DISSERTATION

DEVELOPMENT OF A MODEL FOR HYDRO-SALINITY SIMULATION IN IRRIGATED RIVER BASINS

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WE HEREBY RECOMMEND THAT THE DISSERTATION PREPARED UNDER OUR SUPERVISION BY YUAN-WEN LIN ENTITLED DEVELOPMENT OF A MODEL FOR HYDRO-SALINITY SIMULATION IN IRRIGATED RIVER BASINS BE ACCEPTED AS FULFILLING IN PART REQUIREMENTS FOR THE DEGREE OF DOCTOR OF PHILOSOPHY.

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ABSTRACT OF DISSERTATION

DEVELOPMENT OF A MODEL FOR HYDRO-SALINITY SIMULATION IN IRRIGATED RIVER BASINS

Agricultural irrigation has led to river salinization in many basins. When irrigation water filtrates through the soil, salt dissolves in the water, and the saline water enters the groundwater and returns to the river.

In this study a model is developed to simulate hydro-chemical processes in soil and spatial/temporal distribution of irrigation return flows so that river salinity can be predicted. Major chemical reactions in the root zone soil are simulated including precipitation, dissolution of soil minerals and association and dissociation of major ion pair species. To determine the quantity and quality of the deep percolated water, soil water composition is predicted at five soil depths: surface, ¼, ½, ¾ and bottom of the root zone.

The spatial and temporal distribution of groundwater return flows to the river system are expressed as response functions multiplied by the volume of the stress applied on the basin. The response functions are based on two-dimensional porous media flow equations. A probabilistic approach is developed to simulate travel time of groundwater flow for irrigated fields that have subsurface drainage devices.

All related hydrological components are integrated into the computation of river water quantity and quality. These components include: groundwater return flows, irrigation tail water, tributary inflows, river diversions, phreatophyte consumption, river channel losses and river depletion due to pumping. The river system is divided into reaches, and the river quantity and quality can be simulated from upstream reaches to downstream reaches sequentially.

The study area selected for implementing this hydro-salinity simulation is the Arkansas River Basin in Colorado. The simulation period is from January 1986 to December 2001, a total of 192 months. The first 60 months (1986 to 1990) are used to calibrate the model. The other 132 months (1991 to 2001) are used to validate the calibrated model. A GIS program, ArcView 3.2, is applied to generate a groundwater table so that the response functions for groundwater return flows in each canal service area can be developed. The simulation results show that irrigation return flows, including surface and groundwater return flows, significantly increase river quantity in reaches 12, 13, 14, 21, 22, 23 and 24. Meanwhile, groundwater return flow is a prime driver of river salinity. Both river quantity and river salinity show significant seasonal fluctuation. While the results presented in this dissertation attempt to represent the actual conditions in the Arkansas River Basin no external funding was provided for this study and therefore in some instances the data used is a best estimate of the actual conditions.

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CHAPTER 1

INTRODUCTION

1.1 Problem Description

Agricultural irrigation has led to substantial problems in many river basins. The problems include waterlogging, soil salinization, and saline irrigation return flows (Chiew et al., 1992; Richardson and Narayan, 1995; Rhoades, et al., 1997; Gates, et al., 2002). Waterlogging and soil salinization can impair soil productivity and reduce crop yield. Irrigation return flows, with associated high salinity loads, degrade river quality and contribute to salinization of irrigated lands downstream. Ghassemi et al. (1995) estimated worldwide crop yield loss due to salinity at about $10 billion/year. The National Research Council, 1996, estimated that 27% of the land in the western United States has a moderate to severe potential for salinity problems.

During the irrigation season, water is diverted from river systems to agricultural areas to meet crop growth need. As crop uptakes water, it results in chemical reactions in the soil-water interface since crops consume nearly pure water and leave salts in the soil which can change the chemical balance in the soil (Tanji, 1990). Soil water salinity can increase significantly due to chemical reactions. Excess soil water moves beyond the root zone and recharges the alluvium. In river basins the alluvial aquifer is hydraulically connected with the river system, therefore groundwater can discharge to downstream river reaches. The quantity and quality of groundwater return flows to the river system influences river salinity.

Distribution of groundwater return flows to a river system can be predicted by simulating groundwater flow movement in the alluvial aquifer. One method for doing this involves applying differential equations describing the nonsteady groundwater flow in a nonhomogeneous anisotropic aquifer to the alluvial aquifer being studied. Then, the aquifer can be discretized into cells, and the equations can be solved for each cell using the finite difference or finite element method. By calculating the amount of water received by the river, groundwater return flow can be predicted (Konikow and Bredehoeft, 1974; Person and Konikow, 1986). This approach requires detailed information regarding hydrological parameters and boundary conditions for the study area. The requirements for such large amounts of data usually restrict the use of this method.

Another approach for simulating groundwater flow is to apply steady state flow equations to an isotropic aquifer assumed to have several other ideal conditions such as semi-infinite aerial extent and full penetration by wells and the river. The equation is then solved analytically and "stress response functions" are generated based on this analytical solution (Glover and Balmer, 1954; Jenkins, 1968). This approach can be applied to a basin-wide study area for a long study period. However, if the assumed ideal conditions in the aquifer are far from the real conditions, the generated response functions will be inaccurate.

In this study, a new methodology is proposed that simulates groundwater flow in an irrigated area without having to assume ideal aquifer conditions. The response functions used for simulating spatial and temporal distributions of the return flow are generated based on actual groundwater flow paths and travel time in the aquifer. In addition, this new methodology involves soil chemical reactions to simulate the quantity and quality of soil water recharged to the alluvial aquifer.

The lower Arkansas River Basin in Colorado will be used as the study area for demonstrating the river basin hydro-salinity simulation. This basin consists of irrigated lands totaling 113,987 hectares (based on 1998 data). Due to irrigation return flow, average river salinity (based on records from 1961 to 2002) increases from 0.52 dS/m[1] at Pueblo to 3.55 dS/m at the Kansas state line, a sevenfold increase over a distance of 250-km. Intensive irrigation in the area has caused the groundwater table to rise. As early as 1955, the U.S. Bureau of Reclamation found that 96,815 acres (about 30% of the total irrigated land) had high water table problems (Miles, 1977). A recent field survey (Gates, et al., 2002) also confirmed high groundwater table problems in Otero County, one of the counties that comprise the lower Arkansas River Basin. These salinity problems indicate the need to construct a salinity simulation model to evaluate possible solutions to the salinity problems of the Arkansas Valley.

Recent modeling work in the lower Arkansas River Basin has been done by Dai (1996), Dai and Labadie (2001), Gates et al. (2002), and Triana et al. (2003). Dai (1996) and Dai and Labadie (2003) developed a program, MODSIMQ, which integrates a water rights simulation model, MODSIM, with a river quality simulation model, QUAL2E. MODSIMQ applies the Frank-Wolfe algorithm to make MODSIM simulate river diversion under the constraints of a river quality setting determined in QUAL2E. The

[1] Units for measuring the electrical conductivity of the solution in 10^{-1} siemens per meter (dS/m)

model is applied to the Arkansas River Basin from Nov, 1988 to Oct, 1989, a 12 month period. In the application, a stream depletion factor (SDF) method is used to simulate the distribution of irrigation return flow, and a regression method is used to simulate salinity of the groundwater return flow in the Arkansas River Basin. Gates et al. (2002) reported on a study they are conducting in the irrigated area encompassing western Otero County to western Bent County. The study includes 74 observation wells for collecting weekly data pertaining to water table elevation and salinity. Their results indicated that about 1/3 of the wells were dry in 1999, meaning that groundwater table elevations in these dry well locations were deeper than 6m. During the summer the wells were sampled weekly and the data were fitted into certain types of probability distributions. For example, the data collected in the 1st, 2nd and 3rd week in June 1999 was fitted into Rayleigh, Beta and Lognormal distributions respectively. The surveyed data was also used to set up a steady-state groundwater model, which includes MODFLOW for groundwater flow simulation and MT3DMS for groundwater quality simulation. The goal of the research is to develop a transient groundwater model based on the extensive data collected over three consecutive years. Triana et al. (2003) used an artificial neural network to show the relationship between measurable aquifer stresses and return flow in a test area near La Junta.

One important improvement that this study presents over the previous research is the simulation of deep percolation in irrigated areas. This study integrates the calculation of crop consumptive use with soil water leaching fraction and chemical reactions to obtain the volume and salinity of the water percolating under the crop root zone. In the previous research (Gates et al., 2002), leaching fraction is assumed to be a constant value, 0.70, and chemical reactions taking place in the soil / water interface are neglected. Since leaching fraction is dependent on the amount of crop consumptive use and the amount of irrigation water applied, variables which vary from month to month, leaching fraction should also be time variant instead of a constant value. Based on the values of the leaching fraction, chemical reactions in the soil occur which result in changes in the soil water composition. Therefore, simulations of chemical reactions in the soil water are important for predicting the salinity of deep percolated water.

The methodology used for generating response functions for groundwater return flow in this study improves upon methodologies used in previous research. In this study, a model is designed that can simulate groundwater flow paths and travel time based on the water table and physical characteristics of the river basin. The model solves Darcy's two dimensional porous media flow equation numerically by using groundwater table elevation, hydraulic conductivity and aquifer porosity as parameters. In the research of Dai (1996) and Dai and Labadie (2001), the generation of response functions is based on the SDF methodology. Since several ideal conditions which are assumed in the SDF methodology are not valid for the Arkansas River Basin, such as an aquifer with a semi-infinite aerial extent and streams fully penetrating the aquifer, the generated response functions may contain unwanted bias. Furthermore, the SDF method does not allow water table elevation information to be used in generating response functions. Water table elevation information is an essential factor in forming groundwater flow paths. This study uses the water table elevation data from 965 wells (Major et al., 1970). These data have been compared with some recent field survey data (Appendix C) to make sure the numbers are suitable for the modeling work. A water table was interpolated and response function parameters for groundwater return flow were calculated.

1.2 Objective of the Research

The objective of this study is to develop a model that can simulate water quantity and quality in an irrigated river basin. The advanced features targeted in this study can be summarized as follows:

(1) Soil water composition simulation

The effect of the crop consumptive use on soil water composition is simulated. The composition of soil water is obtained by calculating the chemical balance of mineral dissolution / precipitation and the ion pair association / dissociation in water in the root zone. The composition includes concentrations of Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, Cl^{-}, SO_4^{2-}, HCO_3^{-} and CO_3^{2-}. The simulated minerals include calcite, magnesite and gypsum. A steady state program developed by Rhoades (1990) is modified to be able to handle transient state simulation in monthly time steps. By simulating soil water composition, the salinity and the volume of the irrigation water which has deep percolated can be obtained

(2) Irrigation return flow simulation

Response functions are generated for irrigation return flows caused by deep percolation, tail water and canal leakage. The generation of response functions for groundwater return flow is based on simulation of flow paths and travel time of the groundwater using a two-dimensional, steady state, and Darcy's flow. The governing equations are applied to the aquifer. The aquifer is discretized into rectangular cells, and a GIS program is applied to construct the groundwater table surface. A probabilistic approach is developed that can simulate groundwater travel time for the irrigated fields that have subsurface drainage. Computer codes are designed for all the simulation processes. Using response functions, spatial and temporal distribution of irrigation return flow can be addressed.

