point-source loads were generated from a year's worth of weekly WWTP data rather than the one day of data collected by TRWD during the 2002 study.

Table 11.3 General water quality input coefficients (.wwq) for calibration of QUAL2E and SWAT for the King's Creek study subbasins (2002) and the 1989–2002 tributary study.

Variable Name	Definition	QUAL2E-Espey	SWAT-SSL	SWAT	SWAT
		Cal. Coef	Cal. Coef.	Default	Range
LAO	Light averaging option	2	2	2	2
IGROPT	Algal specific growth rate option	2	2	2	3 options
AI0	Ratio of chlorophyll-a to algal biomass [µg-chla/mg algae]	10	10	50	10–100
AI1	Fraction of algal biomass that is nitrogen [mg N/mg alg]	0.090	0.090	0.080	0.07–0.09
AI2	Fraction of algal biomass that is phosphorus [mg P/mg alg]	0.020	0.020	0.015	0.01–0.02
AI3	The rate of oxygen production per unit of algal photosynthesis [mg O_2/mg alg]	1.600	1.400	1.600	1.4–1.8
AI4	The rate of oxygen uptake per unit of algal respiration [mg O_2/mg alg]	2.300	2.000	2.000	1.6–2.3
AI5	The rate of oxygen uptake per unit of NH_3-N oxidation [mg O_2/mg NH_3-N]	3.500	3.000	3.500	3.0–4.0
AI6	The rate of oxygen uptake per unit of NO_2-N oxidation [mg O_2/mg NO_2-N]	1.000	1.000	1.070	1.0–1.14
MUMAX	Maximum specific algal growth rate at 20ºC [day^{-1}]	1.800	1.000	2.000	1.0–3.0
RHOQ	Algal respiration rate at 20ºC [day^{-1}]	0.100	0.300	0.300	0.05–0.50
TFACT	Fraction of solar radiation computed in the temperature heat balance that is photosynthetically active	0.300	0.300	0.300	0.01–1.0
K_L	Half-saturation coefficient for light [kJ/(m2·min)]	0.418	0.418	0.750	0.2227–1.135
K_N	Michaelis-Menton half-saturation constant for nitrogen [mg N/lL]	0.400	0.400	0.020	0.01–0.30
K_P	Michaelis-Menton half-saturation constant for phosphorus [mg P/l]	0.040	0.040	0.025	0.001–0.05
LAMBDA0	Non-algal portion of the light extinction coefficient [m-1]	1.500	1.500	1.000	-
LAMBDA1	Linear algal self-shading coefficient [m-1·(µg chla/l)-1)]	0.002	0.002	0.030	0.0065–0.065
LAMBDA2	Nonlinear algal self-shading coefficient [m-1·(µg chla/l)-2]	0.054	0.054	0.054	0.054
P_N	Algal preference factor for ammonia	0.100	0.100	0.500	0.01–1.0

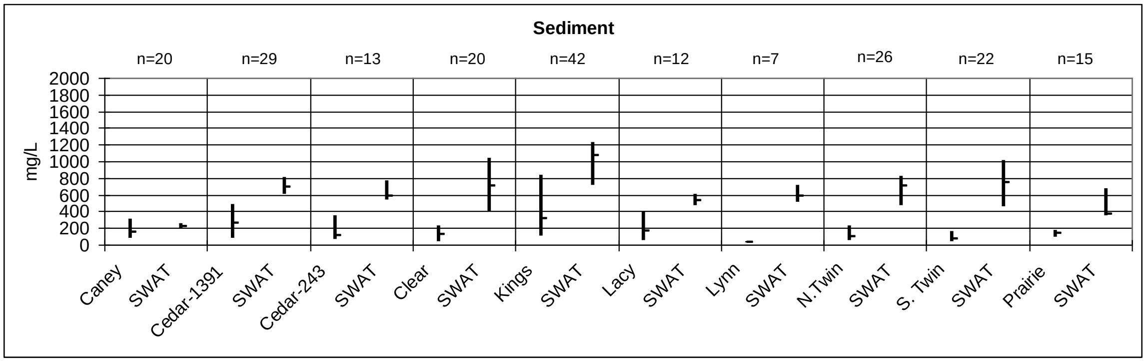


Figure 11.12 Median, 25th and 75th percentile of the measured and predicted sediment (TSS), provided by the 1989 to 2002 tributary study.

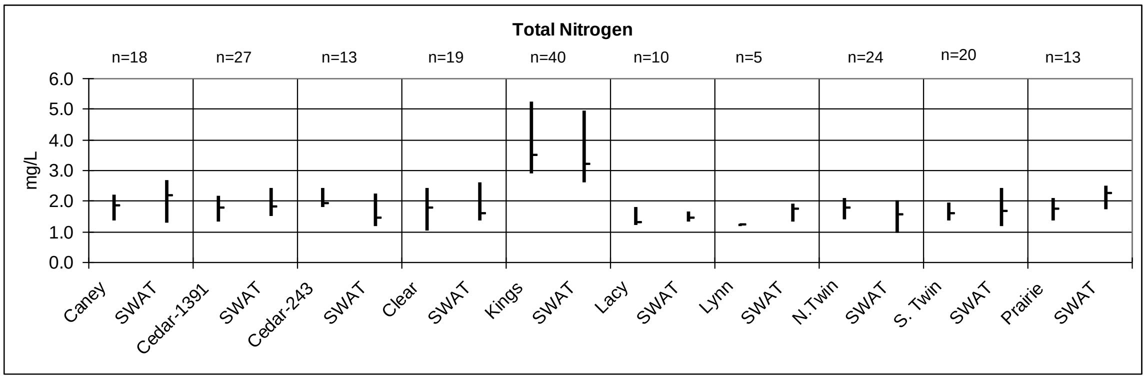


Figure 11.13 Median, 25th and 75th percentile of measured and predicted total nitrogen, provided by the 1989 to 2002 tributary study.

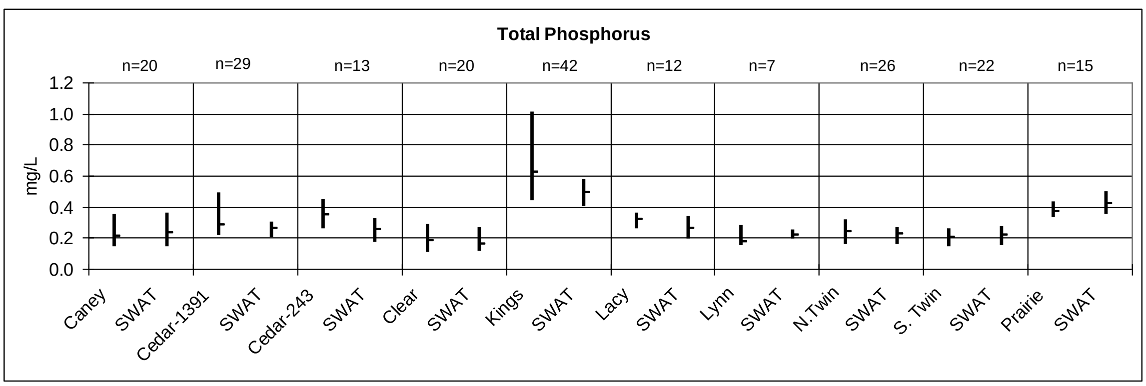


Figure 11.14 Median, 25th and 75th percentile of measured and predicted total phosphorus, provided by the 1989 to 2002 tributary study.

Table 11.4 Estimated sediment and nutrients discharged into Cedar Creek Reservoir.

	Sediment in metric tons/yr (tons/yr)	**Total N in kg/yr (lb/yr)**	**Total P in kg/yr (lb/yr)**
Calibrated model estimation (baseline)	450,000 (496,040)	1,419,380 (416,819)	188,670 (416,819)

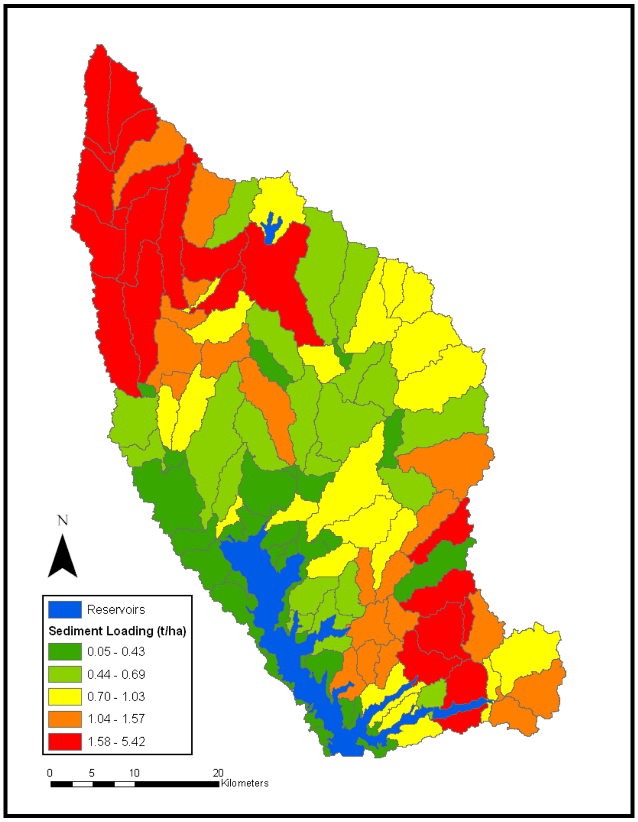


Figure 11.15 SWAT-predicted sediment losses produced by overland flow in Cedar Creek Basin.

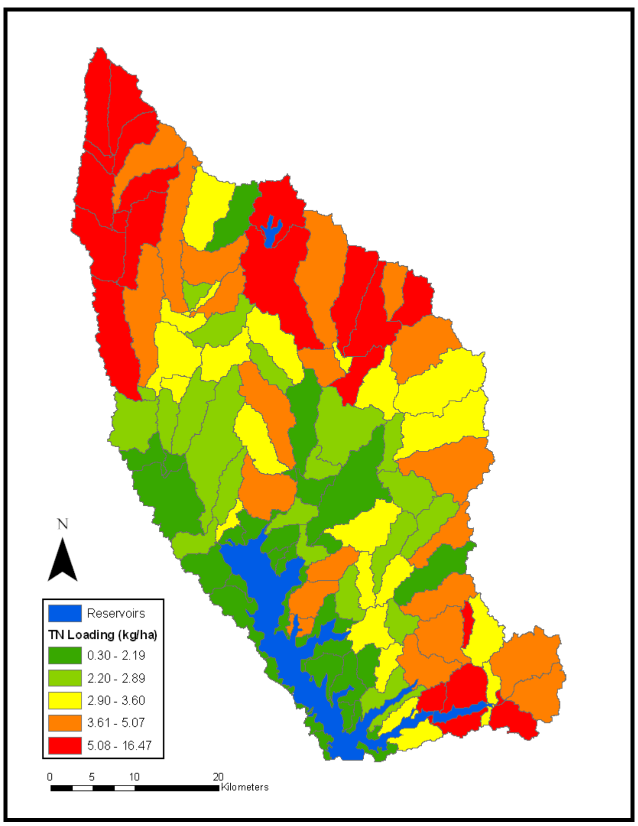


Figure 21.16 SWAT-predicted total nitrogen losses due to overland flow in Cedar Creek Basin.

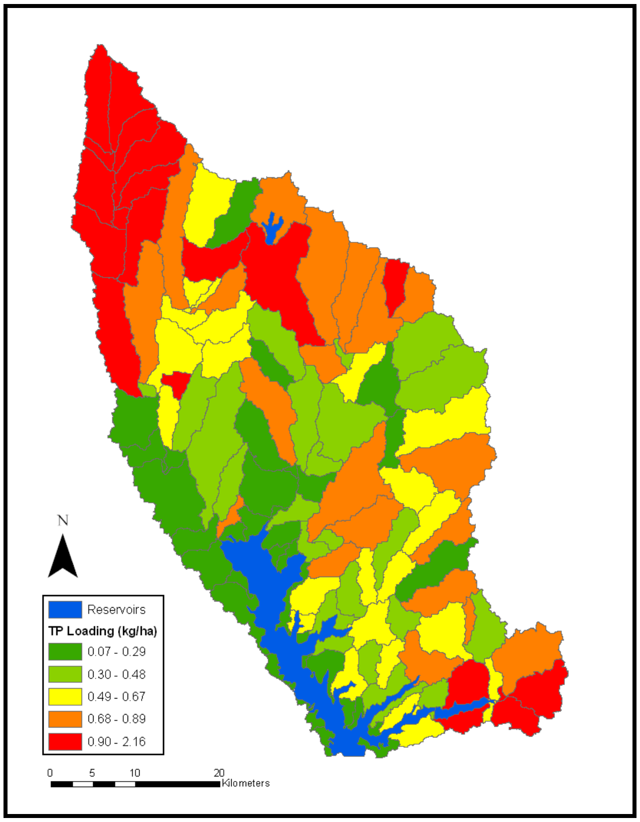


Figure 11.17 SWAT-predicted total phosphorous losses due to overland flow in Cedar Creek Basin.

Scenarios

Conservation Practices

We simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions or increases are shown in a table below each scenario.

Ponds

The 120 inventory-sized NRCS floodwater dams and private ponds simulated by SWAT reduced sediment, total nitrogen and total phosphorus loading of the reservoir by 16%, 6.8% and 6.5%, respectively (table 11.5).

Table 11.5 Impacts of pond removal on sediment and nutrient loading.

	Sediment in metric tons/yr (tons/yr)	**Total N in kg/yr (lb/yr)**	**Total P (lb/yr)**
With ponds (baseline)	450,000 (496,040)	1,422,362 (3,135,771)	189,066 (416,819)
Without ponds	522,200 (575,627)	1,519,776 (3,350,533)	201,402 (444,015)
Difference	+16%	+6.8%	+6.5%

Range Utilization

We removed grazing from the baseline simulation by eliminating all agricultural operations on rangelands. Reservoir loading with sediment, total nitrogen and total phosphorus were reduced by 0.3%, 0.7% and 0.02%, respectively.

Table 11.6 The impacts of grazing removal on sediment and nutrient loading.

	Sediment in metric tons/yr (tons/yr)	Total N in kg/yr (lb/yr)	Total P in kg/yr (lb/yr)
With Grazing (baseline)	450,000 (496,040)	1,422,362 (3,135,771)	189,066 (416,819)
Without Grazing	448,800 (494,717)	1,413,103 (3,115,359)	189,036 (416,753)
Difference	-0.3%	-0.7%	-0.02%

Point Source Load Elimination

By making the discharge of all WWTPs zero, we eliminated nine WWTPs from the model. Reservoir loading with sediment, total nitrogen and total phosphorus was reduced by 0.2%, 3.8% and 6.0%, respectively.