(3) River quantity and quality simulation

All related hydrological components are integrated into the computation of river quantity and quality. These components include: irrigation groundwater return flows, tributary inflows, river diversions, phreatophyte consumption, river channel losses and river depletion due to pumping. The river system is divided into reaches, and for a given period, river water quantity and quality can be simulated sequentially from upstream reaches to downstream reaches.

(4) User-friendly graphical interface

A graphical user interface (GUI) is designed for the hydro-salinity simulation model. The GUI is written using C/C++ to take advantage of its object oriented programming feature (Schildt, 1998a). The GUI provides a diagrammatic illustration of the river basin network, dialog-based data input, and graphical output display. The GUI also allows users to compare the simulation results for different river reaches and for different time periods.

(5) Application to the Arkansas River Basin in Colorado

The model developed in this study is applied to the Arkansas River Basin. The simulated river salinity and river flow is compared with historical records. The model is used for multiple year simulations using data from 1986 to 2001 (a total of 192 months) for the Arkansas River Basin in Colorado. The model is calibrated with data from the first 60 months. Then, the calibrated model is validated using data from the remaining 132 months.

Groundwater return flow is a major process simulated with this model because of its impact on river quantity and quality. As opposed to the ongoing research of Gates et al. (2002), which will simulate spatial and temporal fluctuations of the groundwater table in individual irrigated fields, this study is a basin scale simulation, in which groundwater flow is dominated by the global flow system (Freeze and Cherry, 1979).

1.3 Summary of the Chapters

In the remaining chapters the following topics will be discussed:

Chapter Two: Literature Review

Literature relevant to this study is reviewed. Topics include the nature of agricultural salinity, control measures for salinity, the aquifer recharge process, the groundwater return flow process, the application of GIS to river basin water quantity/quality simulation, and hydro-salinity simulation and control in the Arkansas River Basin.

Chapter Three: Methodology for the River Basin Salinity Simulation

The methods used for simulating soil water chemical reactions are described and applied to predict the quantity and quality of soil water recharging into an alluvial aquifer. Then, response functions are applied to determine how the recharging water returns to the river system. Response functions are also used to determine the influence of pumping, surface runoff, canal leakage, and reservoir leakage on river salinity.

Chapter Four: Irrigation Return Flows Simulation

This chapter illustrates the methodology for deriving response functions for irrigation return flow. A probabilistic approach is developed to calculate groundwater travel time in an irrigated area. The river basin is discretized into cells. Spreadsheets are used as an interface for data and parameter input.

Chapter Five: River Salinity Model Calibration

The hydro-salinity of the Arkansas River Basin is described. A GIS program is applied to generate a groundwater table surface. Response functions for irrigation return flows are simulated. The hydro-salinity model is calibrated against data from 1986 to1990 and then validated with data from 1991 to 1996.

Chapter Six: Implementation of the Calibrated Model

The calibrated model is used to simulate the quantity and quality of water in the Arkansas River in Colorado as well as the soil water composition in the 24 canal service areas in the river basin from 1997 to 2001. The sensitivities of four selected parameters are analyzed to evaluate their influence on the simulation output. The four selected parameters are river diversion amount, hydraulic conductivity, drainage portion and tail water ratio.

Chapter Seven: Conclusions and Recommendations

Conclusions and recommendations for future research are presented.

CHAPTER 2

LITERATURE REVIEW

Literature relating to this study can be divided into six categories: (1) agricultural salinity problems, (2) measures for salinity control, (3) simulation of aquifer recharge processes, (4) simulation of groundwater return flows, (5) application of GIS in water quantity / quality modeling, and (f) research related to the Arkansas River Basin. These topics are examined in this chapter.

2.1 Agricultural Salinity Problems

Salinity is the concentration of dissolved ions, such as Ca^{++}, Mg^{++}, Na^{+}, K^{+}, Cl^{-}, SO_4^{--}, HCO_3^{-}, and NO_3^{-}. The two most common expressions of salinity are electrical conductivity (EC) and total dissolved solids (TDS). Sodium adsorption ratio (SAR) and cation exchange capacity (CEC) are also frequently used to indicate the portion of major ions in salt-affected soil. Salinity problems are major issues in irrigated agriculture because they result in crop yield reduction. Most crops can only survive within a small range of salinity levels. This is because the dissolved ions increase osmotic pressure requiring a plant's roots to expend more energy to acquire water, thus damaging plant growth. The Food and Agriculture Organization of the United Nations (FAO, 1976) gives a list of tolerance levels for different crops based on the EC of saturated soil paste extract within the root zone.

Most salts in an irrigated system originate from mineral weathering and are then transported by water throughout the hydro-system. Salts have three major phases: solid phase, solution phase, and exchange phase. The changes among the three phases are dominated by various chemical reactions such as dissolution, precipitation, absorption, desorption, and ion association in the soil-water interface (Smedema, 1983). Figure 2.1 briefly describes the interchanges among the phases and shows the salts most commonly found in the agricultural environment.

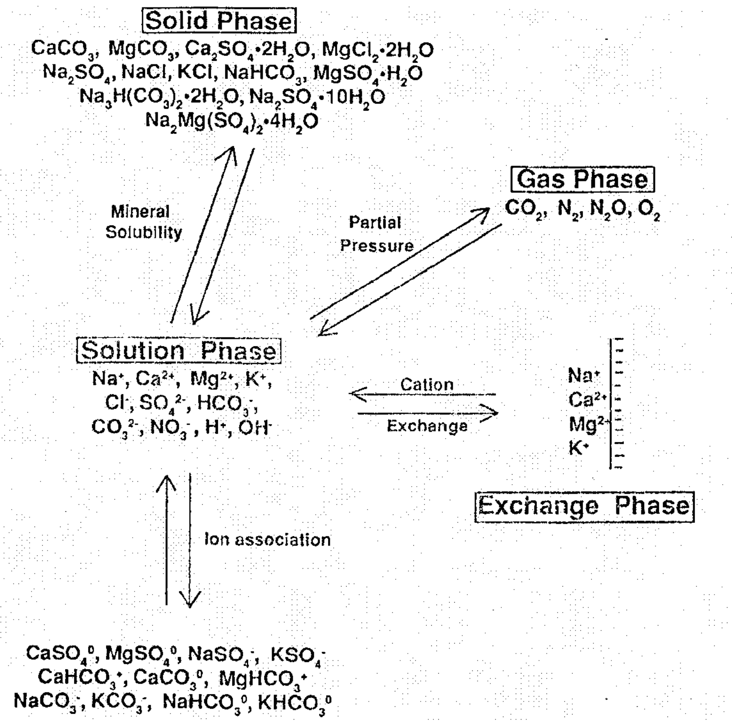


Figure 2.1: Chemical Reactions in Different Soil-Water Phases (from Tanji, 1990)

Irrigation activity can result in increased salinity levels in soils and groundwater. This can be explained by applying a salt balance equation for root-zone soil developed by Smedema (1983):

$$I \cdot C_i + R \cdot C_r + G \cdot C_g = P \cdot C_p + \Delta S \quad (2.1)$$

where I is irrigation water entering the root-zone; R is rainfall entering the root-zone; G is capillary flow from groundwater into the root-zone; P is deep percolation from the root-zone; C is salt concentration of the water; ΔS is change in salt content of the soil solution in the root-zone; with subscripts *i, r, g*, and *p* referring to irrigation, rainfall, capillary and deep percolation respectively.

As indicated in equation (2.1), when salt input from irrigation water and capillary flow exceeds salt output by leaching, salts (ΔS) accumulate in the soil which over time can cause soil to become saline. Saline soils can load large amounts of salts into the leaching water. As leaching water recharges alluvial aquifers, groundwater becomes increasingly salty.

A survey conducted by the USDA (1988) used total dissolved solid (TDS) and Cl^- in surface soil as parameters for investigating the degree of soil salinization present within the 48 contiguous United States. The survey indicated that 30% of this land has the potential to develop saline soils. Many other agricultural areas in the world, such as India, Australia, and the Middle East, also suffer crop yield reductions due to salt problems. The U.S. Salinity Laboratory collected soil and water samples from salt-affected areas around the world (Tanji, 1990). The results of chemical analyses showed that the average salinity level in soil, groundwater and river water are 12.9 dS/m (saturated paste), 2.4 dS/m and 1.4 dS/m respectively. The high level of salinity in the soil samples reflects the influence of irrigation activity on agricultural land.

Irrigation combined with the insufficient natural drainage abilities of aquifers can lead to high groundwater table levels. In many areas around the world, water tables rose from 20-30m to 1-2m below the ground surface only 10 to15 years after an irrigation project started (Smedema, 1983). High groundwater tables aggravate soil salinization by bringing salts upward through capillary flow.

Irrigation activity can create local groundwater flow systems that are totally different from the regional groundwater flow system (Freeze and Cherry, 1979). Over an irrigated area, fields frequently irrigated have higher water table levels than other fields. These frequently irrigated fields create groundwater flows toward neighboring fields as the water follows in the direction of the hydraulic gradient. This can result in seepage problems and create conflicts between landowners (Radosevich, 1978). Figure 2.2 is an illustration of the difference between a local flow system and a regional flow system.

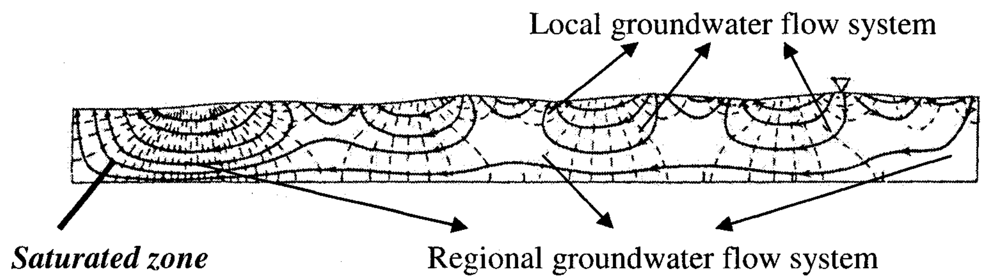


Figure 2.2: Local Groundwater Flow System Created by Irrigation Activity (from Freeze and Cherry, 1979).

2.2 Measures for Salinity Control

Salinity control measures can be categorized by the scale of irrigated land to which they are applied. Figure 2.3 summarizes the differences between small-scale (on-farm level) and large-scale (basin level) salinity control measures.

	On-farm level, small scale ◄- - - -	- - - - ► **Basin level, large scale**
Major concerns	Soil salinity; Groundwater table; Seepage	Total crop production; Downstream salinity damage; Total economic efficiency; Water conservation
Control Measures	Improve irrigation practice; Install drainage system; Line ditches	Optimization by programming tools; System benefit/cost analysis; Improve water delivery system; Water augmentation to meet salinity criteria
Rules and Laws	Water right priorities; Sewage discharge standard	Salinity policy; Intra-state and intra-national treaties
Groups, Authorities Involved	Farmers; Local agencies	State government; Federal government

Figure 2.3: Differences between Salinity Control Measures

On-farm measures are focused on improving farming practices to maintain the salinity in the root zone at a level suitable for crop growth. Some frequently-used practices include land leveling for improving leaching efficiency and minimizing tail water; changing furrow directions to reduce unwanted percolation that can raise the groundwater table; and selecting crop varieties and hybrids that are tolerant of higher salinity concentrations.

Drainage is another frequently used on-farm control measure, especially for irrigated fields already suffering from serious seepage and waterlogging problems. El-Mowelhi et al. (1988) conducted an experiment to test the effect of drainage on soil salinity in the Nile River basin. One year after instituting drainage practices, the results showed that the soil salinity at a depth of 1.5 m was reduced from 5.3 to 2.2 dS/m on average. In this experiment, although no extra water was applied except normal irrigation water and rainfall, crop yield increased by 50-100%. The cost of installing a drainage system depends on the number of drains required and how deep the drains are buried, which is determined by soil texture, salinity of groundwater, and crop type (Tanji, 1990). The simplest drainage type with a cost that is affordable for farmers is the interceptor drain (or mole drainage). It consists of a main pipe installed perpendicular to the flow direction of seepage to catch and remove shallow subsurface flow.

In the San Joaquin Valley, Grismer et al. (1988) combined irrigation practices with drainage operations in order to develop strategies for minimizing waterlogging and the upward flow of salts to the root zone. Their results demonstrated that salinity control can be achieved by frequent water applications.

The Red River Plain Area of Texas once had serious saline-seep problems that damaged crop production (USDA, 1983). After installing subsurface drainage systems, these problems have effectively disappeared.

Basin-scale salinity control is more complicated than small-scale control. Since more water users are involved, there are more regulations and laws with which to comply. The larger areas also mean that there are more complicated physical systems and processes to model. Therefore, the benefits and costs of a basin-scale project must be carefully evaluated. For example, lining canals to prevent leakage is helpful in controlling canal transit loss. However, due to high construction costs, canals are seldom lined. Likewise, the use of pumping wells to control groundwater tables is also limited. High operation and maintenance costs as well as the regulations imposed by legal statutes in some areas, mean that this practice is only applied to specific regions.

Generally, the objectives of a basin scale salinity control plan must be designed to have positive impacts on both a region's economy and on the environment. Tanji (1990) listed some popular objectives such as: (a) improving surface and ground-water quality by reducing irrigation water salinity; (b) reducing the need to develop additional water supply structures by conserving water, including structures that often adversely affect fish and wildlife; (c) reducing salinity damage in water downstream; (d) increasing the overall economic efficiency of agricultural production; (e) reducing saline drainage to groundwater; and (f) increasing crop production by reducing soil salinity.

In situations where river water salinity exceeds criteria at some locations or in segments of the entire river basin, fresh water augmentation is an ideal non-structural measure to solve the problem. McKay et al. (1998) proposed a project to meet water quality objectives in the Truckee River in Nevada. The proposal suggested purchasing 24,000 acre-ft of water from upstream agricultural water rights holders for river augmentation. By doing this, saline river water would be diluted and groundwater return flows and the associated solute loading would be reduced.

Basin scale projects may simultaneously adopt several measures to control salinity problems. An example is the salinity reduction project in the Colorado River Basin at Moapa Valley, Nevada (Soil Conservation Service, 1993). This project includes laser-controlled land leveling for irrigated fields, installation of a pipeline system for delivery of irrigation water, and a training program for farmers. This plan proposed to reduce salt loading to the river by 18,700 tons /year. Estimated costs were $10.35 million. Estimated benefits included increasing water use efficiency from 55% to 66%, reducing downstream salt by 1.9 milligram per liter of salt concentration at the Imperial Dam near

Yuma, increasing surface flow to Lake Mead by 5,500 acre-feet, and reducing groundwater flow by 4,700 acre-ft annually.

2.3 Simulating Aquifer Recharge in an Irrigated Area

In irrigated areas, water leaches through the soil profile and recharges underlying aquifers. The timing of the recharge is determined by the infiltration process, and the chemical composition of the recharging water is determined by chemical reactions in the soil. Literature for simulating the aquifer recharge processes is described in the following sections.

2.3.1 Quantity of Aquifer Recharge

The amount and distribution of aquifer recharge depends on the size of the area and time period of interest. Using a small time-scale such as minutes or hours, aquifer recharge can be derived from a physically-based infiltration model that considers soil suction force and change in soil moisture. There are three frequently used infiltration models: Horton's equation, Phillip's equation, and the Green-Ampt method. They all relate to the one-dimensional form of the Richard's equation that was derived from the continuity equation developed for unsteady, unsaturated flows in porous media (Chow et al., 1988). However, these equations contain the assumption that there is continuous ponding on top of the soil surface. For water that is intermittently applied to the ground surface, Morel-Seytoux and Miracapillo (1984) derived an approximate unit hydrograph model by considering physical factors of gravity and viscosity. With the information concerning the evapotranspiration in the soil root zone and the amount of water input to the ground, a time series for aquifer recharge can be obtained.

At a large time-scale (months or years) over a large irrigated region, calibration for physically-based infiltration models is difficult. Increased spatial variations in the soil properties complicate the model and greater amounts of input data are required. With reliable evapotranspiration and applied water data, Konikow and Bredehoeft (1974) determined the monthly aquifer recharge in a 30-square-mile region using the following equation:

$$\frac{dR}{dA} = (A/E)^i \qquad \text{for } A \le E \tag{2.2}$$

where R is the total recharge, A is the total applied water (irrigation plus precipitation), E is the potential evapotranspiration; i is a recharge parameter greater than zero. Gupta and Paudyal (1988) incorporated equation (2.2) with a one-dimensional groundwater flow equation to produce the following equation:

$$\frac{\partial h}{\partial x} = \frac{S}{T} \cdot \frac{\partial h}{\partial t} - \frac{R(x,t)}{T} \tag{2.3}$$

They used a non-linear regression technique to match historical groundwater level records by adjusting i, S, and T. Where h is groundwater level, S is storage coefficient, T is transmissivity, and t is time. This methodology was applied to two locations in the Tinao Basin aquifer. The results show that the groundwater level data generated by this estimated recharge model is consistent with historical data over the entire 72-month period of comparison.

Chiew et al. (1992) integrated a rainfall-runoff model, HYDROLOG, with a finite element groundwater model, AQUIFEM-N, to estimate groundwater recharge in north-central Victoria, Australia. The model was calibrated against stream flow and groundwater level data, in both irrigated and non-irrigated regions. This model was also used to study different management practices for salinity control. Villalobos de Alba et al. (1996) assumed an aquifer's physical parameters and initial conditions to develop a grid-type model to simulate net flow in the unsaturated zone and aquifer recharge. Groundwater level data were used to correct the net flow in each grid to reduce the simulation error. Villalobos de Alba's et al. model was applied to the San Juan del Rio Aquifer in Mexico and was calibrated over a 10-year period with water table data.

Groundwater recharge can be predicted from tracer elements. By comparing the concentration of a tracer in irrigation water, soil solute, and groundwater, the amount of recharge water can be obtained. Chloride is an ideal tracer since it has high dissolution, which makes it easy to remove by leaching water. Chloride is frequently used for calculating leaching fractions and the relative proportions of water from different sources that enter soil water in irrigated systems (Tanji, 1990). Wood and Sanford (1995) used a chloride mass-balance method to evaluate recharge flux to the High Plains Aquifer underlying the semiarid Southern High Plains of Texas and New Mexico. Their results suggested that the recharge flux was approximately 2% of precipitation, or approximately 11 plus or minus 2 mm/y. Wood and Stanford also found that their results were consistent with previous estimates done using physically-based measurements. Ting et al. (1998) also applied the chloride mass-balance method to estimate groundwater recharge in the Pingtung Plain in Taiwan. Their results indicated that groundwater recharge is 15% of the annual rainfall when annual rainfall in Taiwan is approximately 100 inches. Stone (1984) estimated recharge to the Ogallala-Aquifer in New Mexico.

2.3.2 Quality of Aquifer Recharge

As water infiltrates downward, its chemical composition is subject to change due to physical and chemical processes in the unsaturated zone. There are many discussions on the topic of solute transport in soil (Filep, 1999; Tindall et al., 1999; Selker et al., 1999; Parlange and Hopmans, 1999). Basically, the soil environment and the chemical reactions that occur in it are so complicated that to be modeled they must be simplified. For example, the precipitation-dissolution process of calcite and gypsum might be the only processes modeled for salt affected soils since they are the two most important reactions, while other minerals only making minimal contributions to the system can be neglected (Tanji, 1990). Dutt (1962a, 1962b) developed a model that considered the ion exchange between the calcium ion (Ca^{++}) and the magnesium ion (Mg^{++}) on the soil mineral surface and the dissolution of gypsum ($CaSO_4$). Dutt's model considers the unsaturated zone as a multi-layer system with water quality in the upper layer used as an initial value for the lower layer. Input data required is the initial concentration of Ca^{++}/Mg^{++}, ionic strength (based on the total amount of ions in a solution), and an activity coefficient obtained from the Debye-Huckel theory (Filep, 1999). However, Dutt's model does not consider the dissolution of calcite, which is one of the major reactions in the unsaturated zone. Furthermore, evapotranspiration by plant roots, an important process in an irrigated area, is omitted. There are other models and theories developed for solving the ion exchange process in soils, such as Van der Molen (1956) and Glueckauf (1954).

Rhoades (1987) developed a one-dimensional model, WATSUIT, to assess water quality within the root zone. This model assumes 40% of plant water uptake is from the upper quarter of the root zone, 30% from the second quarter, 20% from the third quarter and 10% from the bottom quarter. Plant water uptake results in the concentration of the soil solute and thus shifts the chemical reactions. The data required by this model for input are concentrations of the major ions Ca^{++}, Mg^{++}, Na^{+}, K^{+}, alkalinity, Cl^{-}, and SO_4^{2+} in the irrigation water. Dissolution / precipitation and ion association / dissociation are simulated. Rhoades' model predicts the composition of the soil water, EC, sodium adsorption ratio, and osmotic potential at five soil depths (the surface of the soil, ¼, ½, ¾ and the bottom of the root zone). However, when using soil solute in the root zone to estimate the quality of the aquifer recharge water, the fact that the depth of the unsaturated zone may exceed or be less than the depth of the root zone needs to be considered and some adjustments may need to be made. Both Dutt's model and Rhoades' model only consider essential chemical processes. This reduces the amount of required input data and parameters and makes the models relatively easy to use.