Table 11.7 The impacts of eliminating WWTP discharge (point sources) on sediment and nutrient loading.

	Sediment in metric tons (tons/yr)	Total N in kg/yr (lb/yr)	Total P in kg/yr (lb/yr)
With WWTPs (baseline)	450,000 (496,040)	1,422,362 (3,135,771)	189,066 (416,819)
Without WWTPs	449,000 (494,938)	1,367,928 (3,015,765)	177,722 (391,810)
Difference	-0.2%	-3.8%	-6.0%

Urbanization

Population projection data provided by the NCTCOG did not cover the entire basin. Therefore, we assumed no change in subbasins without information (figure 11.18). However, with the data provided, the model predicted increases in sediment, total nitrogen and total phosphorus at 3.4%, 7.1% and 6.3%, respectively, in 2030.

Table 11.8 A summary of the changes in sediment and nutrient loads received by Cedar Creek Reservoir under different scenarios as derived from SWAT model simulations.

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	450,000 metric tons/yr (496,040 tons/yr)	189,066 kg/yr (416,819 lbs/yr)	1,422,362 kg/yr (3,135,771 lbs/yr)
No Ponds	+16%	+6.5%	+6.8%
No Range Grazing	-0.3%	-0.02%	-0.7%
Urban *	3.4%	6.3%	7.1%
No Point Sources	-0.2%	-6.0%	-3.8%

* load changes from overland only

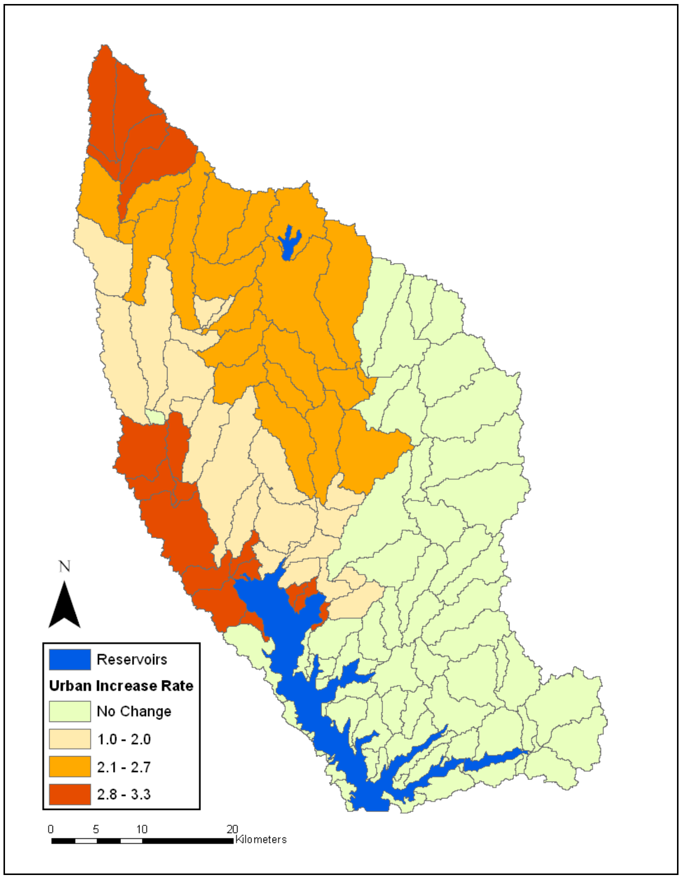


Figure 11.18 Projected changes in urban area from 2000 to 2030.

Conclusions

We modeled the Cedar Creek Reservoir and basin using SWAT to evaluate sediments and nutrients discharged into the reservoir. We set up the model with available data, including GIS, weather, ponds and USGS gauging station flow data. Using various input parameters, SWAT delineated 106 subbasins and channels. The simulation lasted for 37 years (1966–2002). During that time period, flow was calibrated at two USGS gauging stations (08062800 and 08062900) and validated with observed reservoir inflow. The R^2 for monthly flow calibrations at the two gauging stations were 0.819 and 0.813, respectively, and NSE values were 0.81 and 0.83, respectively. Flow validation produced an R^2 of 0.764 and an NSE value of 0.796.

Sediment calibration was based on yearly average sediment loading of the reservoir. The differences between observed and estimated sediment from overland flow and from channel degradation were within 10%. We conducted nutrient calibration using a low flow study and monitoring sites. The results showed that the range of values between 25% and 75% matched well with most of the measured nutrient species at all monitoring locations. The annual average sediment and nutrient loads discharged into the reservoir during the modeling period were 450,000 metric tons (496,040 tons) of sediment per year, 1,422,362 kilograms (3,135,771 pounds) of total nitrogen per year and 189,066 kilograms (416,819 pounds) of total phosphorus per year.

During the scenario analyses, we eliminated ponds, point sources (WWTP) and grazing activity on pastureland. The model results showed that ponds reduced reservoir loading of sediment by 16%, nitrogen by 6.8% and total phosphorus by 6.5%. Removing grazing and maintaining good grass conditions on rangeland reduced sediment, total phosphorus and total nitrogen by 0.3%, 0.7% and 0.02%, respectively. Eliminating point sources showed that there would be less sediment (0.2%), total nitrogen (3.8%) and total phosphorus (6.0%) received by the reservoir if there were no WWTP discharges. The model also predicted urban expansion for 2030. The urban expansion scenario indicated that sediment, total nitrogen and total phosphorus would increase by 3.4%, 7.1% and 6.3%, respectively, with increases in population and urbanization.

References

See the appendix.

Chapter 12: Eagle Mountain Basin

Introduction

The watershed modeling objective of this project was to use the Soil and Water Assessment Tool (SWAT) to assess the effects of urbanization and other land use changes on sediment and nutrient delivery to Eagle Mountain Reservoir. The reservoir is located on the West Fork of the Trinity River primarily in Wise County but also partially in Jack, Clay, Montague Parker and Tarrant counties. Constructed in 1932 as a water supply reservoir for Tarrant County (figure 12.1), the reservoir has a total drainage area of 551,045 acres, and the watershed containing this reservoir will be referred to hereafter as Eagle Mountain Basin. All model data in this report, both observed and simulated, includes inflow to Eagle Mountain Watershed from Bridgeport Reservoir, also constructed in 1932 (figure 12.1). Daily inputs, such as flow, sediment, and nutrients, from Bridgeport Reservoir were represented as a point source in the Eagle Mountain Basin model.

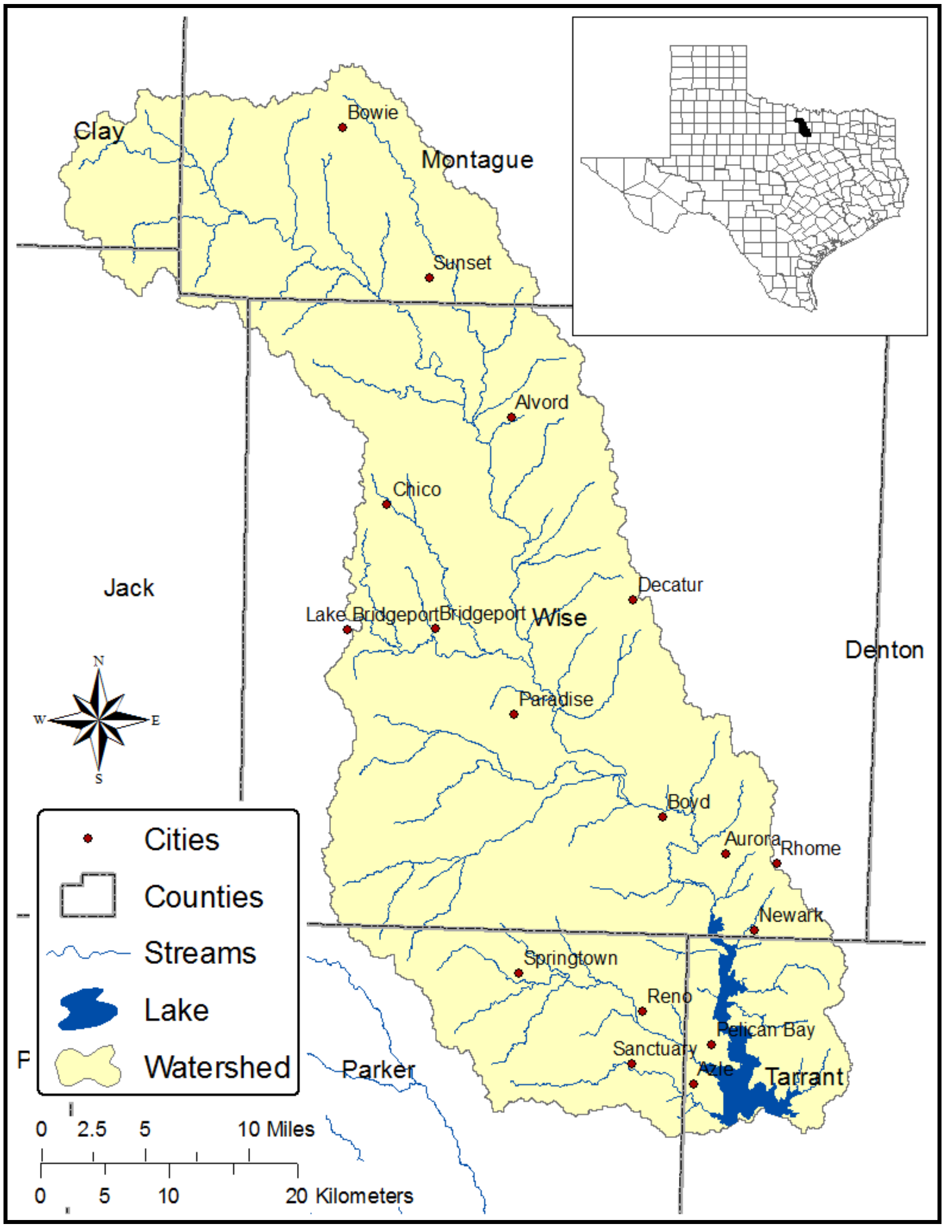


Figure 12.1 The location of Eagle Mountain Basin and the Eagle Mountain Reservoir.

Model Input Data Tables and Figures

Topography

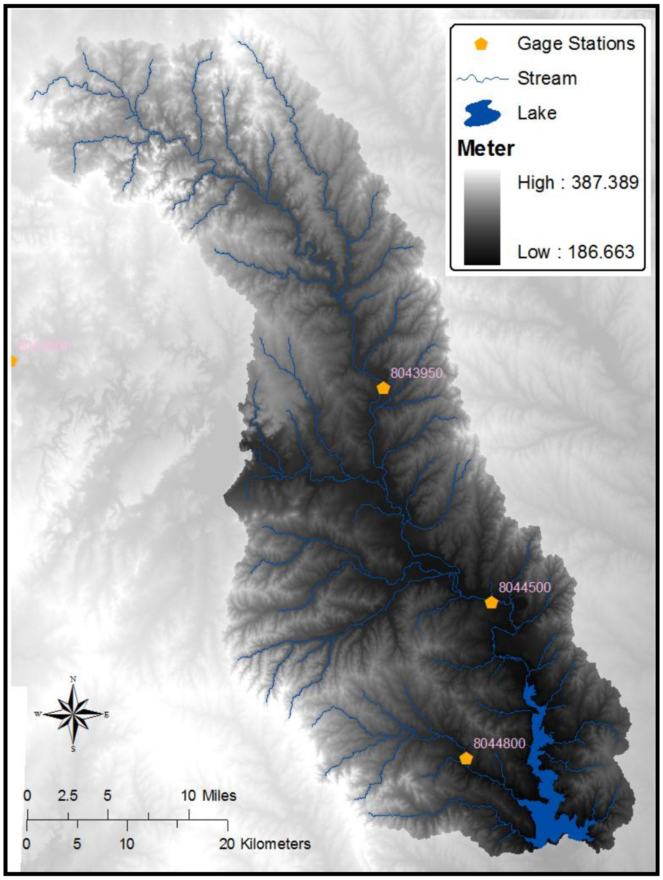


Figure 12.2 A 30-meter (98-foot) Digital Elevation Model provided by the USGS defined the topography of Eagle Mountain Basin.

Soils

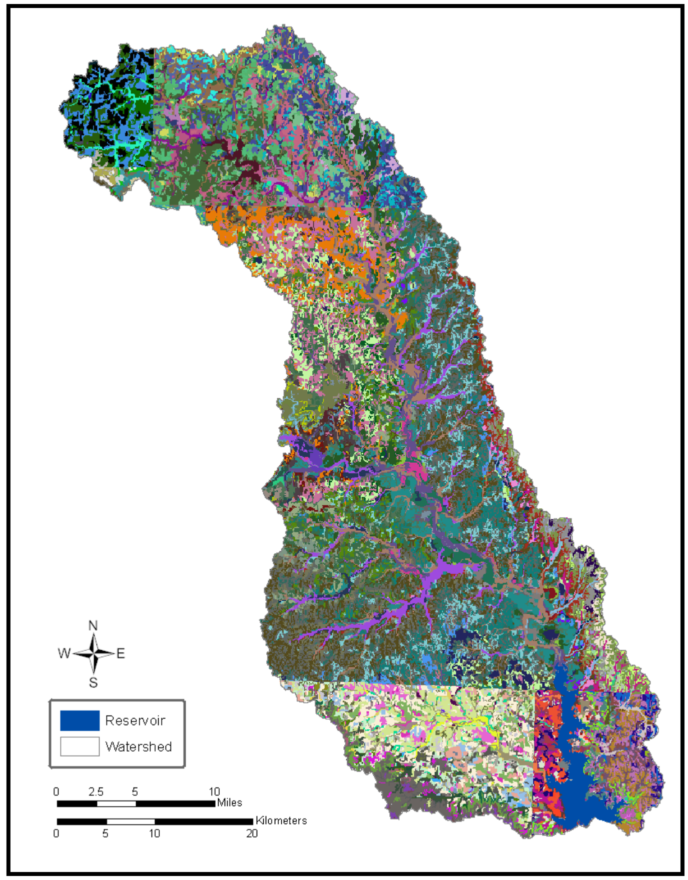


Figure 12.3 SSURGO soils data covers the entire Eagle Mountain Basin, and this figure shows the soil distribution throughout the basin.

Weather

When National Weather Service stations (shown in figure 12.4) lacked precipitation data during the period of record (1950-2004), nearby stations provided substitute data, and SWAT generated missing temperature data.

For rainfall data from 1999–2004, we used NEXRAD data to enhance missing rainfall or create spatially distributed rainfall with finer resolution. We did this by averaging NEXRAD grid data for all subbasins near an individual climate station.

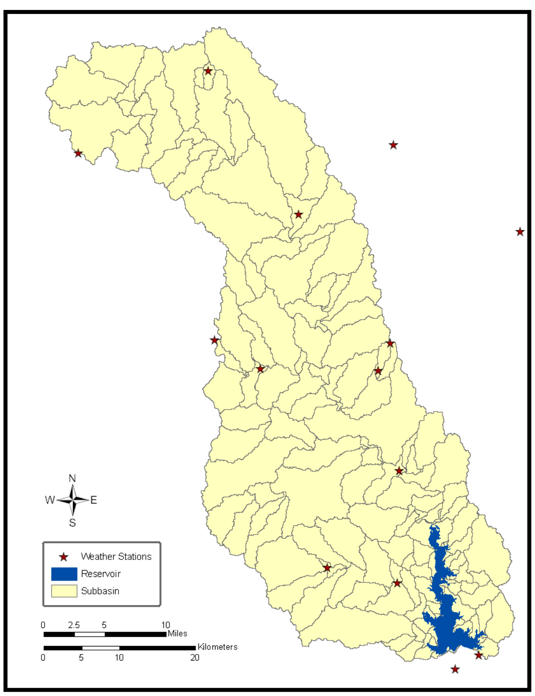


Figure 12.4 The location of National Weather Service stations used to provide temperature and precipitation data for the Eagle Mountain SWAT model (1950–2004).

Land Use

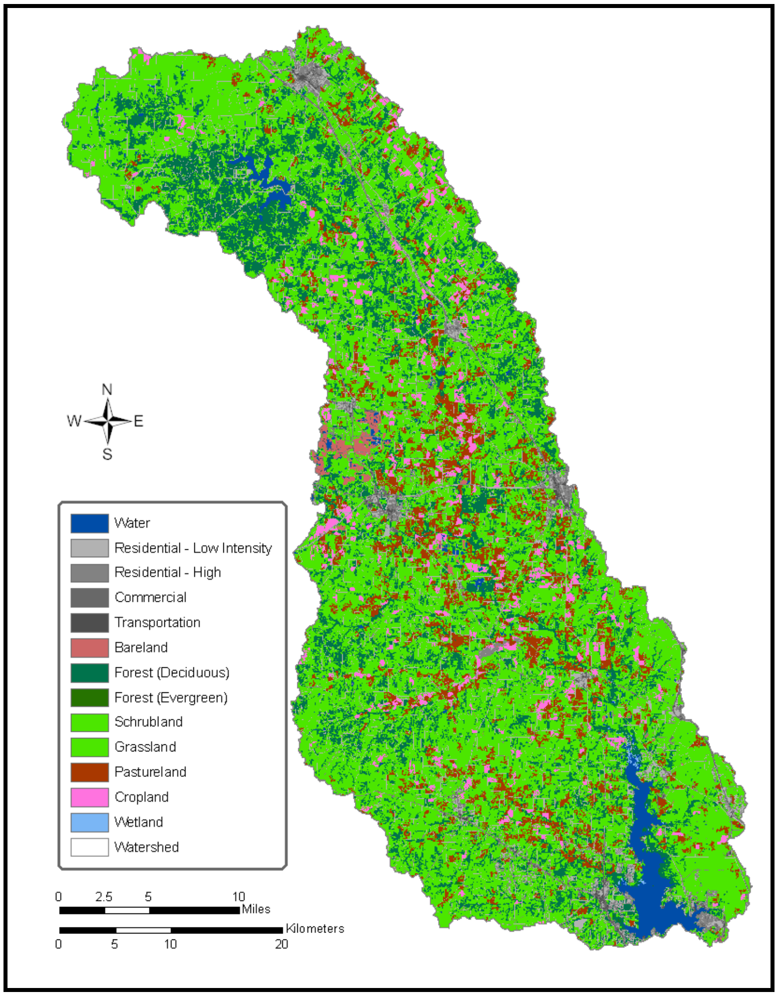
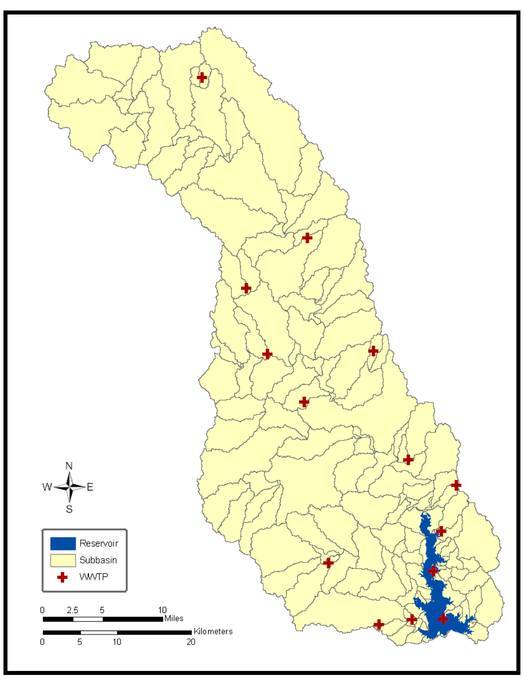


Figure 12.5 The National Land Cover Dataset (NLCD) provided SWAT land use data. Due to rapid urban development in the watershed, the Texas A&M Spatial Sciences Lab enhanced this data for urban expansion using an aerial photograph from 2003.



Point Sources

Eagle Mountain Basin contained a total of 14 Waste Water Treatment Plants distributed across the watershed, and two of these WWTPs discharge directly into the reservoir (figure 12.6). WWTPs voluntarily collected weekly nutrient and flow data for one year, which provided point-source loading inputs. We combined this weekly data into monthly loadings for each WWTP and routed them through the creeks. Point source input data are given in Appendix table A-1.

Figure 12.6 Wastewater treatment plants located in the Eagle Mountain Basin.

Monitoring Stations

In the Eagle Mountain study, we conducted two analyses. One was a low flow study and the other a water quality analysis on samples taken from various monitoring sites. TRWD collected a total of 14 samples in one day (August 18, 2004) and analyzed those samples in the lab. The samples were used to measure the concentration of dissolved oxygen, biological oxygen demand, ammonia, phosphorus, Chlorophyll-a, organic nitrogen and nitrate-nitrite concentrations. The Spatial Sciences Laboratory (SSL) then used observed data from 10 of the 14 locations to calibrate nutrients under low flow conditions. SSL also set up an independent QUAL-2E model based on the measured channel geometry and hydraulics developed during a dye study. Then the calibrated QUAL-2E kinetic terms and coefficients were used as initial estimates of instream water quality parameters in SWAT.

TRWD also set up six monitoring sites where they periodically collected grab samples from 1991 to 2004 to test for water quality (figure 12.7). For SWAT calibration, we used data from five monitoring sites to modify SWAT's instream model parameters.

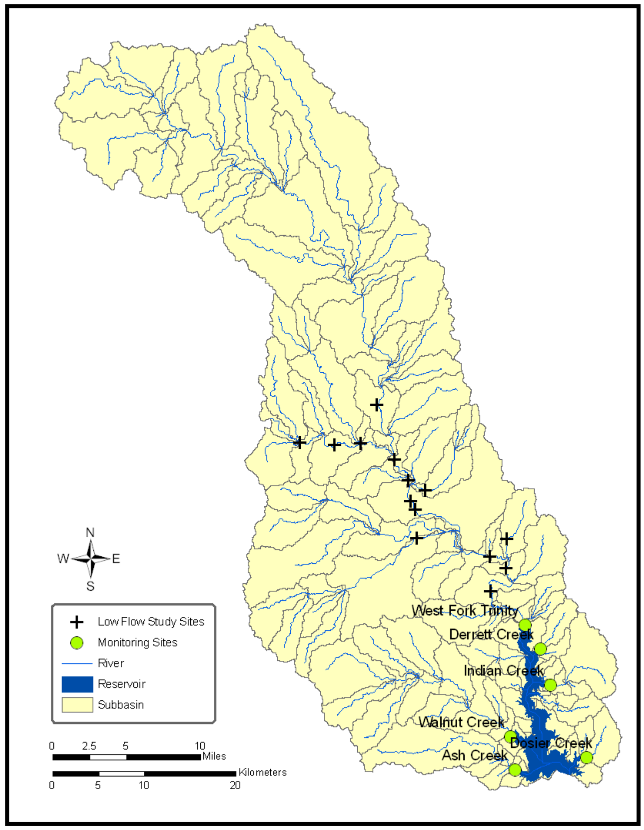


Figure 12.7 The location of monitoring stations that provided water quality data for model calibration.

Ponds

The Eagle Mountain Basin contains a total of 56 inventory-sized dams, as defined by the Texas Commission on Environmental Quality. These include NRCS flood prevention dams, farm ponds and other privately owned dams. We input physical data such as surface area, storage, drainage area and discharge rates for these dams into SWAT to allow routing of runoff through the impoundments. Four were big enough to be simulated as reservoirs while the rest were simulated as small ponds (figure 12.8).

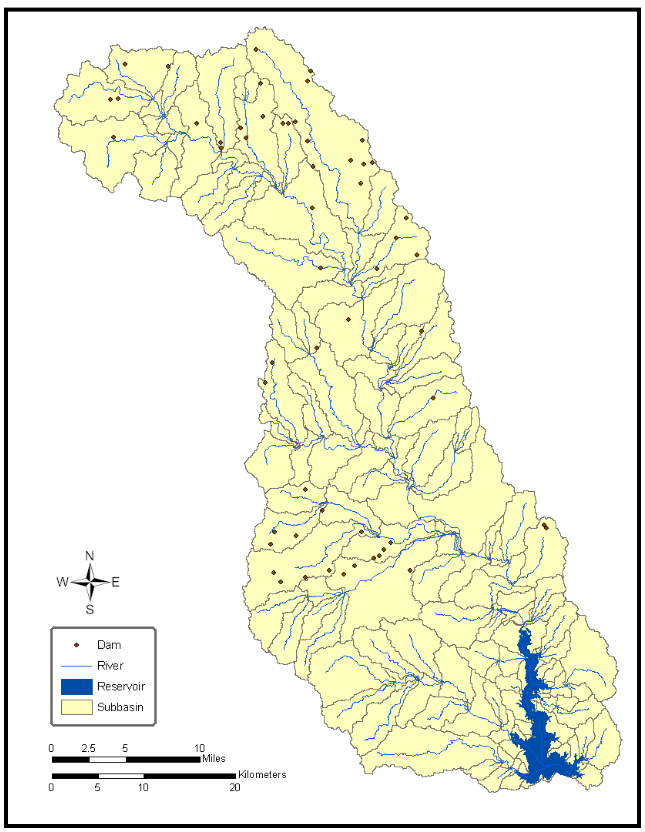


Figure 12.8 The distribution of impoundments in Eagle Mountain Basin.

Table 12.1 The National Inventory of Dams provided the reservoir characteristics for Eagle Mountain Reservoir.

Reservoir	Subbasin	Surface Area at Principle Spillway (acres)	Volume at Principle Spillway (10^4 acre-feet)	Surface Area at Emergency Spillway (acres)	Volume at Emergency Spillway (10^4 acre-feet)	Release
Eagle Mountain	129	8,694	19.0	21,853	68.0	Measured

Subbasin Delineation

We used SWAT 2005 to automatically delineate subbasins within the watershed using a stream definition threshold of 1,236 acres. To improve the accuracy of the subbasin delineation, SWAT also used a stream burn-in theme. We inserted additional subbasin outlets at USGS stream gauges, TRWD tributary sampling points, municipal wastewater discharge points and WASP model input locations (Eagle Mountain Reservoir boundaries). The resulting subbasin map contained 150 subbasins (figure 12.6).

We included inflow from Bridgeport Reservoir as a point source using discharge data that included flow, sediment and nutrient loads. Rather than include Eagle Mountain Reservoir as a reservoir, SWAT simulated several subbasins as partially submerged by the lake. We accounted for the effects of submergence in main channel inputs (channel erodibility and channel cover were set to “0.0”), and we turned QUAL-2E in SWAT off in affected subbasins. SWAT simulated the land cover for these submerged areas as water.

HRU distribution

SWAT’s input interface divided each subbasin into HRUs with unique soil and land use combinations. We determined the number of HRU’s within each subbasin by: (1) creating an HRU for each land use that equaled or exceeded two percent of each subbasin’s area and (2) creating an HRU for each soil type that equaled or exceeded 10 percent of any of the land uses selected in (1). Using these parameters, the interface created 1,516 HRUs within the watershed.

Management

Cropland

NRCS field office personnel provided data on typical crops and management practices in the watershed. We assumed that growers planted grain sorghum on all cropland and applied no conservation practices (Universal Soil Loss Equation (USLE) “P” = 1.0). In the model, SWAT applied fertilizer to cropland at a rate of 61.7 pounds of nitrogen and 6.7 pounds of phosphorus per acre, and conventional tillage was assumed.

Rangeland

According to personal communication with Homer Sanchez, NRCS State Range Conservationist, pastureland in the watershed is in fair hydrologic condition. In the simulation, SWAT applied fertilizer to pastureland at a rate of 122.9 pounds of nitrogen and 17 pounds of phosphorus per acre.

Urban

We assumed that pervious urban land areas were planted with Bermuda grass. SWAT applied fertilizer automatically to those areas, basing application rates and amounts on a nitrogen stress level of 0.9.

Model Calibration and Validation

To evaluate model predicted streamflow during calibration and validation, we used mean, standard deviation, coefficient of determination (R^2), and Nash-Sutcliffe modeling efficiency (NSE) (Nash and Sutcliffe, 1970). We compared mean observed data with mean simulated flow and sediment and nutrient loadings for the days with available grab sample data.

Flow

Two USGS gauges (08043950 and 08044500), shown in figure 12.2, provided streamflow in the watershed from 1991 through 2004. The availability of streamflow data at these two sites determined the calibration period. We used a baseflow filter program (Arnold et al., 1995a) to determine the fraction of baseflow and surface runoff at selected gauging stations.