Simunek and Suarez (1993) developed a finite element model, UNSATCHEM-2D, for modeling major ion equilibrium and kinetic non-equilibrium chemistry in unsaturated and saturated porous media. The governing flow equation is a modified version of the Richards' equation, and the governing chemical transport equation is a two-dimensional advective-dispersive equation, shown below:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial x_i}\left[K\left(K_{ij}^{A}\frac{\partial h}{\partial x_j} + K^{A}\right)\right] - Sink \qquad (2.4)$$

$$\frac{\partial \theta c_k}{\partial t} + \rho\frac{\partial \bar{c}_k}{\partial t} + \rho\frac{\partial \hat{c}_k}{\partial t} = \frac{\partial}{\partial x_i}\left[\theta D_{ij}\frac{\partial c_k}{\partial x_j} - q_i c_k\right] \qquad k = 1,2,\ldots,N_c \qquad (2.5)$$

where θ is water content, h is pressure head, *Sink* is a sink term, x_i (i=1, 2) are spatial coordinates, t is time, K_{ij}^A are components of a dimensionless anisotropy tensor K^A, and K is the unsaturated hydraulic conductivity function. c_k is the total dissolved concentration of the aqueous component k, $\overline{c}_k$ is the total adsorbed concentration of the aqueous component k in the minerals which can precipitate or dissolve, ρ is the bulk density of the medium, D_{ij} is the dispersion coefficient tensor, q_i is the volumetric flux and N_c is the number of aqueous components.

UNSATCHEM-2D predicts the concentration of major ions (Ca^{++}, Mg^{++}, Na^+, K^+, SO_4^{--}, Cl^-, NO_3^-, H_4SiO_4), alkalinity and CO_2. The physical and chemical processes considered in this model are: water uptake by plant roots; solute flux in both the saturated and unsaturated zones; complexation; cation exchange; and precipitation / dissolution of major minerals such as calcite, dolomite, gypsum, hydromagnesite, nesquehonite, and sepolite. Since UNSATCHEM-2D is a fully dynamic model, it requires information about soil parameters, boundary conditions, and input/output data in order to calibrate the model. There are other transient-state physical-based solute transport models, such as LEACHM by Wagenet and Hutson (1987), CSUID by Garcia et al. (1995), and HYDRUS-2D by Simunek et al. (1999).

There are uncertainty factors in applying soil solute models. For instance, the Darcian flow assumption may not be applied to water flow in an irrigated field since there are usually macropores in the soil zone that make water move rapidly by gravity (Tindall et al., 1999). Another factor is that equilibrium of chemical reactions may not be attained. For example, dolomite ($CaMg(CO_3)_2$) has approximately the same molar solubility as calcite ($CaCO_3$) but its dissolution rate is 70 times slower. This creates bias in the prediction of soil solute since most models are based on the assumption that equilibrium will be achieved. This assumption is extremely weak when water flux is high (Tanji, 1990).

2.4 Simulating Groundwater Return Flow in a Stream-Aquifer System

The quantity and quality of groundwater return flows depend on the process of groundwater moving through the saturated aquifer. The velocity and direction of groundwater flows are decided by the attributes of the aquifer, such as porosity, transmissivity, storage coefficient, homogeneity, and the geomorphology of bedrock (McWhorter and Sunada, 1977). The solute transport in an aquifer is decided by the geologic formation and processes of advection, dispersion, and chemical reactions.

2.4.1 Quantity of Groundwater Return Flow

Groundwater return flows in a river basin can be simulated by methodologies having different levels of complexity, from lumped parameter models to simplified steady-state distributed models to fully dynamic numerical models.

One method to determine groundwater return flows is by using statistical analysis on historical stream flows and aquifer storage data. In many river systems, stream flow is fed by outflow from shallow groundwater aquifers. Therefore, there is a close relationship between river discharge, Q, and aquifer storage, S. Wittenberg (1999a, and 1999b) used a nonlinear equation to describe it as follows:

$$S = aQ^b \tag{2.6}$$

where a and b are parameters that can be obtained by calibration. This model has been applied to a semi-arid catchment in Australia. The advantage of this statistical approach is that no information about aquifer parameters is required. However, it cannot handle watersheds that are used intensely for agriculture, where there are activities such as well pumping or irrigation recharge.

Under several idealized assumptions, a stream depletion model can be used to simulate the effects on streams of a hydraulic stress, such as pumping or recharge, place on an aquifer (Glover and Balmer, 1954). The major assumptions in stream depletion models include: a homogeneous, isotropic aquifer with semi-infinite aerial extent; a steady state groundwater flow with constant transmissivity in space and time; pumping wells and streams that fully penetrate the aquifer; and a perfect hydraulic connection between the streambed and the aquifer (Jenkins, 1968; Sophocleous et al., 1995). The effect on the stream by a point stress can be expressed as:

$$\frac{q}{Q} = erfc\left(\frac{a}{\sqrt{4tT/S}}\right) \tag{2.7}$$

where q is the rate of the effect of the stress on the stream [L^3/T]; Q is the average recharge / discharge rate from the stress [L^3/T]; a is the distance from the stream to the stress point [L]; T is the aquifer transmissivity [L^2/T]; t is time since the stress started [T], $0 < t < t_p$, t_p is the total time period the stress is applied; S is the specific yield of the aquifer [dimensionless]; *erfc* is the complementary error function.

Equation (2.7) is valid for any time in the period of t_p. After t_p, the stress ceases but the effect of the stress will continue to be felt in the stream-aquifer system until hydraulic equilibrium is achieved. The rate, q, at any time after t_p can be calculated from the method of superposition by introducing an imaginary well in the stress point (Jenkins, 1968; McWhorter and Sunada, 1977).

The stream depletion model allows the effect of each point stress to be calculated independently. This provides rapid evaluation of the changing availability of water in a system. Moulder and Jenkins (1969) applied a stream depletion model to simulate the effect on streams from numerous irrigated recharge sources and pumping wells in an agricultural area. They wanted to predict the availability of surface water at successive diversion points downstream, and to optimize operation plans of stream-aquifer systems by deciding when, where, and how much to pump or recharge.

Burns (1988) used equation (2.7) to develop monthly-lagged unit-response functions that can determine the percentage of return flows in each month. This is achieved by integrating equation (2.7) against *t* to obtain the volume of return flows and then dividing it by the total volume of the applied water at the stress point. Therefore, whenever a stress is applied on the groundwater reservoir, its return flow can be calculated by multiplying it with the appropriate unit-response function.

In addition to developing time-lagged unit response functions for calculating groundwater return flows, the percentage of return flows to different segments of the river can also be derived. For a recharge source located at (a, 0) and a stream located at y-distance, the groundwater return for a finite reach from –y to +y can be described as equation (2.8) (Glover, 1974):

$$\frac{q_{|y|}}{Q} = \frac{2}{\pi}\int_0^x \frac{ae^{-z^2}}{y^2 + a^2}\,dy \qquad (2.8)$$

where $z^2 = (y^2 + a^2)S/4Tt$. Again, if equation (2.8) is integrated against *t*, discretized, and divided by the total volume of recharge water, the time-lagged and spatially-varied unit-response function, c_{lt}, can be obtained. Where *l* represents the number of reaches in the river that return flow discharges to, t represents time lags for the return flow, and $\sum_l \sum_t c_{lt} = 1$.

The application of a stream depletion model to an aquifer is limited by the suitability of its various assumptions. Since it is difficult to fulfill all the assumptions, the evaluation of their relative importance becomes crucial. Sophocleous et al. (1995) analyzed the relative importance of the assumptions, showing that the three most important assumptions that influence the prediction of groundwater return flows are: streambed clogging, degree of stream partial penetration, and aquifer heterogeneity. This study also conducted a comparison between the stream flow depletion model and a numerical groundwater transport model, MODFLOW (McDonald and Harbaugh, 1988), in the accuracy of predicting groundwater return.

Without assuming idealized hydrogeologic conditions, the two-dimensional transient groundwater flow in a heterogeneous anisotropic unconfined aquifer can be described by equation (2.9) (Konikow and Bredehoeft, 1974).

$$\frac{\partial}{\partial x_i}\left(T_{ij}\frac{\partial h}{\partial x_j}\right) = S\frac{\partial h}{\partial t} + W(x,y,t) \qquad i,j=1,2 \tag{2.9}$$

where T_{ij} is the transmissivity tensor (L^2/T); h is the hydraulic head (L); S is the storage coefficient (dimensionless); t is time (T); and $W(x,y,t)$ is the recharge volume flux per unit area (L/T).

Charbeneau (2000) has stated the suitability of using a two-dimensional model instead of a three-dimensional model for groundwater flow in an unconfined aquifer if the aquifer has much greater lateral extent (kilometers) than thickness (meters or tens of meters).

Equation (2.9) needs to be solved by numerical methods, that is, the aquifer needs to be subdivided into a large number of relatively small cells and finite difference equations or finite element equations need to be used to approximate equation (2.9). By using numerical methods, spatial variability in the aquifer can be simulated by assigning appropriate physical parameters (T_{ij} and S) to each cell. Based on the Dupuit's theory (Bear, 1972), h equals the elevation of the groundwater table, which is always fluctuating in most agricultural areas. With proper information of W, h, T_{ij} and S, the transport of groundwater among the cells can be calculated. The solution of equation (2.9) is the contours of the groundwater table. By using these contours, the amount of return flow can be calculated.

Konikow and Bredehoeft (1974) applied equation (2.9) to simulate groundwater transport and stream-aquifer interaction in a 30 square-mile study area. The authors calibrated the model based on information collected during a one-year period, including groundwater level data from sixty-three wells, estimated groundwater recharge in irrigated fields, and volume of water pumped. The results show that the predicted groundwater table is within 1 ft of the observed values approximately 90% of the time. Lefkoff and Gorelick (1987) developed a computer program, AQMAN, which can generate response functions for each stress point based on equation (9). By superimposing various response functions, the effects of different stresses on any location in the stream-aquifer system are obtained. This study also incorporated linear and quadratic objective functions to optimize the groundwater pumping and recharge.

MODFLOW is a groundwater flow program that solves a governing equation similar to equation (9) using a finite difference method. Fredericks et al. (1998) used a modified version of MODFLOW, MODRSP, to generate spatially-varied and monthly-lagged response functions for 196 wells and 32 recharge sites in the South Platte River Basin. Their results showed that the total amount of return flow in a 84-month simulation period was twice that calculated by using the aforementioned stream flow depletion model. The return flows calculated by MODRSP also have larger seasonal variation than those calculated with the stream flow depletion model. By comparing the historical net sum of recharge and pumping, MODRSP was considered to be more dependable than the stream depletion model (Fredericks et al., 1998).

ADMIN96 is a computer program which also uses spatially-varied and monthly-lagged response functions to calculate groundwater returns resulting from irrigation return flows and canal leakage. However, this program does not show the governing equations upon which these response functions are based.

2.4.2 Quality of Groundwater Return Flow

The type of approach used for simulating the composition of groundwater return flows depends on the availability of information about the river basin. If the information is limited, statistical-type or black-box models are the models of choice. However, if the information about the physical and geo-chemical properties over the entire aquifer is sufficient, a physical type solute transport model can be developed.