For each model simulation, we input appropriate plant growth parameters for brush, native grass and other land covers. Initial inputs were based on known or estimated watershed characteristics. We then calibrated SWAT for flow by adjusting appropriate inputs that affect surface runoff and baseflow including the runoff curve number, soil evaporation compensation factor, shallow aquifer storage, shallow aquifer re-evaporation and channel transmission loss. We adjusted these inputs until the simulated total flow and baseflow fraction were approximately equal to the measured total flow and baseflow, respectively.

We validated the model by applying the same model parameter to different time periods from 1971 through 1990. The validation time period was earlier than the calibration period because the land cover data used in this model was from 2001. Therefore, it was better to calibrate the model for the period of time that included the year of the land cover dataset. Figure 12.9 shows the result of flow calibration and validation at USGS gauging station 08044500. For the calibration period, R^2, NSE, observed mean and modeled mean were 0.947, 0.913, 7.15 m^3/s and 7.04 m^3/s, respectively. For the validation period, they were 0.964, 0.921, 8.59 m^3/s and 8.50 m^3/s, respectively.

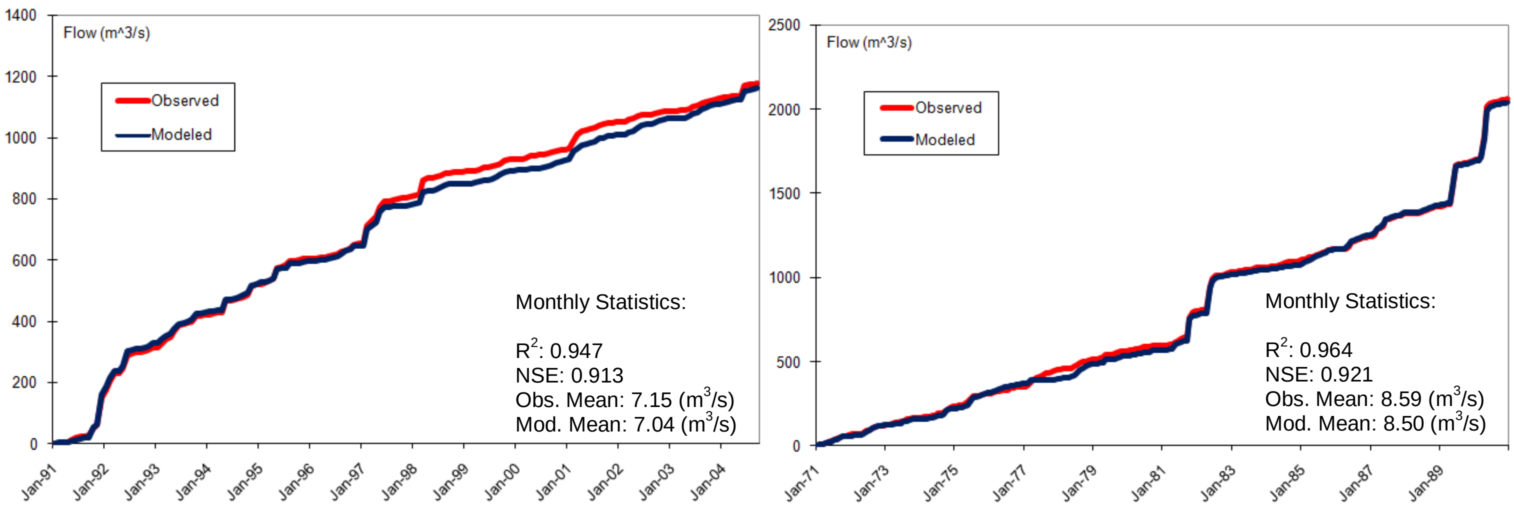


Figure12.9 Flow calibration and validation at USGS gauging station 08044500.

Sediment

Two sediment studies were conducted during this project. Baylor University conducted the first study before our modeling project took place (Allen et al., 2006). Then, the Texas Water Development Board did a second study during our modeling project (TWDB, 2008). Therefore, we conducted sediment calibration again based on the second study because it was considered more accurate and reasonable.

1. **The first study:**

Baylor University undertook a lake sediment survey in early 2006, collecting sediment cores to estimate the average density and thickness of sediment at the lake bottom. Allen et al. (2006) also conducted a watershed survey to identify stream segments with channel erosion problems and to quantify channel erosion using NRCS field assessment techniques such as RAP-M. Allen et al. calculated sedimentation in the Eagle Mountain Reservoir by averaging three historical surveys. They concluded that the sedimentation rate of the lake was 427.3 acre-feet per year, which is equivalent to 376,000 metric tons (414,469 tons) per year. The delta sediment density was 98 pounds per cubic foot, prodelta sediment density was 26 pounds per cubic foot and average density was 40.4 pounds per cubic foot. Based on the lake sediment survey and the watershed survey, researchers estimated that the erosion rate within Eagle Mountain Basin was about 340,883 metric tons (375,759 tons) per year. Out of this total, channel erosion contributed about 110,144 metric tons (121,413 tons) per year (32.3%) while the remaining sediment (230,739 metric tons [254,346 tons] per year) came from overland erosion (Allen et al., 2006). We compared measured sediment with SWAT-simulated sediment over a 34-year period from 1971 to 2004 and adjusted the appropriate input parameters until the predicted annual sediment load from overland and channel erosion were approximately equal to the measured load. Table 12.2 shows the final values for SWAT input coefficients used in flow and sediment calibration.

2. **The second study:**

The TWDB conducted the second study in 2008 using a duel frequency method. Using this technique, they estimated the thickness of the reservoir's post-impoundment sediment. However, they could not measure shallow areas due to boat inaccessibility. This included shallow areas full of sediment near the mouth of major tributaries. Despite this drawback, the second study used the most recent technology and measured with very fine resolution. Therefore, we adopted the measurements of this study and recalibrated sediment. However, we adopted the ratio of channel erosion sediment to overland sediment from Allen et al. (2006).

According to TWDB measurements, the reservoir sedimentation rate was 295,822 metric tons (326,087 tons) per year, which is 45,061 metric tons (49,671 tons) per year less than the study by Allen et al. Channel contribution was estimated at 98,569 metric tons (108,653 tons) per year (33.3%) and overland erosion contribution was estimated at 197,313 metric tons (217,500 tons) per year. Because the TWDB survey did not estimate the contribution from overland and channel flow, it was estimated based on Baylor's study (Allen et al, 2006).

SWAT simulated sediment loads for 34 years, from 1971 to 2004. We then compared simulated and measured sediment and adjusted the appropriate input parameters until the predicted annual sediment load from overland and channel erosion were approximately equal to their corresponding measured loads. Tables 12.4 and 12.5 summarize sediment calibration for overland erosion and for the entire watershed, respectively. Measured and simulated sediment were accepted with 10% difference. Table 12.2 shows the final values for SWAT input coefficients used in flow and sediment calibration.

Table 12.2 SWAT input coefficients adjusted for calibration of flow and sediment.

Component	Parameter (file)	Description	Input Value
Flow	CN2 (*.mgt)	SCS runoff curve number (adjustment range)	+5 to -5
	ESCO (*.hru)	Soil evaporation factor	0.85
	GW_REVAP (*.gw)	groundwater re-evaporation coefficient	0.02
	REVAPMN (*gw)	Groundwater storage required for revap (mm)	1
	ALPHA_BF (*gw)	Baseflow alpha factor (days^{-1})	0.0431 to 0.0670
	CH_N2 (*.rte)	Mannings "n" roughness for channel flow	0.125
	CH_K2 (*.rte)	Hydraulic conductivity of channel alluvium (mm/hr)	0.5 to 5.0
Sediment	USLE_C (crop.dat)	Minimum "C" value for pastureland in fair condition	0.007
	SPCON (basins.bsn)	Linear parameter for calculating the maximum amount of sediment that can be reentrained during channel sediment routing	0.003
	SPEXP (basins.bsn)	Exponent parameter for calculating sediment reentrained in channel sediment routing	0.67
	TRNSRCH basins.bsn	Reach transmission loss partitioning to deep aquifer	0.2
	CH_COV (*.rte)	Channel cover factor	0.001 to 0.9
	CH_EROD (*.rte)	Channel erodibility factor	0.001 to 0.9

Nutrients

Nutrient calibration also consisted of two parts. First, we calibrated the model based on a low flow study conducted August 18, 2004. Secondly, we calibrated SWAT using monitoring data.
In the first step, we adjusted SWAT parameters to agree with measured data at 10 sampling sites where sediment, nutrients and biochemical species data were collected during low flow conditions (figure 12.7). Due to low baseflow conditions, nutrients discharged from WWTPs and channel processes were very high during calibration. However, there was a 0.67 inch rain event in the northeast part of watershed on Aug 16, 2004, and it may have impacted the data.

In the second step of the calibration, we adjusted the remaining subbasin parameters using monitoring station data. The simulation period lasted from 1971 through 2004. A year's worth of monthly data collected by the TRWD in 2001 and 2002 defined WWTP loads, and we assumed WWTP loadings at each facility were the same every year starting with the first year of operation.

We compared the output from this simulation to water quality data collected by TRWD from 1991 through 2004 in each major tributary (Ash, Derrett, Dosier, Walnut and West Fork 4688 as shown in figure 12.7). In order to account for daily SWAT variability, we averaged simulated output for the three days surrounding the day of the measured grab sample. We compared the median, 25th percentile and 75th percentile of the three-day averages derived from SWAT simulations with those derived from the measured grab samples (figure 12.10–12.12). Then we adjusted coefficients for all subbasins within the watershed to match observed data (table 12.3). Some sites showed disagreement between observed and measured data, but the West Fork 4688 site, located at the end of the main channel before the lake entrance, agreed relatively well.

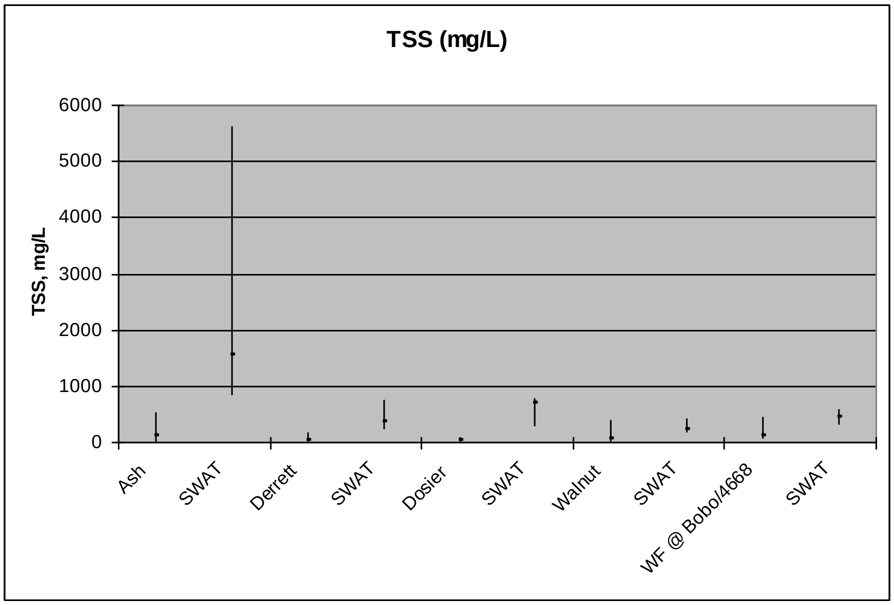


Figure 12.10 Median, 25th and 75th percentile of total suspended sediment loads simulated and measured at monitoring sites throughout the Eagle Mountain Basin from 1991 to 2004.

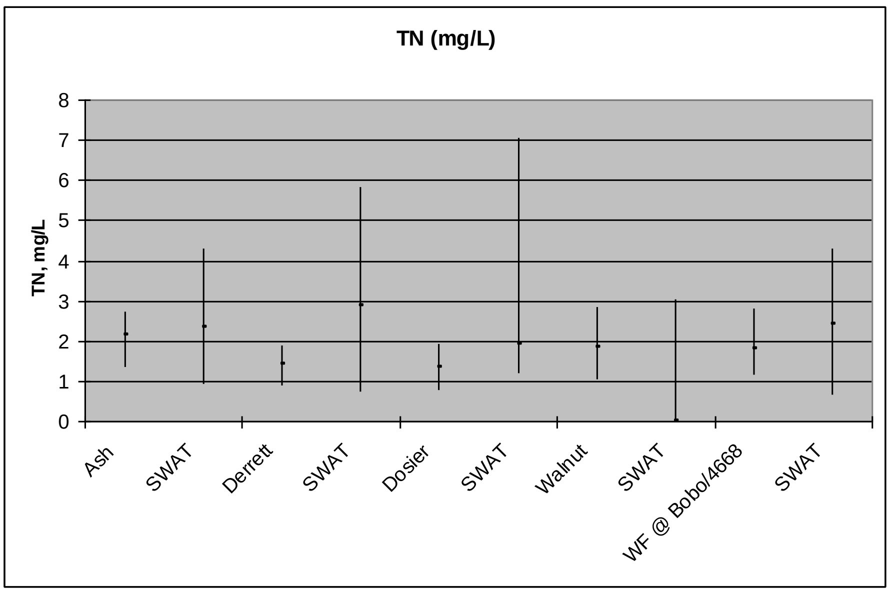


Figure 12.11 Median, 25th and 75th percentile of total nitrogen loads simulated and measured at monitoring sites throughout the Eagle Mountain Basin from 1991 to 2004.

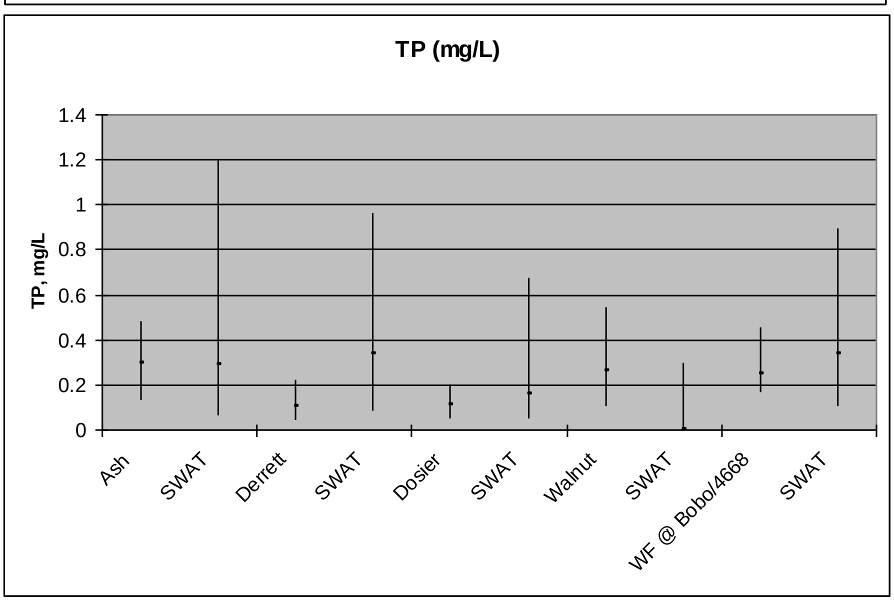


Figure 11.12 Median, 25th and 75th percentile of total phosphorus loads simulated and measured at monitoring sites throughout the Eagle Mountain Basin from 1991 to 2004.