Herczeg et al. (1993) used chemical characteristics of surface water to verify the quality of groundwater return flows. Over an eight-year period, surface water samples were collected at thirty-five sites around a 1.06×10^6 km^2 river basin in southeastern Australia. Since this is a semi-arid basin (annual rainfall ranged from 0.25 to 1.2 m/year) with 80% of its river water used for irrigation, the changes in the chemical composition of surface water directly reflected the quality of the groundwater return flows. The results showed that over a 2,500 km distance, Cl^- increased from 1.9 mg/l to 171 mg/l and Na^+ increased from 3.2 mg/l upstream to 101 mg/l downstream. This significant increase is due to the sea-derived salts contained in the groundwater. The increase of HCO_3^-, Ca^{++}, Mg^{++} and SO_4^{--} in surface water is due to rock-derived salts in groundwater.

Miles (1977) developed a salt budget model to calculate the amount of "salt" returned to the Arkansas River system. The model first calculated the amount of groundwater return flow in each river segment by using flow rates and diversion records, then the amount of salt influx was calculated by using the concentration of TDS and ions recorded at the gaging stations. Miles (1977) found that some river segments had significant salt input while others did not. Miles attributed the results to the non-homogeneity in the occurrence of salt dissolution / precipitation reactions in the aquifer. A similar study was also conducted by Guymon et al. (1992).

Herczeg's and Miles' models are useful for predicting the quality of groundwater return flows for large river basins. However, these lumped-type models do not simulate spatial changes in groundwater quality. With the assistance of well data, Lefkoff and Gorelick (1990a) developed a regression model to predict changes in groundwater salinity for each irrigated field in the Arkansas River Basin. Changes in groundwater salinity were simulated as functions of hydrologic conditions and water use decisions. The data required for this regression model were the amount of water pumped and the amount of recharge; initial conditions of groundwater salinity and river salinity, and river stage and canal stage. The results showed that the simulated salinity values for 1982 were reasonable when compared with the observed data.

A physically-based solute transport model can simulate the changes of groundwater quality spatially and temporally. Therefore, the chemical composition of groundwater return flows can be obtained. Equation (2.10) (Javendel et al., 1984) is a three-dimensional equation that simulates advection, dispersion, sink/source and chemical reactions in groundwater:

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial C}{\partial x_j}\right) - \frac{\partial}{\partial x_i}(v_i C) + \frac{q_s}{\theta}C_s + \sum_{k=1}^{N} R_k \qquad i, j = 1,2,3 \qquad (2.10)$$

where C is the concentration of the substance dissolved in groundwater (M/L^3); t is time (T); x_i is the distance along the respective Cartesian coordinate axis (L); D_{ij} is the hydrodynamic dispersion coefficient (L^2/T); v_i is the seepage or linear pore water velocity (L/T); q_s is the volumetric flux of water per unit volume of aquifer representing sources (positive) and sinks (negative) (1/T); C_s is the concentration of the sources or sinks (M/L^3); θ is the porosity of the porous medium (dimensionless); and $\sum_{k=1}^{N} R_k$ is a chemical reaction term (M/L^3/T).

Equation (2.10) needs to be solved numerically instead of analytically due to its complexity. There are computer programs developed based on this equation such as MT3D (Zheng; 1990), UNSATCHEM-2D (Simunek and Suarez, 1993), and FLOTRAN (Holder et al., 2000). These programs were developed to be used conjunctively with other groundwater flow models (such as MODFLOW) that can provide volumetric flux of water between numerical cells.

Konikow and Bredehoeft (1974) simplified equation (2.10) and used it to simulate groundwater salinity in a stream-aquifer system. The simplification includes the reduction of the flow dimension from 3-D to 2-D and neglects chemical reactions. Since information about effective porosity and longitudinal dispersivity of the porous medium is lacking when using this simplified equation, trial and error is used to decide the most suitable values. Best results occurred when porosity equaled 20% and longitudinal dispersivity which equaled 100 ft.

2.5 Application of GIS in Water Quantity/Quality Modeling

Geographical information systems (GIS) are effective tools for storing, managing, and displaying spatial data (Tsihrintzis et al., 1996). GIS has mainly been used in watershed modeling for two purposes: (1) to provide accurate spatial discretization for the execution of cell-by-cell type transport models; (2) to serve as a powerful graphical interface for integrating models of different scales and for input/ output control.

Corwin et al. (1999) developed a GIS-linked one-dimensional vadose-zone solute transport model, TETrans. This model can predict and visually display salt loading in groundwater in the San Joaquin Valley in California. To handle the variability of soil salinity in their study, a field survey was conducted by using an EM-38 soil salinity meter in 2,368 locations over the 2,396 *ha* study area. Other required input data and parameters included: (a) quantity and quality of irrigation water; (b) crop ET; (c) the maximum root penetration depth for each crop type; (d) thickness and density of each soil layer; and (e) initial condition of water content and soil solution in each layer. The researchers found that the predicted salt loading simulated by TETrans was close to the measured salt loading in the study area.

Richards and Kump (1997) tested the utility of a GIS approach to watershed mass balance problems. The GIS database, including watershed divide, lithology and land use, was constructed to evaluate the effects of chemical weathering in the Susquehanna River

Basin in Pennsylvania. Historical water quality data from 52 stations retrieved from STORET were associated to derive equations for calculating annual magnesium flux. The results showed that the prediction error is within 10% of the true annual flux.

Constructing the hydraulic head surface (HHS) over a floodplain aquifer is an essential step in modeling groundwater flows. Salama et al. (1996) constructed a HHS in a 2.3 km^2 area in southwestern Australia by using the relationship between reduced water levels and surface topography prepared from a GIS digital elevation model (DEM) and the relationship between the depths of groundwater and hydrogeomorphic units (prepared from GIS land use data). Their results revealed that the GIS approach produced more realistic HHS than manual contouring methods and geostatistical methods. Manual contouring and geostatistical methods have problems with groundwater contours intersecting surface contours and the inability to identify groundwater discharge areas.

GIS can be used as an interface to control the input / output for different models. Vermulst and De Lange (1999) integrated a small scale unsaturated flow model and a regional scale saturated groundwater flow model by using a GIS-interface. The GIS-interface computed the flow transport between an upper aquifer and a lower aquifer based on a single database. The methodology was successfully applied in the Netherlands (approximately 32,400 km^2) to calculate seepage fluxes (upward and downward), and changes in the groundwater table. GIS was also used as an interface to control the input / output for different models by Ross and Tara (1993). They integrated a groundwater model (MODFLOW), a surface-water model (HSPF), an evapotranspiration model and a GIS program (SPAN) under different spatial discretizations.

2.6 Research Related to the Arkansas River Basin

There is considerable amount of research related to the Arkansas River Basin in Colorado. Reviewing these articles helps in understanding the characteristics of this specific basin and provides guidelines for further study. These articles can be divided into four categories:

(1) Stream-aquifer system simulation

As previously mentioned, Konikow and Bredehoeft (1974) constructed a numerical simulation model for groundwater flow and solute transport in an 11-mile reach near La Junta. Later on, this study was improved by adding a two-month time lag for salt transport in the unsaturated zone and by re-examining the reliability of some input data (Person and Konikow, 1986). In an application of the simplified stream depletion model, Jenkins and Taylor (1972) developed stream depletion factor contours for the entire lower Arkansas River in Colorado to facilitate the estimation of changes in stream flows caused by recharge to or withdrawal of groundwater from the aquifer. Moulder and Jenkins (1969) combined the stream depletion model with a water-management plan for surface water and groundwater conjunctive use.

Watts and Lindner-Lunsford (1992) used a finite-difference model, MODFLOW, to simulate the fluctuations of the groundwater table over an irrigated area of approximately 1.5 miles$\times$5 miles near La Junta from 1960 to 1984. Their sensitivity analysis showed that the groundwater table is highly sensitive to the altitudes of the riverbed, pumping rates, conductance of the beds of the Arkansas River and Fort Lyon Canal, aerial recharge rates, and evapotranspiration, but are only moderately sensitive to hydraulic conductivity values.

Instead of focusing on fluctuations of the groundwater table in a small area, Burns (1988, 1989) developed a network model to simulate water quantity and quality in the Arkansas River from Pueblo to the Kansas state line. The effects of irrigation, canal leakage and groundwater pumping on return flows were calculated by stream depletion functions. Quantity and quality of incremental stream flow was determined by regression equations with information pertaining to hydrological data, overland runoff, return flows, diversions and specific conductance. This model could also simulate water supply operations, including: direct diversions, groundwater pumping, reservoir releases, trans-mountain deliveries and storage diversions. The model was calibrated against historic Arkansas River Basin hydrological and operational records using monthly time steps for the period of 1940 to 1985.

Gates et al. (2002) installed 74 observation wells to monitor water table depths and EC of groundwater. In the survey of 1999, about 24 wells out of the 74 wells were dry, which indicated that the water table depths in these well locations are deeper than 6 m. In the rest of the wells, water table depth ranged from 0.08m to 5.08 m. The surveyed data were used to calibrate a groundwater model, MODFLOW in a 65,300 acre irrigation area from western Otero County to western Bent County. The computed water table elevation was compared with observed water table data at 63 points. Based on the ground water flow simulated by MODFLOW, a ground water quality model, MT3DMS, was applied to simulate contaminate transport. The simulation results indicate that improvements in certain areas can be expected from vertical drainage derived from increased pumping or from decreased recharge bought about by reducing over- irrigation.

Triana, et al. (2003) developed relationships between measurable aquifer stresses and river return flow using an artificial neural network (ANN). The study area was adjacent to the Arkansas River between Crooked Arroyo and Timpas Creek. The ANN was integrated with GIS and a groundwater model developed by Gates et al. (2002).

(2) Impact of agriculture on the environment

Hearne et al. (1987) studied the effects of agriculture on surface and groundwater quality. Sayer et al. (1997) studied canal seepage problems in the Catlin Canal and provided alternative solutions. Sayer et al. estimated that about 25% of water was lost in transit due to seepage. Konikow and Person (1985) assessed long-term salinity changes in groundwater. In their study, several methods of testing hypotheses (including K-S test, Wilcoxon rank test, Mann-Whitney U test and t test) were applied to compare the differences between 1972 data and 1982 data. Their results showed that salinity in the irrigated area near La Junta has reached dynamic equilibrium in response to irrigation practices. Goff et al. (1998) evaluated the effects of different irrigation practices on the quantity and quality of water. They found that decreases in groundwater irrigation from pumping or surface irrigation can result in decreases in groundwater salinity and river salinity but has an insignificant influence on aquifer water levels.

(3) Basin operation, economic and legal aspects

Abbott (1985) described information that is essential for managing the Arkansas River Basin, including (a) the spatial distribution of the irrigation canals, major tributaries, reservoirs and water districts, (b) major water right holders and their decree dates, and (c) aspects of the Arkansas River Compact that dictate the operation of John Martin Reservoir. Radosevich (1978) discussed legal aspects of irrigation return flows as well as the distribution and administration of water in the western United States. Radosevich concluded that although river salinity problems caused by irrigation return flows are controlled by the Water Pollution Control Act, practical implementations still need to be enforced by feasible water management technologies. Boyle Engineering Corporation (1990) studied water budgets in streams, soils and aquifers in order to describe the historic movements of water in, within, and out of selected reaches of the Arkansas River for compact years 1940-1985.