Table 12.3 General water quality input coefficients (.wwq) for both the low flow study and monitoring site calibration.

Variable Name	Definition	SWAT-SSL	SWAT	SWAT
		Cal. Coef.	Default	Range
LAO	Light averaging option	2	2	2
IGROPT	Specific algal growth rate option	2	2	3 options
AI0	Ratio of chlorophyll-a to algal biomass [µg-chla/mg algae]	10	50	10–100
AI1	Fraction of algal biomass that is nitrogen [mg N/mg alg]	0.090	0.080	0.07–0.09
AI2	Fraction of algal biomass that is phosphorus [mg P/mg alg]	0.020	0.015	0.01–0.02
AI3	The rate of oxygen production per unit of algal photosynthesis [mg O_2/mg alg]	1.500	1.600	1.4–1.8
AI4	The rate of oxygen uptake per unit of algal respiration [mg O_2/mg alg]	2.300	2.000	1.6–2.3
AI5	The rate of oxygen uptake per unit of NH_3-N oxidation [mg O_2/mg NH_3-N]	3.500	3.500	3.0–4.0
AI6	The rate of oxygen uptake per unit of NO_2-N oxidation [mg O_2/mg NO_2-N]	1.000	1.070	1.0–1.14
MUMAX	Maximum specific algal growth rate at 20ºC [day-1]	2.000	2.000	1.0–3.0
RHOQ	Algal respiration rate at 20ºC [day-1]	0.300	0.300	0.05–0.50
TFACT	Fraction of solar radiation computed in the temperature heat balance that is photosynthetically active	0.440	0.300	0.01–1.0
K_L	Half-saturation coefficient for light [kJ/(m2·min)]	0.418	0.750	0.2227–1.135
K_N	Michaelis-Menton half-saturation constant for nitrogen [mg N/lL]	0.400	0.020	0.01–0.30
K_P	Michaelis-Menton half-saturation constant for phosphorus [mg P/l]	0.040	0.025	0.001–0.05
LAMBDA0	Non-algal portion of the light extinction coefficient [m-1]	1.500	1.000	-
LAMBDA1	Linear algal self-shading coefficient [m-1·(µg chla/l)-1)]	0.002	0.030	0.0065–0.065
LAMBDA2	Nonlinear algal self-shading coefficient [m-1·(µg chla/l)-2]	0.054	0.054	0.054
P_N	Algal preference factor for ammonia	0.100	0.500	0.01–1.0

Table 12.4 Calibration and validation for sediment loading from overland flow.

	Observed Metric tons (tons)	**Modeled Metric tons** (tons)	**Difference** (%)
Total (y^{-1})	197,313 (217,500)	196,909 (217,055)	-0.2
Calibration (1994 – 2004)		206,294 (227,400)	+4.6
Validation (1970 – 1990)		191,748 (211,366)	-2.8

Table 12.5 Calibration and validation of reservoir sediment loading.

	Observed in Metric tons (tons)	**Modeled in Metric Tons** (tons)	**Difference** (%)
Total (y^{-1})	295,822 (326,087)	290,400 (320,111)	+0.2
Calibration (1994 – 2004)		263,827 (290,819)	-10.8
Validation (1970 – 1990)		324,880 (358,118)	+9.8

Table 12.6 Estimated yearly sediment and nutrients discharged into Eagle Mountain Reservoir.

	Sediment in metric tons (tons)	**Total N in kg/yr** (lb/yr)	**Total P in kg/yr** (lb/yr)
Calibrated model estimation (baseline)	296,400 (326,725)	1,057,437 (2,331,250)	173,383 (382,244)

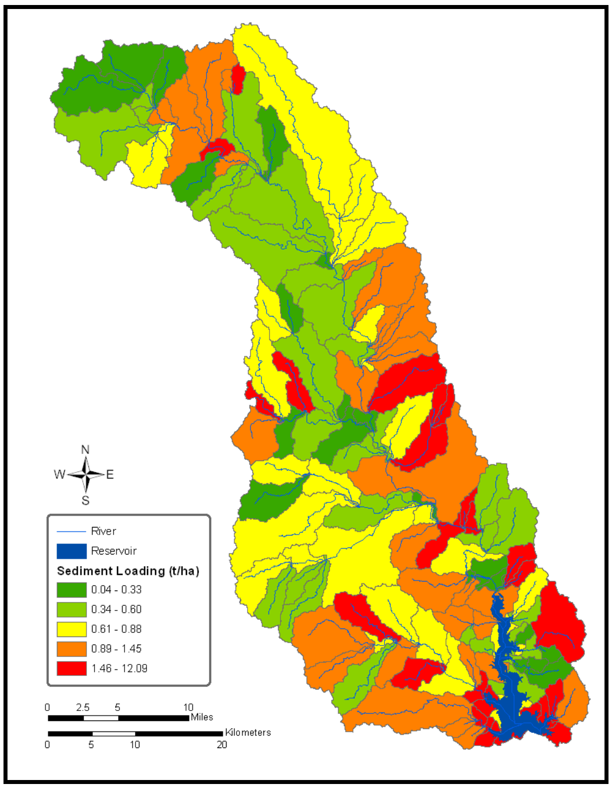


Figure 12.13 SWAT-predicted sediment losses produced by overland flow.

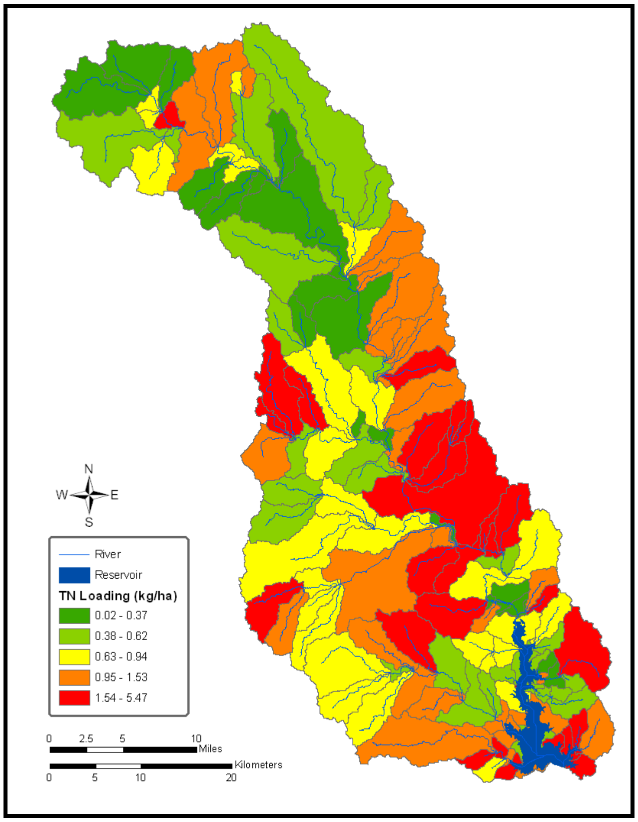


Figure 12.14. SWAT-predicted total nitrogen losses produced by overland flow.

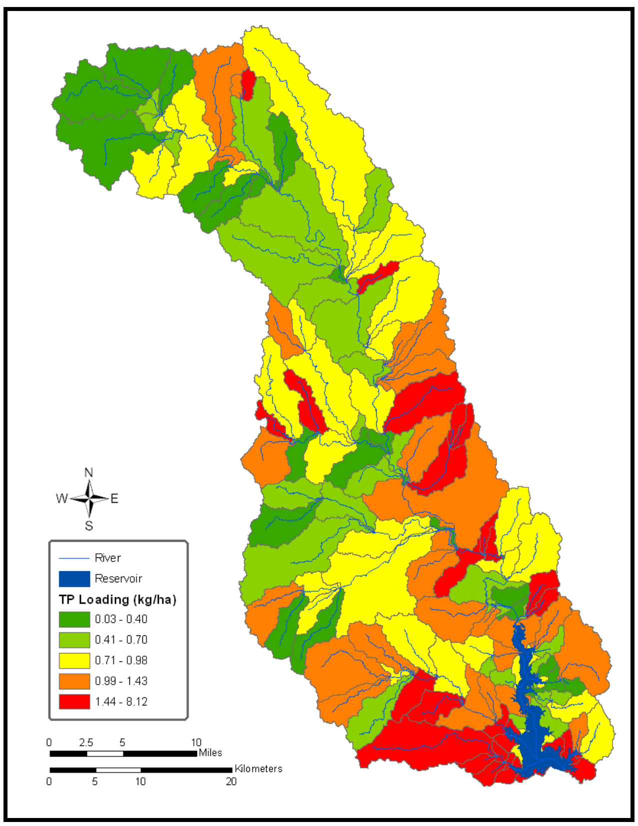


Figure 12.15 SWAT-predicted total phosphorus losses produced by overland flow.

Scenarios

Conservation Practices

We simulated several conservation practices to evaluate potential reductions in sediment and nutrient loads. SWAT-predicted reductions or increases are shown in a table below each scenario.

Ponds

According to SWAT, the 56 NRCS floodwater dams and private ponds simulated in the Eagle Mountain Basin reduced reservoir loading of sediment, total nitrogen and total phosphorus by 8.2%, 9.6% and 7.9%, respectively (table 12.7).

Table12.7 The impact of ponds on yearly sediment and nutrient loading as predicted by SWAT.

	Sediment in metric tons (tons)	**Total N in kg/yr** (lb/y)	**Total P in kg/yr** (lb/y)
With ponds –baseline-	296,400 (326,725)	1,057,437 (2,331,250)	173,383 (382,244)
Without ponds	320,600 (353,301)	1,159,331 (2,555,887)	187,012 (412,291)
Difference	+8.2%	+9.6%	+7.9%

Range Utilization

By removing grazing from rangeland within the basin, reservoir loading of sediment, total nitrogen and total phosphorus decreased by 4.8%, 3.3% and 7.8%, respectively.

Table 12.8 The impact of grazing on yearly sediment and nutrient loading as predicted by SWAT.

	Sediment in metric tons (tons)	**Total N in kg/yr** (lb/y)	**Total P in kg/yr** (lb/y)
With grazing -baseline-	296,400 (326,725)	1,057,437 (2,331,250)	173,383 (382,244)
Without grazing	282,300 (311,182)	1,022,604 (2,254,456)	159,835 (352,376)
Difference	-4.8%	-3.3%	-7.8%

Point Source Load Elimination

We eliminated 10 WWTPs from the model by making all discharge zero, reducing reservoir loading of sediment, total nitrogen and total phosphorus by 1.9%, 1.2% and 1.1%, respectively.

Table 12.9 The impact of removing point sources on yearly sediment and nutrient loading as predicted by SWAT.

	Sediment in metric tons (tons)	**Total N in kg/yr** (lb/y)	**Total P in kg/yr** (lb/y)
With WWTPs -baseline-	296,400 (326,725)	1,057,437 (2,331,250)	173,383 (382,244)
Without WWTPs	290,800 (320,552)	1,044,270 (2,302,221)	171,480 (378,049)
Difference	-1.9%	-1.2%	-1.1%

Urbanization

The North Central Texas Council of Governments' population projection data did not cover the entire watershed, so for those subbasins without information, we assumed there was no change (figure 12.16). For subbasins in which we applied the fraction increase, we employed the same methodology as in previous chapters (see Chapter 2). In the 2030 projections, SWAT predicted increases in sediment, total nitrogen and total phosphorus of 31.6%, 16.9% and 3.3%, respectively.

Table 12.10 A summary of the changes in sediment and nutrient loads received by Eagle Mountain Reservoir under different scenarios as derived from SWAT model simulations.

Scenario	Sediment	Total Phosphorus	Total Nitrogen
Baseline	296,400 metric tons/yr (326,725 tons/yr)	1,057,437 kg/yr (2,331,250 lb/yr)	173,383 kg/yr (382,244 lb/yr)
No Ponds	+8.2%	+7.9%	+9.6%
No Range Grazing	-4.8%	-7.8%	-3.3%
Urban *	31.6%	3.3%	16.9%
No WWTPs (Point Sources)	-1.9%	-1.1%	-1.2%

* load changes from overland only

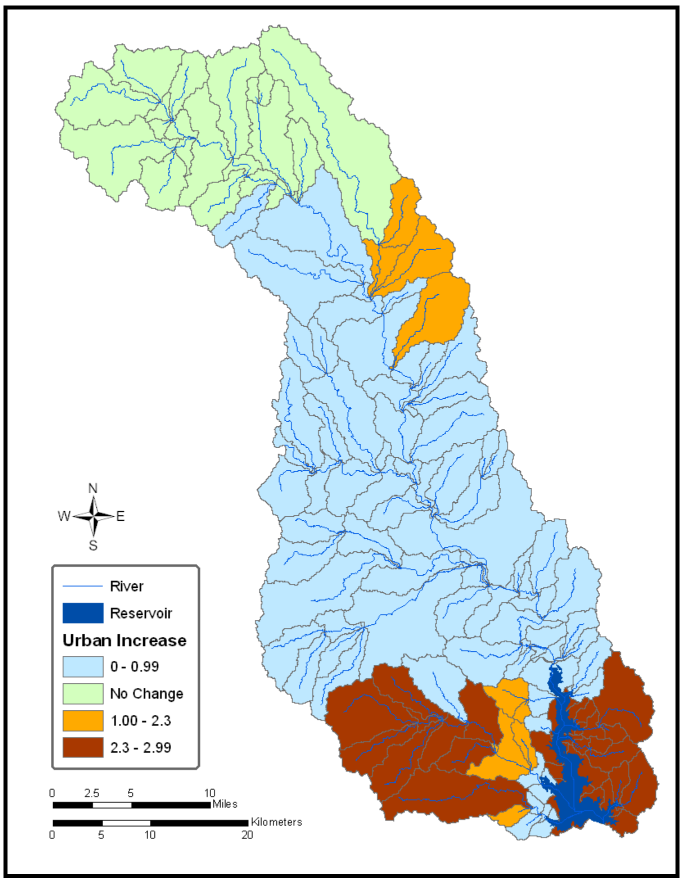


Figure 12.16 Projected changes in urban area from 2000 to 2030.

Conclusions

Using SWAT, we modeled the Eagle Mountain Reservoir and basin, evaluating flow, sediments and nutrients discharged into the reservoir. With available data, including GIS, weather, pond and USGS flow data, we set up the SWAT model and delineated 150 subbasins and channels.

SWAT simulated the reservoir for 36 years (1969–2004). We calibrated flow at two USGS gauging stations (08043950 and 08044500) from 1994 to 2004 and validated the model from 1971 to 1990. For monthly flow calibration and validation, R^2 was 0.947 and 0.964, respectively, and NSE was 0.913 and 0.921, respectively.

We calibrated the model for sediment with yearly average reservoir loading data based on a study by Allen et al. (2006) then re-calibrated based on a TWDB survey. The differences in observed and estimated sediment from both overland and channel erosion were within approximately 10%. We calibrated nutrients using data from our low flow study and TRWD's sampling data. The results showed that the range of values between 25% and 75% matched well with most of the measured nutrient species at all monitoring locations. The yearly average sediment and nutrient loads discharged into the reservoir for the modeling period were 296,400 metric tons (326,725 tons) of sediment per year, 1,057,437 kilograms (2,331,250 pounds) of total nitrogen per year and 173,383 kilograms (382,244 pounds) of total phosphorus per year.

Scenario analyses included the removal of ponds, WWTPs and rangeland grazing. The model results showed that ponds reduced 8.2% of sediment, 9.6% of total nitrogen and 7.9% of total phosphorus received by the reservoir. Eliminating rangeland grazing and maintaining range grass reduced sediment, total phosphorus and total nitrogen by 4.8%, 3.3% and 7.8%, respectively. Eliminating point sources revealed that there would be less sediment (1.9%), total nitrogen (1.2%) and total phosphorus (1.1%) received by the reservoir. The model also predicted the effects of urban expansion for 2030, indicating that sediment, total nitrogen and total phosphorus would increase by 31.6%, 16.9% and 3.3%, respectively.

References

See the appendix.

Chapter 13: Conclusion

Summary of Conclusions

The ten watersheds simulated in this study ranged in size from 143,321 acres (Joe Pool) to 1,274,322 acres (Richland-Chambers). The watersheds exhibited wide variations in the generation and transport of constituent pollutants, reflecting the variability among watersheds in land use/land cover, land management, soils, slope and climate. For example, annual overland erosion rates ranged from .0647 metric tons per acre (.0714 tons per acre) for the Bridgeport Basin to 0.643 metric tons per acre (0.709 tons per acre) for the Lewisville Basin. Annual overland total nitrogen rates ranged from 0.170 kilograms per acre (.375 pounds per acre) for Bridgeport to 3.79 kilograms per acre (8.35 pounds per acre) for Ray Hubbard. Finally, annual total phosphorus rates ranged from 0.105 kilograms per acre (0.232 pounds per acre) for Bridgeport to 0.834 kilograms per acre (1.84 pounds per acre) for Ray Hubbard.

As simulated by the SWAT model, in most watersheds (especially Richland-Chambers, Lavon and Ray Hubbard) more sediment reached the reservoir than was generated from overland erosion. This indicates that eroding streambanks and beds were important factors in reservoir sedimentation. Detailed studies of fluvial morphology and streambank and bed erosion would be needed to confirm these results and identify sensitive reaches needing stabilization and restoration.

Half of the Ray Hubbard watershed is developed, and wastewater treatment plants (WWTPs) serving urban areas within the watershed appear to contribute a large portion of the nutrient loads reaching the lake. Therefore, upgrading these WWTPs could substantially reduce nutrient loading of the lake. Point sources also appear to contribute substantially to Lake Lavon and Ray Hubbard's nutrient loads. In all other basins, point-source contributions appear to be relatively less important than nonpoint sources.

Among the watersheds simulated, there was a wide range in the number and area drained by small flood control reservoirs (ponds), most of which are known as PL-566 structures after the law authorizing their construction. Ponds receive runoff from upstream drainage area and aid in the settling of sediment from overland erosion. Therefore, ponds greatly reduce sediment loading from overland and aid in the removal of nutrients entering ponds. The removal efficiency is in part a function of detention time. Many small ponds discharge only a small fraction of the water they receive in a typical year, and have very long detention times. Simulations with and without these ponds suggested that their effectiveness in reducing sedimentation of water supply reservoirs varied among watersheds. The percentages of basin area drained into ponds, locations of ponds, and basin's erosion characteristics all contribute to the different effectiveness. In general, there is a positive relationship between the reductions of sediment loadings to the receiving reservoirs and areas drained into ponds. For example, the percentages of basins' areas drained into ponds are approximately 12% for each of the Bridgeport, the Joe Pool, and the Benbrook basin, 18% for Ray Roberts, and 24% for Lewisville basin. While comparing modeling results between with and without ponds' scenarios, the percentages of sediment loadings to receiving reservoirs were reduced by 9.6%, 9.9%, 13.1%, 19.4% and 48.3% (table 13.1), respectively. However, if the main contribution of sediment loadings to reservoirs are from streambanks (vulnerable streambanks), then one wouldn't expect to see great effectiveness in the reduction of total sediment loading to receiving reservoirs. For example, the percentages of basins' areas drained into ponds are 34% for Lavon basin and 32% for Richland-Chambers basin, the highest areas drained into ponds among these study basins. However, with these ponds the percentages of sediment loadings to receiving reservoirs were reduced by only 19% and 8%, respectively (table 13.1). The Ray Hubbard basin has approximately 15% areas drained into ponds, but has only 4% reduction in sediment loading to the Ray Hubbard reservoir, the smallest reduction among these study basins. This is because that the three watersheds have the highest streambank/channel contribution of total sediment loadings to their respective receiving reservoirs, with streambank/channel contributed 84% of total sediment loading to Richland-Chambers, 69% for Ray Hubbard and 55% for Lake Lavon (see also figure 13.1 for the differences between the total and overland portion). While ponds are effective in settling sediment received from overland, with more clear water from upstream than that compared to the without ponds condition, the flow energy is greater to erode vulnerable streambanks and carry on more sediment being re-entrained from channel. Therefore, while implementing conservation practices on upland areas, it is also necessary to have corresponding conservation practices in channels for maximizing potential gains.

The simulated effects of eliminating grazing also varied among watersheds, producing reductions in sediment that ranged from 0.3% to 37%, 0.3% to 19% in total nitrogen, and 0.02% to 11% in total phosphorus.

The water quality effects of predicted increases in urban area (by 2030) also varied among watersheds. For most reservoirs, soil erosion from overland areas increased with increasing urbanization (from 1% to 32%), but simulated overland erosion decreased in Ray Hubbard, Lavon, Lewisville and Joe Pool because urban land uses replaced agricultural land uses that were more susceptible to erosion. Increased urbanization caused changes in overland total nitrogen losses ranging from a decrease of 3% to an increase of 24%. Similarly, projected urbanization increased simulated total phosphorus losses from upland areas by 3% to 111%, except for Richland-Chambers where total phosphorus losses decreased by 1%. This decrease was due to the fact that pastureland and cropland dominate the Richland-Chambers watershed and are large contributors to nutrient loading. Therefore, nutrient loads in this watershed decreased with urbanization due to proportional decreases in pastureland and cropland. However, note that increases in point source loading (WWTP discharges) due to urbanization were not considered in this study.

Simulated sediment and nutrient losses and reservoir loadings are illustrated in the following figures (13.1-13.3). Percent reductions in constituent loading of the lakes due to various management scenarios are presented in table 13.1.

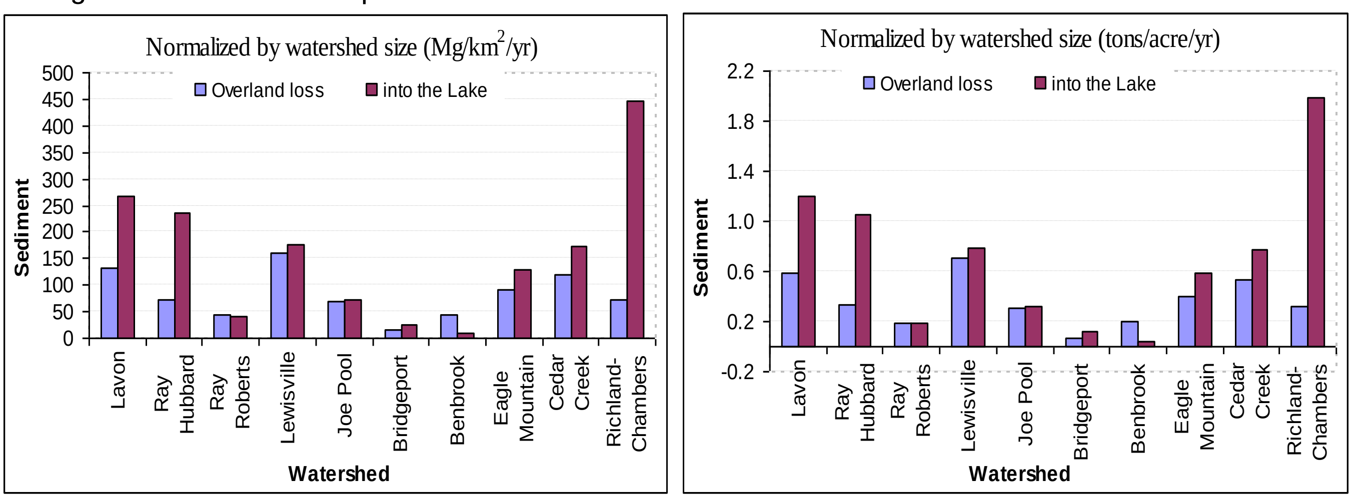


Figure 13.1 Sediment loss from overland and sediment into the lakes.

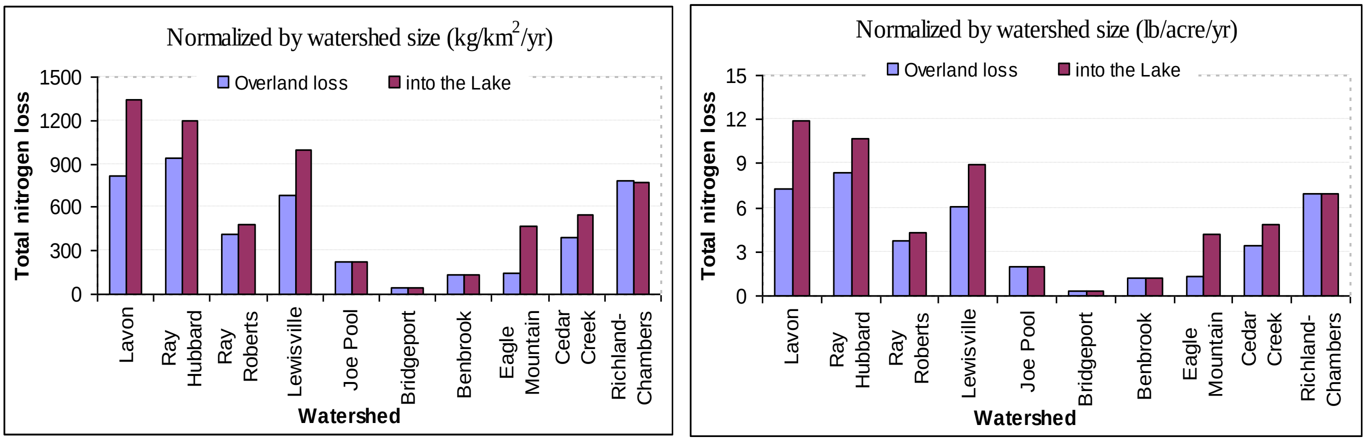


Figure 13.2 Total nitrogen loss from overland and into the lakes. Note that point sources discharging into stream reaches increases eventual lake loading. Organic nitrogen is closely associated with sediment loads.

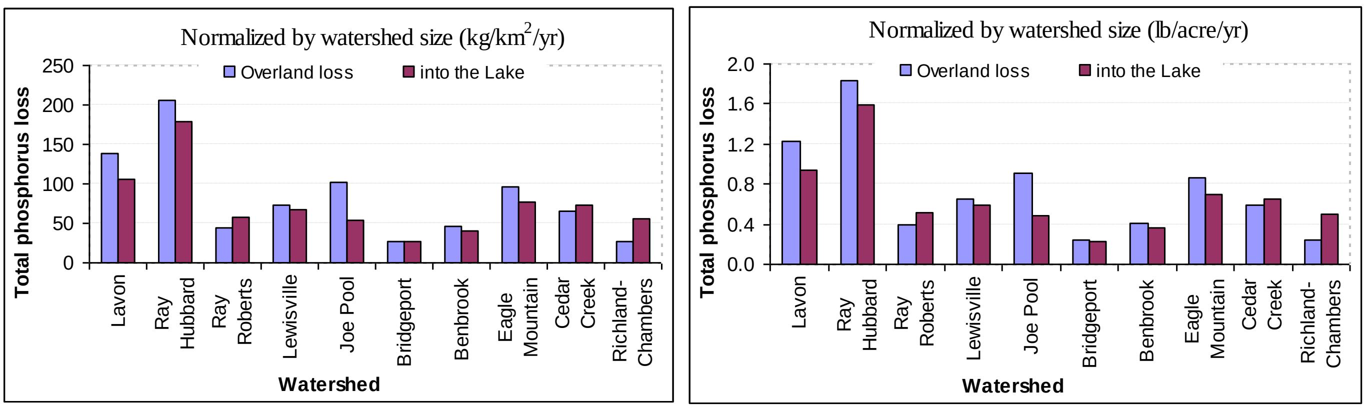


Figure 13.3 Total phsophorus loss from overland and into the lakes. Note that point sources discharging into stream reaches increases eventual lake loading. Organic phosphorus is closely associated with sediment loads.

Table 13.1 Summary of baseline constituent loads compared with percent reductions in sediment, total nitrogen and total phosphorus loading of the lakes as produced by different management scenarios.