(4) Salinity management

Miles (1977) listed six salinity management alternatives and analyzed their feasibility. His results indicated that the only feasible alternative is to improve irrigation efficiency to reduce return flows and salt pickup. Other management alternatives such as dilution of return flows, disposing of return flows and abandonment of saline irrigated lands are either economically or legally impossible. Watts and Lindner-Lunsford (1992) proposed five alternatives to lowering groundwater tables and used MODFLOW to evaluate their impacts on the stream-aquifer system. The results showed: (a) deepening the Arkansas River will cause decreases of groundwater head in most areas of the aquifer; (b) lining a major canal will cause decrease of groundwater head near the canal but has almost no influence in groundwater head near the river; (c) increasing pumping (from municipal wells or irrigation wells) or installing subsurface drainage will cause localized decreases in groundwater head and increased leakage from the river. Valliant (1999) demonstrated the effect of using polyacrylamides for reducing canal seepage. Two kinds of polyacrylamide were tested: one (HYDROGEL) was used to coat the ditch before the irrigation season while the other (PAM) was used to reduce soil erosion. The results of experiments in a dirt ditch showed the reduction of seepage to be about 45%, making this a promising technique for reducing seepage.

Dai (1996) and Dai and Labadie (2001) developed a program, MODSIMQ, which integrated river salinity management with river quantity and quality simulations. Major features of this program include: (a) a water right simulation model, MODSIM (Labadie, 1987), which simulates the monthly river diversion amount for each canal; (b) a river quality simulation model, QUAL2E, which can simulate up to 15 organic and inorganic quality constituents in the river; (c) simulation of spatial and temporal distribution of ground water return flow and pumping using the stream depletion factor (SDF) method; (d) simulation of the salinity of ground water return flow using a regression method; (e) interaction of MODSIM and QUAL2E using a non-linear programming method, the Frank-Wolfe algorithm. The algorithm allows MODSIM to simulate river diversions under the constraint of a river quality setting in QUAL2E. This work was done using one year of data (Nov, 1988 to Oct, 1989) and their results show that when using a river water quality constraint the total amount of diversions during the irrigation season would decrease.

CHAPTER 3

METHODOLOGY FOR RIVER BASIN SALINITY SIMULATION

As discussed in the previous two chapters, irrigated agriculture has a substantial influence on river basin salinity. In this chapter a model is described which simulates soil and river salinity in irrigated river basins. The model development involves three steps: (a) simulation of the chemical reactions that occur in the soil water, (b) simulation of the soil water composition in irrigated fields with the chemical reaction mechanism, and (c) simulation of irrigation return flows with integration of soil water salinity in the river basin. A C++ program with a graphical user interface was developed for the simulation and provides a user-friendly interface.

3.1 Soil Water Chemical Reaction Mechanism

This section will illustrate major chemical reactions which occur in the soil water and how plants cause salinity changes in the soil water.

3.1.1 Concept and Assumptions

When irrigation water is applied to a field, it filtrates downward and fills the soil pores. A portion of the water is consumed by crops. Since crops consume nearly pure water, dissolved salts remain in the soil water. As a result, concentrations of cations and anions in the soil water increase. This process occurs on irrigated land and is usually referred to as the concentrating effect (Miles, 1977). To predict changes of ion concentrations, it is necessary to simulate chemical reactions in the soil water and to build a soil profile chemical composition model based on the reactions.

There are numerous chemical reactions between solid, solution and gaseous phases in soils. Since this study is focused on salinity problems, only reactions that influence soil salinity are simulated. Therefore, chemical reactions such as dissolution of organic compounds are not simulated. In this study, the simulated chemical reactions are dissolution of carbon dioxide gas, precipitation / dissolution of major soil minerals, and association / dissociation of ion pairs. Primary species involved in these chemical reactions are CO_2, $CaCO_3$, $MgCO_3$, $CaSO_4$, Na^+, Ca^{2+}, Mg^{2+}, K^+, Cl^-, SO_4^{2-}, HCO_3^-, CO_3^{2-}, H^+, OH^-, $CaSO_4^0$, $MgSO_4^0$, $NaSO_4^-$, KSO_4^-, $CaHCO_3^+$, $MgHCO_3^+$, $NaHCO_3^0$, $CaCO_3^0$, $MgCO_3^0$, and $NaCO_3^-$. Figure 3.1 illustrates how these species are transformed during the three phases (gaseous, solid, and solution) in soil. As shown in the figure, dissolution of CO_2 determines the concentration of HCO_3^- and CO_3^{2-} in the solution phase, hence impacting the solubility of $CaCO_3$, $MgCO_3$, and the balance of other species. Concentrations of the dissolved ions are based on chemical equilibrium laws which will be described in more detail in the next section.

There are more chemical species in the soil solution than are shown in figure 3.1. These species are not addressed by this model because their contribution to salinity is negligible in comparison to those shown in figure 3.1. For example, calcium related minerals including $Ca(HCO_3)_2$ (aragonite), $CaMg(CO_3)_2$ (dolomite), $Ca_5(PO_4)_3OH$ (hydroxyapatite) and $Ca_5(PO_4)_3F$ (fluoroapatite) are not simulated in this model. Their impacts on salinity are negligible when compared to the impacts of $CaCO_3$ and $CaSO_4$ in soil.

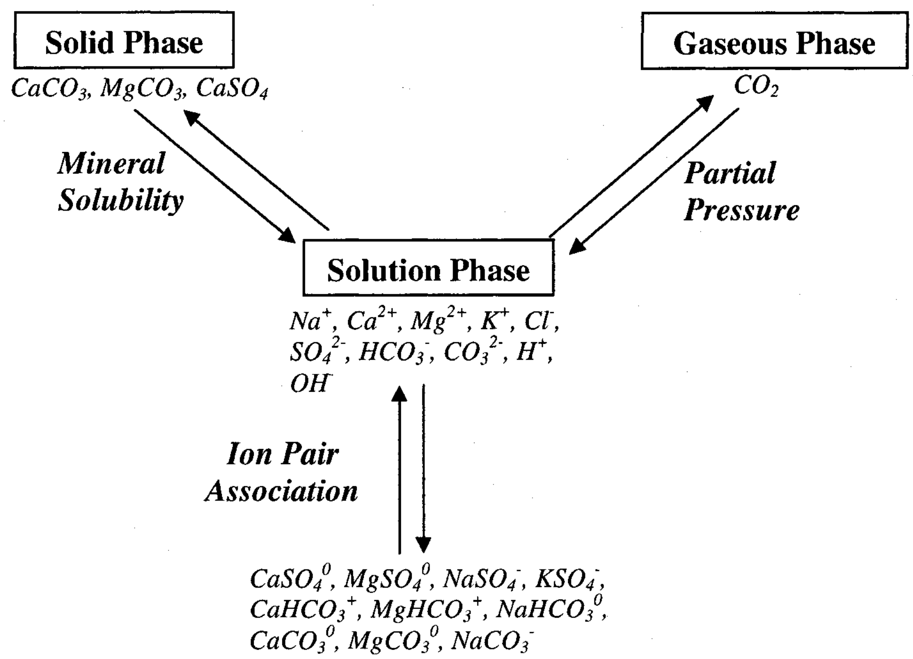


Figure 3.1: Phase Changes of Major Chemical Species in Soil

This study assumes a steady state of chemical composition in the soil, meaning that the concentration of chemical species is assumed to not change over time. A steady state is assumed instead of a dynamic state because this model is designed to predict seasonal salinity changes in a river basin. The steady state model uses "months" as the simulation time step. On the other hand, with smaller time steps, such as hours or days, a dynamic model is usually used to simulate concentrations of chemical species (Simunek, 1993). The assumption of a steady state implies that the cation adsorption effect on soil colloid can be ignored since they won't affect soil salinity (Tanji, 1990) and the chemical composition of soil water only changes with soil depth.

Chemical composition of soil water is determined by the crops' consumptive use at different depths. To illustrate this phenomenon, let's divide the root zone into *n* layers from top to bottom (i.e. $1^{st}, 2^{nd}, \ldots$ to n^{th} layer). Mass balance in the i^{th} layer can be expressed as:

$$V_{in}(i) = V_{out}(i) + V_{crop}(i) \quad (3.1)$$

where $V_{in}(i)$ represents volume of input water from layer $i\text{-}1$ to layer i; $V_{out}(i)$ represents the volume of output water from layer i to layer $i+1$; $V_{crop}(i)$ represents the volume of water lost to crop consumptive use in the i^{th} layer. Output from the i^{th} layer becomes input for the $i+1^{th}$ layer (i.e. $V_{out}(i) = V_{in}(i+1)$)

If $V_{crop}(i)$, $V_{in}(i)$ and chemical composition of $V_{in}(i)$ are known, composition of soil water in the bottom of the i^{th} layer can be simulated from the top layer to the bottom layer with the chemical reactions shown in figure 3.1.

In this study, a four layer cascading model was developed to predict the composition of the soil water in the plant root zone. This type of four-layer model was first introduced by Oster and Rhoades (1974) for constructing an irrigation water suitability model (WATSUIT). Figure 3.2 illustrates this model diagrammatically. As shown in the figure, a total $X\%$ of the irrigated water is consumed by plants. The rest of the irrigated water ($1\text{-}X\%$) then percolates below the root zone into the saturated aquifer. $1\text{-}X\%$ is usually referred to as the leaching fraction (Filep, 1999). Due to the fact that the plant's upper root zone uptakes more water than its lower root zone does, distribution of the plant's water consumption over the four root zone layers is assumed to be 40%, 30%, 20% and 10% respectively for the 1^{st} (top), 2^{nd}, 3^{rd} and the 4^{th} (bottom) layers. Based on this distribution, the concentration of chemical species is simulated from the top soil layer to the bottom soil layer serially. The simulated concentration of chemical species is expected to be higher in the deeper soil layers because of the concentrating effect of plants. After the soil water passes through the root zone, its composition is assumed to have no further changes.

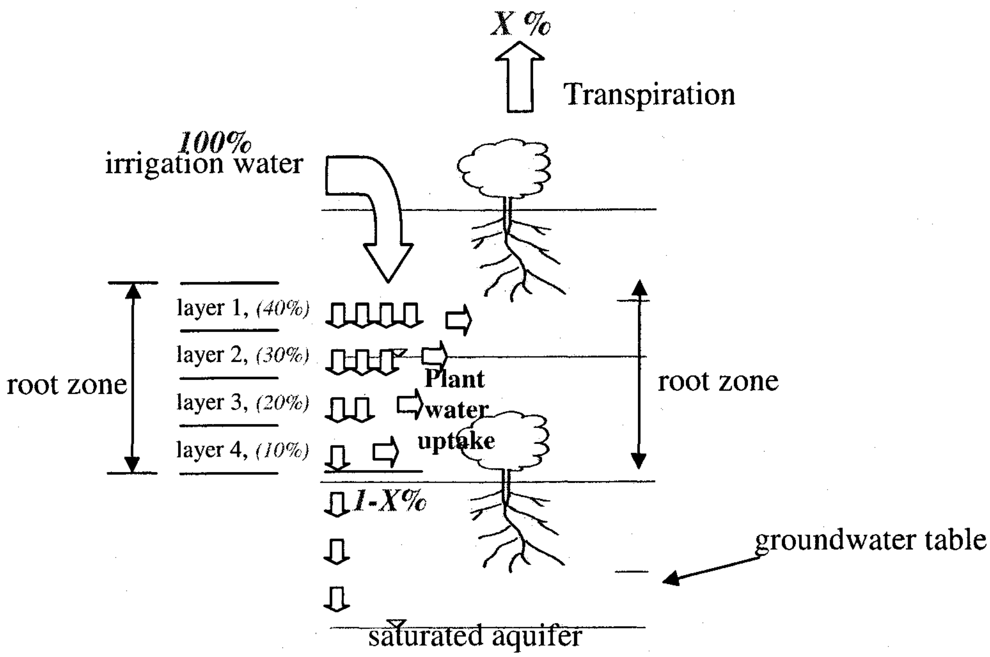


Figure 3.2: Concept for Simulating Water Consumption in Plant Root Zone

3.1.2 Functional Representation

Soil water composition in each soil layer is determined by equilibrium of chemical reactions. This study simulates chemical reactions of carbon dioxide dissolution, soil mineral dissolution / precipitation, and ion-pair association / dissociation.

Carbon dioxide gas, $CO_{2(g)}$, is generated by plant root activity and microorganisms in soil (Oster and Rhoades, 1974). Since $CO_{2(g)}$ can be dissolved in water and changed into HCO_3^- and CO_3^{2-}, it will affect the solubility of carbonate minerals. The dissolution of $CO_{2(g)}$ can be expressed as the following two balance equations:

$$CO_{2(g)} + H_2O \leftrightarrow H^+ + HCO_3^- \qquad K_{CO_2 1} = \frac{(HCO_3^-)(H^+)}{P_{CO_2}} \tag{3.2}$$

$$CO_{2(g)} + H_2O \leftrightarrow 2H^+ + CO_3^{2-} \qquad K_{CO_2 2} = \frac{(CO_3^-)(H^+)^2}{P_{CO_2}} \tag{3.3}$$

where $K_{CO_2 1}$ and $K_{CO_2 2}$ are Henry's Law constants, which are 1.49×10^{-8} and 6.93×10^{-19} respectively. P_{CO_2} is the partial pressure of $CO_{2(g)}$ (in atm). The parentheses, ( ), represent ion activity of the species, the calculation of which is discussed later. The P_{CO_2} is assumed as 0.0007, 0.005, 0.015, 0.023, and 0.030 atm at the surface, first quarter, second quarter, third quarter and bottom of the root zone respectively (Rhoades et al., 1987).

The concept of ion activity is based on the non-ideal behavior of ions in solution. It is based on the Debye-Huckel theory (Glasstone, 1947), which states that the ion activity of a dissolved ion species is equal to its concentration (in moles/kg) multiplied by its activity coefficient. For solutions containing n kinds of ions, the activity coefficients of the i^{th} ion ($i=1,2,...n$) can be expressed as the following equation (Rhoades, 1990):

$$\log \gamma_i = -0.509 z_i^2 \left[\frac{\sqrt{I}}{1+0.328 a_i \sqrt{I}} \right] \tag{3.4}$$

where γ_i is the activity coefficient, z_i is valence, a_i is an adjustment factor of the i^{th} ion. I is the ionic strength of the solution, defined as: $I = 0.5\sum_i m_i z_i^2$; m_i is the modal concentration (in moles/kg) of the i^{th} ion in solution. Mole/kg can be approximated by moles/liter (usually expressed as bracket [ ]) except in extremely saline solutions. The adjustment factor, a_i, is *9.0* for H^+, *3.5* for OH^-, *4.0* for HCO^- and Na^+, *4.5* for CO_3^{2-}, *6.0* for Ca^{2+}, *8.0* for Mg^{2+}, *4.0* for SO_4^{2-}, *3.0* for K^+ and Cl^-, *1.0* for all ion pair species.

In the dissolution/ precipitation reactions between soil solid phase and solution phase, three kinds of minerals are simulated. They are calcite ($CaCO_3$), gypsum ($CaSO_4 \cdot 2H_2O$), and magnesite ($MgCO_3 \cdot 3H_2O$). These minerals are commonly found in salt-affected areas. Their dissolution/ precipitation reaction equations can be expressed as the following:

$$CaCO_3 + CO_{2(g)} + H_2O \leftrightarrow Ca^{2+} + 2HCO_3^- \qquad K_{sp}{}^C = \frac{(Ca^{2+})(HCO_3^-)^2}{P_{CO_2}} \tag{3.5}$$

$$CaSO_4 \cdot 2H_2O \leftrightarrow Ca^{2+} + SO_4^{2-} + 2H_2O \qquad K_{sp}{}^G = (Ca^{2+})(SO_4^{2-})(H_2O)^2 \tag{3.6}$$

$$MgCO_3 \cdot 3H_2O + CO_{2(g)} \leftrightarrow Mg^{2+} + 2HCO_3^- + 2H_2O$$

$$K_{sp}{}^M = \frac{(Mg^{2+})(HCO_3^-)^2(H_2O)^2}{P_{CO_2}} \tag{3.7}$$

where K_{sp} is the solubility product. Index ^{C}, ^{G} and ^{M} refer to calcite, gypsum and megesite. There are maximum levels for $K_{sp}{}^C$, $K_{sp}{}^G$ and $K_{sp}{}^M$. Once these maximum levels are reached, the reaction equations will shift to the left hand side (mineral precipitation) in order to keep the K_{sp} at maximum levels (Snoeyink and Jenkins, 1982). The maximum level of $K_{sp}{}^G$ is 2.45×10^{-5}. The maximum levels of $K_{sp}{}^C$ and $K_{sp}{}^M$ are dependent on the partial pressure of P_{CO_2} in the soil. They are $3.21\times10^{-6}/P_{CO_2}$ and $4.14\times10^{-6}/P_{CO_2}$ for $K_{sp}{}^C$ and $K_{sp}{}^M$.

The formation of ion pair species in soil solute can enhance the solubility of minerals (Filep, 1999). The ion pair species simulated in this study include $CaSO_4^0$,

$MgSO_4^0$, $NaSO_4^-$, KSO_4^-, $CaHCO_3^+$, $MgHCO_3^+$, $NaHCO_3^0$, $CaCO_3^0$, $MgCO_3^0$ and $NaCO_3^-$. Their association/dissociation are expressed in the following balance equations:

$$CaSO_4^{\ 0} \leftrightarrow Ca^{2+} + SO_4^{\ 2-} \qquad K_1 = \frac{(Ca^{2+})(SO_4^{\ 2-})}{(CaSO_4^{\ 0})} \tag{3.8}$$

$$MgSO_4^{\ 0} \leftrightarrow Mg^{2+} + SO_4^{\ 2-} \qquad K_2 = \frac{(Mg^{2+})(SO_4^{\ 2-})}{(MgSO_4^{\ 0})} \tag{3.9}$$

$$NaSO_4^{\ -} \leftrightarrow Na^{2+} + SO_4^{\ 2-} \qquad K_3 = \frac{(Na^{+})(SO_4^{\ 2-})}{(NaSO_4^{\ 0})} \tag{3.10}$$

$$KSO_4^{\ -} \leftrightarrow K^{+} + SO_4^{\ 2-} \qquad K_4 = \frac{(K^{+})(SO_4^{\ 2-})}{(KSO_4^{\ 0})} \tag{3.11}$$

$$CaHCO_3^{\ +} \leftrightarrow Ca^{2+} + HCO_3^{\ -} \qquad K_5 = \frac{(Ca^{2+})(HCO_3^{\ -})}{(CaHCO_3^{\ +})} \tag{3.12}$$

$$MgHCO_3^{\ +} \leftrightarrow Mg^{2+} + HCO_3^{\ -} \qquad K_6 = \frac{(Mg^{2+})(HCO_3^{\ -})}{(MgHCO_3^{\ +})} \tag{3.13}$$

$$NaHCO_3^{\ 0} \leftrightarrow Na^{+} + HCO_3^{\ -} \qquad K_7 = \frac{(Na^{+})(HCO_3^{\ -})}{(NaHCO_3^{\ 0})} \tag{3.14}$$

$$CaCO_3^{\ 0} \leftrightarrow Ca^{2+} + CO_3^{\ 2-} \qquad K_8 = \frac{(Ca^{2+})(CO_3^{\ 2-})}{(CaCO_3^{\ 0})} \tag{3.15}$$

$$MgCO_3^{\ 0} \leftrightarrow Mg^{2+} + CO_3^{\ 2-} \qquad K_9 = \frac{(Mg^{2+})(CO_3^{\ 2-})}{(MgCO_3^{\ 0})} \tag{3.16}$$

$$NaCO_3^{\ -} \leftrightarrow Na^{+} + CO_3^{\ 2-} \qquad K_{10} = \frac{(Na^{+})(CO_3^{\ 2-})}{(NaCO_3^{\ -})} \tag{3.17}$$

where K_i is the equilibrium constant for the i^{th} ion pair species. The value is 4.9×10^{-3} for $CaSO_4^0$, 3.29×10^{-5} for $CaCO_3^0$, 5.53×10^{-2} for $CaHCO_3^+$, 6.3×10^{-3} for $MgSO_4^0$,

4.68×10^{-4} for $MgCO_3^0$, 8.32×10^{-2} for $MgHCO_3^+$, 0.191 for $NaSO_4^-$, 0.282 for $NaCO_3^-$, 0.677 for $NaHCO_3^0$, 0.11 for KSO_4^-.

In addition to the chemical reactions shown above, dissociation of water (H_2O) and hydrogen-carbonate (HCO_3^-) should also be simulated. They are shown as follows:

$$H_2O \leftrightarrow H^+ + OH^- \qquad K_w = \frac{(H^+)(OH^-)}{(H_2O)} \tag{3.18}$$

$$HCO_3^- \leftrightarrow H^+ + CO_3^{2-} \qquad K_{HCO_3^-} = \frac{(H^+)(CO_3^{2-})}{(HCO_3^-)} \tag{3.19}$$

where balance constant $K_w = 10^{-14}$ and $K_{HCO_3^-} = 4.65\times10^{-11}$.