	Percent reduction from baseline scenario									
	Lavon	**Ray Hubbard**	**Ray Roberts**	**Lewisville**	**Joe Pool**	**Bridgeport**	**Benbrook**	**Eagle Mountain**	**Cedar Creek**	**Richland-Chambers**
Sediment										
Baseline lake loading, metric tons/yr (tons/yr)	535,400 (590,177)	214,700 (236,666)	73,860 (81,417)	440,800 (485,899)	42,368 (46,703)	74,000 (81,571)	10,700 (11,795)	296,400 (326,725)	450,000 (496,040)	2,302,469 (2,538,038)
No ponds (%)	19.40	3.50	19.40	48.30	9.90	9.60	13.10	8.20	16.00	7.50
No range grazing (%)	-9.20	-6.30	-2.90	-37.20	-7.70	-7.60	-16.70	-4.80	-0.30	-1.00
Urban* (%)	-2.90	-10.30	1.00	-2.5	-3.60	-1.70	-0.03	31.60	3.40	3.80
No point sources (%)	-5.50	-0.60	-0.10	-0.20	-4.40	0.00	0.00	-1.90	-0.20	-0.35
Total Nitrogen										
Baseline lake loading, kg/yr (lb/yr)	2,671,500 (5,889,648)	1,088,650 (2,400,062)	646,060 (1,424,318)	2,518,920 (5,553,268)	129,885 (286,347)	113,000 (249,122)	145,000 (319,670)	1,055,220 (2,326,362)	1,419,380 (3,129,197)	4,011,580 (8,844,020)
No ponds (%)	7.50	8.30	7.90	6.30	4.40	5.90	5.10	9.60	6.80	5.30
No range grazing (%)	-0.30	-1.57	-3.10	-3.30	-2.00	-19.00	-7.30	-3.30	-0.70	-9.10
Urban* (%)	9.20	-2.60	4.70	2.90	16.40	10.30	23.60	16.90	7.10	-2.60
No point sources (%)	-55.70	-20.40	-5.90	-27.20	-7.30	-13.00	-23.40	-1.10	-3.80	-4.00
Total Phosphorus										
Baseline lake loading, kg/yr (lb/yr)	210,750 (464,624)	162,480 (358,207)	103,590 (228,377)	168,080 (370,553)	31,339 (69,091)	74,000 (163,142	44,500 (98,106)	173,020 (381,444)	188,670 (415,946)	285,104 (628,547)
No ponds (%)	12.70	7.40	6.40	10.00	5.80	8.60	5.80	7.90	6.50	5.80
No range grazing (%)	-11.30	-2.58	-5.00	-9.10	-2.80	-9.70	-3.10	-7.80	-0.02	-5.70
Urban* (%)	14.40	3.20	16.60	23.50	37.40	30.70	111.00	3.30	6.30	-1.05
No point sources (%)	-24.20	-7.80	-6.00	-14.40	-1.00	-2.90	-9.30	-1.20	-6.00	-0.25

* load changes from overland only

This study provides an overall estimate of the magnitude and spatial distribution of pollutants generated by overland flow as well as the amount actually delivered to the 12 major reservoirs within the Trinity River Basin. A more detailed study for each of the basins assessed in this study will provide greater insight into the sources and sinks of various constituents. It would also allow planners to devise management practices that reduce the generation and transport of pollutants in the landscape.

Although the SWAT model has been successfully applied in ungauged basins, thorough model calibration and validation increases the confidence in its applicability. Insufficient data, especially related to point sources and observed data on sediment and nutrients makes model calibration and validation a challenging task. A detailed study including a revision of some of the assumptions made in this report and an uncertainty analysis would help quantify the uncertainty bounds around the predicted values. Nevertheless, modeling studies using a comprehensive semi-distributed model such as SWAT do help simulate and assess the pollutant generation and transport potential of the landscape.

Appendix

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List of Abbreviations

AI1	Fraction of algal biomass that is nitrogen
AI2	Fraction of algal biomass that is phosphorus
ARS	Agricultural Research Service
BASINS	Better Assessment Science for Integrated Point and Nonpoint Sources
BC2	Rate constant for biological oxidation of NO_2 to NO_3 in the reach at 20^{o}C (day^{-1})
BC3	Rate constant for hydrolysis of organic N to NH_4 in the reach at 20ºC (day^{-1})
BC4	Rate constant for mineralization of organic P to dissolved P in the reach at 20^{o}C (day^{-1})
BMP	Best Management Practice
CDN	Denitrification exponential rate coefficient
C-factor	Land surface cover factor
CFRG	Coarse Fragment Factor
CH_COV	Channel cover factor
CH_EROD	Channel erodibility factor
CH_K(1)	Effective hydraulic conductivity in tributary channel alluvium (mm/hr)

CH_N(1)	Manning's "n" value for the tributary channel
CH_N(2)	Manning's "n" value for the main channel
CII	Concentration Impact Index
CMN	Rate factor for humus mineralization of active organic nutrients (N and P)
CN	Curve Number (Soil Conservation Service)
CN2	Initial SCS runoff curve number for moisture condition II
DEM	Digital Elevation Model
DEPTIL	Depth of Mixing caused by tillage operation
EFFMIX	Mixing Efficiency
EPCO	Plant uptake compensation factor
ESCO	Soil evaporation compensation factor
FILTERW	Width of edge-of-field filter strip (m)
FRT_SURFACE	Fraction of fertilizer applied to top 10mm of soil
GIS	Geographical Information System
GW_REVAP	Groundwater revap coefficient
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur
HI	Harvest Index
HRU	Hydrologic Response Unit
HUC	Hydrologic Unit Code
LII	Load Impact Index
LUAII	Load per Unit Area Impact Index
MinN	Mineral Nitrogen
MinP	Mineral Phosphorus
MUMAX	Maximum specific algal growth rate (day^{-1})
N	Nitrogen
NCDC	National Climatic Data Center
NID	National Inventory of Dams
NPERCO	Nitrate percolation coefficient
NRCS	Natural Resources Conservation Service
NSE	Nash-Sutcliffe Efficiency
NWS	National Weather Service
OrgN	Organic Nitrogen
OrgP	Organic Phosphorus
P	Phosphorous
PHOSKD	Phosphorus soil partitioning coefficient
PPERCO	Phosphorus percolation coefficient
R2	Coefficient of Determination
RHOQ	Algal respiration rate at 20ºC (day^{-1})
RS4	Rate coefficient for organic N settling in the reach at 20ºC (day^{-1})
RS5	Organic phosphorus settling rate in the reach at 20^{o}C (day^{-1})
RSDCO	Residue decomposition coefficient
SCS	Soil Conservation Service
SDNCO	Denitrification threshold water content (fraction of field capacity water content above which denitrification takes place
SLSUBBSN	Slope Length
SPCON	Linear parameter for estimating maximum amount of sediment that can

	be reentrained during channel sediment routing
SPEXP	Exponent parameter for estimating maximum amount of sediment that can be reentrained during channel sediment routing
SSURGO	Soil Survey Geographic
SWAT	Soil and Water Assessment Tool
TCEQ	Texas Commission on Environmental Quality
TIAER	Texas Institute for Applied Environmental Research
TMDL	Total Maximum Daily Load
TN	Total Nitrogen
TP	Total Phosphorus
TSS	Total Suspended Solids
TSSWCB	Texas State Soil and Water Conservation Board
USDA	U.S. Department of Agriculture
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
USLE	Universal Soil Loss Equation
WAF	Waste Application Field
WWTP	Wastewater Treatment Plant

Table A-1. A complete list of point source input data including dischargers and permitted limits for each watershed.

Watershed	Subbasin#	Permit No.	Facility Description	Discharge[1]		Organic Phosphorus	Soluble Phosphorus	Organic Nitrogen	Nitrate Nitrogen	Ammonia Nitrogen
				mgd	m³/day	kg/day (lb/day)				
Joe Pool	1	02427-000.001	ASH GROVE TEXAS LP	0	6.33	0 (0)	0.01 (0.02)	0.02 (0.04)	0.01 (0.02)	0.09 (0.20)
	4	04379-000.001	TXI OPERATIONS LP	1.29	4876.47	1.83 (4.03)	11.22 (24.74)	14.59 (32.17)	8.21 (18.10)	11.78 (26.0)
		10348-001.001; 002	TRINITY RIVER AUTHORITY OF TEXAS							
	6	14630-001.001	MOUSER FAMILY PARTNERSHIP #1	0.04	147	0.06 (0.13)	0.34 (0.75)	0.44 (1.0)	0.25 (0.55)	2.06 (4.54)
		13868-001.001	CREEK PARK CORP							
		14790-001.001	TXDOT							
		14101-001.001	ALVARADO ISD							
		13769-001.001	COUNTRY VISTA LTD							
Lavon	1	10502-001.001	CITY OF VAN ALSTYNE	0.95	3596.14	1.37 (3.01)	7.55 (16.64)	10.76 (23.72)	6.05 (13.34)	50.44 (111.19)
	3	10057-001.001	CITY OF TOM BEAN	0.15	567.81	0.22 (0.48)	1.19 (2.63)	1.70 (3.75)	0.96 (2.11)	7.96 (17.56)
	6	10704-001.001	CITY OF TRENTON	0.11	397.47	0.15 (0.33)	0.83 (1.84)	1.19 (2.62)	0.67 (1.47)	5.57 (12.29)
	7	10920-001.001	CITY OF LEONARD	0.80	3028.33	1.15 (2.54)	6.36 (14.01)	9.06 (19.98)	5.10 (11.24)	42.47 (93.63)
	8	11283-001.001	CITY OF ANNA	0.75	2839.06	1.08 (2.38)	5.96 (13.14)	8.49 (18.73)	4.78 (10.53)	39.82 (87.78)
	20	12446-001.001	NTMWD - WILSON CREEK	48.00	181699.68	13.63 (30.04)	77.22 (170.24)	543.65 (1198.52)	305.80 (674.17)	2548.34 (5618.07)
		12054-001.001	US ARMY COE - CADDO PARK	0.004	15.14	0.01 (0.01)	0.03 (0.07)	0.05 (0.10)	0.03 (0.06)	0.21 (0.47)
		12052-001.001	US ARMY COE - EAST FORK PARK	0.02	68.14	0.03 (0.06)	0.14 (0.32)	0.20 (0.45)	0.11 (0.25)	0.96 (2.11)
		12055-001.001	US ARMY COE - AVALON PARK	0.02	68.14	0.03 (0.06)	0.14 (0.32)	0.20 (0.45)	0.11 (0.25)	0.96 (2.11)
		12059-001.001	US ARMY COE - MALLARD PARK	0.01	45.42	0.02 (0.04)	0.10 (0.21)	0.14 (0.30)	0.08 (0.17)	0.64 (1.40)
		12061-001.001	US ARMY COE - LAVONIA PARK	0.02	64.35	0.02 (0.05)	0.14 (0.30)	0.19 (0.42)	0.11 (0.24)	0.90 (1.99)
		12051-001.001	COLLIN PARK MARINA INC	0.02	75.71	0.03 (0.06)	0.16 (0.35)	0.23 (0.50)	0.13 (0.28)	1.06 (2.34)
		12049-001.001	US ARMY COE - CLEAR LAKE PARK	0.01	34.07	0.01 (0.03)	0.07 (0.16)	0.10 (0.22)	0.06 (0.13)	0.48 (1.05)
		01923-000.001	CITY OF GARLAND STEAM ELECTRIC STATION	0.25	946.35	0.36 (0.79)	1.99 (4.38)	2.83 (6.24)	1.59 (3.51)	13.27 (29.26)
		11451-001.001	NTMWD - SEIS LAGOS	0.14	529.96	0.20 (0.44)	1.11 (2.45)	1.59 (3.50)	0.89 (1.97)	7.43 (16.39)
		14432-001.001	TIM BENNET ENG AND CONSTRUCTION	0.01	18.93	0.01 (0.02)	0.04 (0.09)	0.06 (0.12)	0.03 (0.07)	0.27 (0.59)
		12060-001.001	US ARMY COE - LAKELAND PARK	0.14	529.96	0.20 (0.44)	1.11 (2.45)	1.59 (3.50)	0.89 (1.97)	7.43 (16.39)
		10039-001	CITY OF BLUE RIDGE	0.23	851.72	0.32 (0.71)	1.79 (3.94)	2.55 (5.62)	1.43 (3.16)	11.95 (26.33)
		10442-001	CITY OF FARMERSVILLE	0.53	2006.27	0.76 (1.68)	4.21 (9.29)	6.00 (13.23)	3.38 (7.44)	28.14(62.03)
Lewisville	8	10569-001.001	CITY OF GUNTER	0.11	399	0.15 (0.33)	0.92 (2.03)	1.44 (3.17)	0.67 (1.48)	5.6 (12.35)
	11	14246-001.001	CITY OF CELINA	0.33	1231	0.46 (1.01)	2.83 (6.24)	4.43 (9.77)	2.07 (4.56)	17.26 (38.05)

	12	13647-001.001	CITY OF AUBREY	0.04	168	0.06 (0.13)	0.39 (0.86)	0.6 (1.32)	0.28 (0.62)	2.36 9 (5.20)
	13	14416-001.001	CITY OF DENTON	0.43	1625	0.61 (1.34)	3.74 (8.25)	5.85 (12.90)	2.73 (6.02)	22.79 (50.24)
		14372-001.001	CITY OF SANGER							
	17	10698-002.002	UPPER TRINITY REGIONAL WATER DISTRICT	0.32	1201	0.45 (0.99)	2.76 (6.08)	4.32 (9.5)	2.02 (4.45)	16.84 (37.13)
		10698-002.001	UPPER TRINITY REGIONAL WATER DISTRICT							
	18	10915-001.001	TOWN OF PROSPER	0.09	355	0.13 (0.29)	0.82 (1.81)	1.28 (2.82)	0.6 (1.32)	4.98 (11.0)
		14516-001.001	14875 PARTNERS LTD							
	19	10729-001.001	CITY OF KRUM	0.11	417	0.16 (0.35)	0.96 (2.12)	1.5 (3.31)	0.7 (1.54)	5.85 (12.90)
		14306-001.001	SLIDELL ISD							
	20	04336-000.001	SAFETY-KLEEN SYSTEMS INC.	0.77	2903	1.09 (2.40)	6.68 (14.73)	10.45 (23.04)	4.89 (10.78)	40.71 (89.75)
	23	14323-001.001	UPPER TRINITY REGIONAL WATER DISTRICT	0.09	340	0.13 (0.29)	0.78 (1.72)	1.22 (2.69)	0.57 (1.26)	4.77 (10.52)
	25	10027-003.001	CITY OF DENTON	12.62	47765	17.91 (39.48)	42.99 (94.78)	171.95 (379.08)	95.53 (210.61)	669.9 (1476.88)
	28	10172-002.001	CITY OF FRISCO	0.21	807	0.3 (0.66)	1.86 (4.10)	2.91 (6.42)	1.61 (3.55)	11.32 (24.96)
	30	14008-001.001	NORTH TEXAS MUNICIPAL WATER DISTRICT	3.41	12922	4.85 (10.69)	11.63 (25.64)	46.52 (102.56)	25.84 (57.0)	181.23 (399.54)
		02964-000.001	EXIDE TECHNOLOGIES							
	31	12605-002.001	BRIARWOOD LUTHERAN MINISTRIES	4.49	17003	6.38 (14.07)	15.3 (33.73)	61.21 (134.95)	34.01 (74.98)	238.47 (525.74)
		11570-001.001	CITY OF THE COLONY							
		03840-000.001	ACME BRICK CO.							
		13785-001.001	MARINE QUEST-HIDDEN COVE LP							
		10698-001.001	UPPER TRINITY REGIONAL WATER DISTRICT							
		10903-001.001	TOWN OF LAKEWOOOD VILLAGE							
		13434-001.001	TOWN OF HACKBERRY							
		11600-001.001	TOWN OF LITTLE ELM							
	32	10662-001.001	CITY OF LEWISVILLE	7.88	29842	11.19 (24.67)	26.86 (59.22)	107.43 (236.84)	59.68 (131.57)	418.53 (922.70)
Ray Roberts	1	14496-001.001	CITY OF SAINT JO	0.19	723	0.27 (0.60)	1.66 (3.66)	2.16 (4.76)	1.22 (2.69)	10.14 (22.35)
	2	10923-001.001	TOWN OF LINDSAY	0.15	551	0.21 (0.46)	1.27 (2.80)	1.65 (3.64)	0.93 (2.05)	7.73 (17.04)
	5	10341-001.001	CITY OF MUENSTER	0.24	925	0.35 (0.77)	2.13 (4.70)	2.77 (6.11)	1.56 (3.44)	12.97 (28.60)
	7	10726-001.001	CITY OF GAINESVILLE	1.45	5470	2.05 (4.52)	12.58 (27.73)	16.37 (36.09)	9.21 (20.30)	76.72 (169.14)
	10	11840-001.001	CITY OF CALLISBURG	0.03	109	0.04 (0.09)	0.25 (0.55)	0.33 (0.73)	0.18 (0.40)	1.53 (3.37)
	11	10151-001.001	CITY OF COLLINSVILLE	0.14	527	0.2 (0.44)	1.21 (2.67)	1.58 (3.48)	0.89 (1.96)	7.39 (16.29)

	19	13393-001.001	CALLISBURG ISD	0.03	113	0.04 (0.09)	0.26 (0.57)	0.34 (0.75)	0.19 (0.42)	1.58 (3.48)
		13514-001.001	TOWN OF OAK RIDGE							
	28	10361-001.001	CITY OF PILOT POINT	0.34	1284	0.48 (1.06)	2.95 (6.50)	3.84 (8.47)	2.16 (4.76)	18.01 (39.71)
		13199-001.001	CITY OF TIOGA							
Ray Hubbard	3	10363-001.001	NTMWD ROWLETT CREEK	24.00	90816.00	13.62 (30.02)	77.19 (170.14)	271.72 (599.04)	152.84 (336.96)	1273.69 (2807.99)
	11	10262-001.001	NTMWD - SQUABBLE CREEK	1.20	4542.00	1.70 (3.80)	9.54 (21.02)	13.59 (29.96)	7.64 (16.85)	63.70 (140.44)
	12	14216-001.001	NTMWD - MUDDY CREEK	5.00	18925.00	7.10 (15.85)	39.74 (87.57)	56.62 (124.83)	31.85 (70.22)	265.42 (585.15)
	17	10304-001	NTMWD - WYLIE	2.00	7570.00	2.84 (6.34)	15.90 (35.03)	22.65 (49.93)	12.74 (28.09)	106.17 (234.06)
Benbrook	10	10380-002.001	CITY OF WEATHERFORD	2.66	10078	3.8 (8.38)	21.4 (47.18)	30.2 (66.58)	17 (37.48)	14.1 (31.09)
	15	10847-001.001	CITY OF ALEDO	0.26	981	0.37 (0.82)	2.08 (4.59)	2.93 (6.46)	1.65 (3.64)	13.8 (30.42)
Bridgeport	51	10994-001.001	JACKSBORO RUNAWAY	0.43	1628	0.61 (1.35)	3.46 (7.63)	4.87 (10.73)	2.74 (6.04)	22.8 (50.27)
	53	10862-001.001	CITY OF RUNAWAY BAY	0.09	338	0.1 (0.22)	0.7 (1.5)	1 (2.2)	0.6 (1.3)	5 (11)
	54	10903-001 001	TOWN OF LAKEWOOD VILLAGE	0.04	136	0.1 (0.22)	0.3 (0.66)	0.4 (0.8)	0.2 (0.4)	2 (4.4)
Richland-Chambers	3	10567-001.001	CITY OF ALVARADO	0.18	671.93	0.26 (0.57)	1.41 (3.11)	2.01 (4.43)	1.13 (2.49)	9.05 (19.95)
	4	14411-001.001	BLUE WATER OAKS PROPERTY OWNERS	0.29	1103.49	0.42 (0.92)	2.32 (5.11)	3.3 (7.30)	1.85 (4.08)	0.29 (0.64)
	5	10379-001.001	CITY OF WAXAHACHIE	4.18	15804.89	6.01 (13.25)	33.19 (73.17)	47.26 (104.19)	26.55 (58.53)	6.18 (13.62)
	17	10431-001.001	CITY OF MAYPEARL	0.11	416.11	0.16 (0.35)	0.87 (1.92)	1.24 (2.73)	0.7 (1.54)	0.11 (0.24)
	27	13675-001.001	CITY OF BARDWELL	0.04	150.12	0.06 (0.13)	0.32 (0.71)	0.45 (0.99)	0.25 (0.55)	2.1 (4.63)
	31	10180-001.001	CITY OF GRANDVIEW	0.11	428.46	0.16 (0.35)	0.9 (1.98)	1.28 (2.82)	0.72 (1.59)	1 (2.20)
	39	13981-001.001	AVALON WATER SUPPLY & SEWER CORP	0.02	86.56	0.03 (0.07)	0.18 (0.40)	0.26 (0.57)	0.15 (0.33)	1.21 (2.67)
	44	10443-002.001	CITY OF ENNIS	1.86	7046.43	2.68 (5.91)	14.8 (32.63)	21.07 (46.45)	11.84 (26.10)	8.7 (19.18)
	46	11103-001.001	FORRESTON SEWER SERVICE & WSC	0	12.59	0 (0)	0.03 (0.07)	0.04 (0.09)	0.02 (0.04)	0.18 (0.40)
	75	10444-001.001	CITY OF FROST	0.02	58.95	0.02 (0.04)	0.12 (0.26)	0.18 (0.40)	0.1 (0.22)	0.83 (1.83)
	77	13937-001.001	CITY OF MILFORD	0.07	247.8	0.09 (0.20)	0.52 (1.15)	0.74 (1.63)	0.42 (0.93)	3.47 (7.65)
	92	10402-003.001	CITY OF CORSICANA	1.36	5160.38	1.96 (4.32)	10.84 (23.90)	15.43 (34.02)	8.67 (19.11)	1.5 (3.31)
	95	10402-003.002	CITY OF CORSICANA	0.25	928.29	0.35 (0.77)	1.95 (4.30)	2.78 (6.13)	1.56 (3.44)	1.09 (2.40)
	99	11606-001.001	CITY OF BLOOMING GROVE	0.04	157.9	0.06 (0.13)	0.33 (0.73)	0.47 (1.04)	0.27 (0.60)	2.21 (4.90)
	117	13528-001.001	BOSQUE UTILITIES CORP	0.01	39.5	0.02 (0.04)	0.08 (0.18)	0.12 (0.26)	0.07 (0.15)	0.55 (1.21)
	120	11864-001.001	CITY OF ANGUS	0.01	19.85	0.01 (0.02)	0.04 (0.08)	0.06 (0.13)	0.03 (0.07)	0.28 (0.62)
	140	11542-001.001	CITY OF BYNUM	0.02	65.81	0.03 (0.07)	0.14 (0.31)	0.2 (0.44)	0.11 (0.24)	0.92 (2.03)
	142	10514-001.001	CITY OF MALONE	0.02	74.95	0.03 (0.07)	0.16 (0.35)	0.22 (0.49)	0.13 (0.29)	1.05 (2.31)
	146	10026-001.001	CITY OF DAWSON	0.06	215.18	0.08 (0.18)	0.45 (0.99)	0.64 (1.41)	0.36 (0.79)	0.02 (0.04)

	154	10534-001.001	CITY OF HUBBARD	0.14	511.91	0.19 (0.42)	1.08 (2.38)	1.53 (3.37)	0.86 (1.90)	0.02 (0.04)
Cedar Creek[2]	67	TPDES 10143-001	CITY OF ATHENS	0.48	1808.65	2.08 (4.59)	3.08 (6.79)	2.43 (5.36)	19 (41.89)	3.75 (8.27)
	94	TPDES 11858-001	EAST CEDAR CREEK SPECIATION	0.22	832.74	1.05 (2.31)	3.14 (6.92)	9.02 (19.89)	3.11 (6.86)	31.49 (69.42)
	97	TPDES 13879-001	CHEROKEE SHORE SPECIATION	0.003	11.36	0.25 (0.55)	0.92 (2.03)	1.56 (3.44)	0.44 (0.97)	9.03 (19.91)
	87	TPDES 11132-001	CITY OF EUSTACE	0.05	171.58	0.28 (0.62)	0.56 (1.23)	2.19 (4.83)	0.08 (0.18)	1.96 (4.32)
	37	TPDES 12114-001	CITY OF KAUFMAN	0.72	2712.09	3.11 (6.86)	4.46 (9.83)	3.91 (8.62)	29.04 (64.02)	4.4 (9.7)
	53	TPDES 10695-001	CITY OF KEMP	0.09	342.28	0.15 (0.33)	0.95 (2.09)	0.69 (1.52)	3.74 (8.25)	0.14 (0.31)
	84	TPDES 10579-001	CITY OF MABANK	0.18	664.53	0.36 (0.79)	2.3 (5.07)	2.87 (6.33)	4.35 (9.59)	0.86 (1.9)
	13	TPDES 10747-001	CIT OF TERRELL	3.15	11923.12	2.42 (5.34)	48.77 (107.52)	40.68 (89.68)	139.38 (307.28)	53.48 (117.9)
	17	TPDES 10623-001	CITY OF WILLS POINT	0.367	1350.20	1.37 (3.02)	2.58 (5.69)	9.49 (20.92)	2.36 (5.20)	4.47 (9.85)
Eagle Mountain[2]	79	TPDES 10131-001	CITY OF BOYD	0.08	316.82	0.07 (0.15)	0.42 (0.93)	0.4 (0.88)	1.28 (2.82)	0.41 (0.9)
	54	TPDES 10009-001	CITY OF DECATUR	0.66	2492.68	1.39 (3.06)	4.3 (9.48)	3.92 (8.64)	21.23 (46.8)	0.98 (2.16)
	105	TPDES 10649-001	CITY OF SPRINGTOWN	0.2	794.09	0.69 (1.52)	1.06 (2.34)	1.21 (2.67)	5.09 (11.22)	0.89 (1.96)
	4	TPDES 10071-001	CITY OF BOWIE	0.54	2026.5	1.34 (2.95)	5.33 (11.75)	1.98 (4.37)	41.38 (91.23)	0.08 (0.18)
	136	TPDES 11626-001	CITY OF NEWARK	0.04	149.41	0.23 (0.51)	0.31 (0.68)	0.3 (0.66)	0.09 (0.2)	1.38 (3.04)
	148	TPDES 11123-001	FORTH WORTH BOAT CLUB	0.01	43.32	0.04 (0.09)	0.06 (0.13)	0.02 (0.04)	0.36 (0.79)	0.01 (0.02)
	95	TPDES 10701-002	CITY OF RHOME	0.04	166.23	0.14 (0.31)	0.2 (0.44)	0.2 (0.44)	0.65 (1.43)	0.71 (1.57)
	123	TPDES 11183-003	CITY OF AZLE	0.75	2827	0.29 (0.64)	1.2 (2.65)	3.01 (6.64)	4.6 (10.14)	2.72 (6)
	44	TPDES 10389-002	CITY OF BRIDGEPORT	0.58	2198.62	1.15 (2.54)	3.31 (7.3)	20.66 (45.55)	2.76 (6.08)	0.48 (1.06)
	113	TPDES 12909-001	LARRY R. BUCK - SAGINAW	0.01	10.72	0.01 (0.02)	0.01 (0.02)	0.01 (0.02)	0.09 (0.2)	0 (0)
	58	TPDES13427-001	PARADISE ISD	0.01	60.56	0.05 (0.11)	0.08 (0.18)	0.03 (0.07)	0.49 (1.08)	0.02 (0.04)
	35	TPDES 10023-001	CITY OF CHICO	0.06	215.75	0.34 (0.75)	0.72 (1.59)	2.53 (5.58)	0.1 (0.22)	2.7 (5.95)
	34	TPDES 14339-001	CITY OF ALVORD	0.05	179.48	0.15 (0.33)	0.24 (0.53)	0.28 (0.62)	1.14 (2.51)	0.21 (0.46)
Assumed concentrations (mg/L) [3]						0.38 0.075[4] ; 0.15[4]	2.1 0.425[4] ; 0.85[4]	2.99	1.68	14.03

[1] The wastewater treatment plant (WWTP) discharge rates are the permitted discharge rates. The permit limit is not the actual amount.

[2] For Cedar Creek and Eagle Mountain, the actual wastewater effluent concentrations and flows were used.

[3] While local data were not available, assumed concentrations were derived from a comprehensive survey of municipal wastewater dischargers in the Virginia portion of the Chesapeake Bay Basin (Wiedeman and Cosgrove, 1998).

[4] The soluble phosphorus concentration of 0.425 mg/L and organic phosphorus concentration of 0.075 used for NTMWD - Wilson Creek in Lavon Lake watershed were based on the total phosphorus (TP) permit limit of 0.5 mg/L. for that plant. For NTMWD Rowlett Creek in the Ray Hubbard watershed, the soluble phosphorus concentration of 0.85 mg/L and organic phosphorus concentration of 0.15 were based on local data.