Equations (3.1) to (3.19) are used to simulate chemical reaction mechanisms in soil. By solving these equations, concentrations of the chemical species involved can be calculated. After soil water composition is obtained, total concentration of calcium (*Ca*), magnesium (*Mg*), sodium (*Na*), potassium (*K*), sulfate (SO_4), chloride (*Cl*), carbonate (CO_3), and bicarbonate (HCO_3) can be obtained by the following equations:

$$Ca = [Ca^{2+}] + [CaSO_4^{\,0}] + [CaCO_3^{\,0}] + [CaHCO_3^{\,+}] \tag{3.20}$$

$$Mg = [Mg^{2+}] + [MgSO_4^{\,0}] + [MgCO_3^{\,0}] + [MgHCO_3^{\,+}] \tag{3.21}$$

$$Na = [Na^+] + [NaSO_4^{\,-}] + [NaCO_3^{\,-}] + [NaHCO_3^{\,0}] \tag{3.22}$$

$$K = [K^+] + [KSO_4^{\,-}] \tag{3.23}$$

$$SO_4 = [SO_4^{\,2-}] + [CaSO_4^{\,0}] + [MgSO_4^{\,0}] + [NaSO_4^{\,-}] + [KSO_4^{\,-}] \tag{3.24}$$

$$Cl = [Cl^-] \tag{3.25}$$

$$CO_3 = [CO_3^{\,2-}] + [CaCO_3^{\,0}] + [MgCO_3^{\,0}] + [NaCO_3^{\,-}] \tag{3.26}$$

$$HCO_3 = [HCO_3^-] + [CaHCO_3^-] + [MgHCO_3^+] + [NaHCO_3^0] \quad (3.27)$$

where the bracket, [ ], represents concentrations of species in moles/liter. In the above equations, total concentration of a certain chemical specie type is the sum of all related species. For example, *Ca* (in mole/liter) equals to the sum of all calcium-related species, which are $[Ca^{2+}]$, $[CaSO_4^0]$, $[CaCO_3^0]$ and $[CaHCO_3^+]$.

As aforementioned, this study assumes that the root zone soil can be divided into four layers. For each layer, concentrations of *Ca, Mg, Na, K,* SO_4, *Cl,* CO_3 and HCO_3 can be calculated by applying equations 3.2 to 3.27. The required input data for these calculations are the composition of irrigation water. A C++ program was designed to solve equation 3.1 to 3.27. Detailed calculation algorithms of this program are described in section 3.2.2.

3.2 Soil Water Salinity Simulation in Irrigated Fields

This section will illustrate how to simulate soil water chemical composition in irrigated fields with the chemical reactions mechanism developed in section 3.1.

3.2.1 Components in Soil Water Salinity Simulation

Agricultural irrigation is a process that varies over time. It results in a dynamic soil moisture and soil salinity. Figure 3.3 briefly illustrates the irrigation process in a field. As shown in the figure, irrigation water is first applied to the field from time t_0 to t_2. The water soon percolates into the soil. A portion of the water is absorbed by crops through the root zone. The rest of the water recharges into the saturated aquifer from time t_1 to t_3. In the root zone soil, salinity increases with soil depth due to the crop's concentrating effect. Soil moisture fluctuates with time, depending on the amount of irrigation water and the crops' water needs. When the soil moisture is low, soil salinity increases. If soil moisture drops below a certain level, minerals will precipitate in the root zone soil. Rainfall, as random events, can provide crops extra moisture and also leach down salts in the soil. Based on crop need for water and tolerance of soil salinity, another irrigation starts at time t_5 and the fluctuation of soil moisture and soil salinity continues (Figure 3.3). Since soil water salinity is dynamic, it is important to set up simulation time-step, input data and environmental parameters when simulating soil water chemical composition.

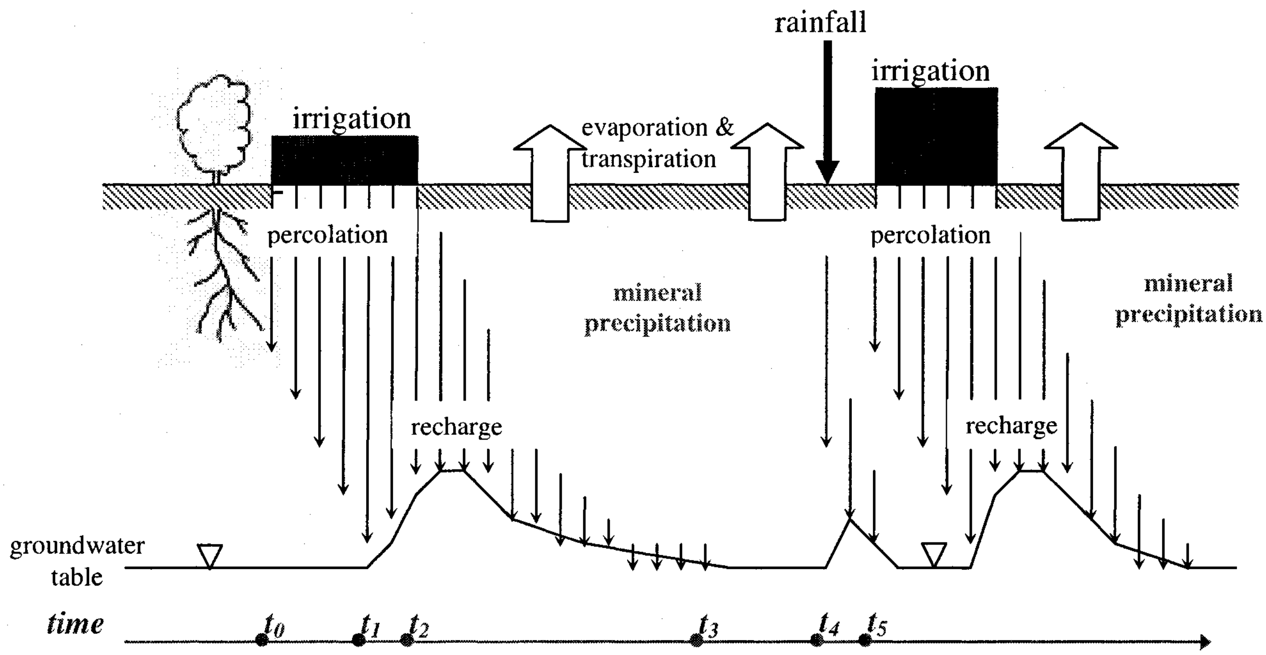


Figure 3.3: Hydrological Dynamics in Irrigated Fields

In this study, the "month" is chosen as the simulation time step. Monthly time steps can provide accuracy for simulating seasonal salinity change. If a smaller time step, such as "day" or "hour" were chosen, it would be difficult to obtain data since all the data would need to be "daily" or "hourly". Since it is almost impossible to find such detailed data for each irrigated field over a long period, the simulation would most likely not be accurate. If monthly time steps are used, the required input data for soil water salinity simulation are monthly irrigation quantity and quality. Monthly irrigation water can be obtained by aggregating all irrigation water and rainfall applied to a field within a month. The quality of the monthly irrigation water can be obtained by mixing all the different sources of water. The items pertaining to water quality include concentrations of Ca, Mg, Na, K, Cl, SO_4, HCO_3^- and CO_3^{2-} in mole/ meter^3 for each source of water. HCO_3^- and CO_3^{2} can be combined and expressed as alkalinity.

By using the input data and the assumed crop consumptive use in each soil layer (shown in figure 3.2), soil water composition is simulated at five depths, soil surface, ¼, ½, ¾ and bottom of the root zone soil. At the soil surface, only mineral dissolution is simulated. The water composition obtained from the simulation at the soil surface is used as input data for simulating water composition at the ¼ root zone depth. Then, the water composition obtained from simulation at the ¼ root zone depth is used as input data for simulating water composition at the ½ root zone depth, and so forth.

As shown in equations 3.5 to 3.7, $CaCO_3$, $CaSO_4$ and $MgSO_4$ may precipitate as soil water becomes more concentrated. When using monthly time steps for simulation, mineral precipitation in a month becomes the initial mineral condition for the next month. For example, if the result of a simulation shows *10* moles of $CaCO_3$ has precipitated in the first ¼ of the root zone soil depth in this month, simulation for the next month will add *10* moles of $CaCO_3$ into the soil water at the first ¼ of the root zone depth. By using this strategy, the phenomenon of salt pick-up, which is commonly found in salt-affected areas (Rhoads, 1999), is simulated in this model. This strategy also allows seasonal fluctuation of soil water salinity to be simulated.

Environmental parameters for simulating soil salinity in irrigated fields include the leaching fraction and soil treatments. Leaching fraction is a parameter that indicates the portion of irrigation water being leached below the root zone soil. This amount is determined by the amount of water absorbed by the crops. Crops absorb water through their roots, transmit it through the plant and release it through pores in their leaves. This process is based on a particular crop's demand for growth, and it is usually referred as evapo-tranpiration (ET). ET is influenced by conditions of solar radiation, temperature, wind velocity, and vapor pressure gradients (Viessman, et al., 1977, p43). Different species of crops also show different ET demands. Also, the ET demands fluctuate within growing seasons. Crops usually need more water in the middle of a growing season and need less water at the beginning and the end of the season. Viessman, et al., 1977 (p54), has pointed out the difficulty of acquiring precise values of ET. In this study, Blaney-Criddle's approach is chosen to calculate crop ET using a monthly time step. In an irrigated field, monthly ET for a specific type of crop based on Blaney-Criddle's approach (Viessman, et al., 1977) can be expressed as:

$$U_i = k_i \cdot t_i \cdot p_i \tag{3.28}$$

where U_i represents ET (in inches) in the i^{th} month, i starts from 1 to the last month of the growing season; k_i and t_i represents consumptive use coefficient and average temperature (°F) in the i^{th} month respectively; p_i is daytime hours in the i^{th} month given as percentage of the growing season. With the addition of information concerning crop acreage, monthly consumptive use for a specific type of crop can be calculated. If more than one type of crop is growing in the irrigated field, the total consumptive use for the field will be the sum of the consumptive use for each individual crop type.

Garcia (2000) developed a computer program, IDSCU, to compute consumptive use. IDSCU allows the use of several ET equations including the Blanney Criddle method to calculate crop ET. IDSCU provides information for parameters used in equation (3.28) for various crop types. This program also has a graphical user interface (GUI) which can facilitate data input, executing the program and displaying the output. The output can be displayed in both a spreadsheet form and graphically. After the crop consumptive use is calculated by IDSCU, the monthly leaching fraction can be calculated.

This study also simulates soil treatment. Two popular treatment methods are simulated: adding sulfuric acid and adding gypsum. Soil amendments are normally applied in order to reduce soil sodicity which is referred to as the amount of sodium contained in the soil. High sodicity will impair soil structure and reduce crop yield. Adding sulfuric acid and adding gypsum have the effect of decreasing the sodium absorption ratio (*SAR*), defined as equation (3.29). The relationship between soil sodicity and *SAR* can be represented using the Gapon equation (Smedema and Rycroft, 1983).

$$SAR = \frac{Na^{+}}{\sqrt{\frac{(Ca^{2+} + Mg^{2+})}{2}}} \qquad (3.29)$$

Adding sulfuric acid will reduce HCO_3^- and CO_3^{2-} in the soil solute. It makes equations (3.1) and (3.2) shift to the left (reduction of HCO_3^- to $CO_{2(g)}$). Due to this change, equations (3.5) and (3.7) will shift to the right (dissolution of calcite and megesite) to maintain chemical equilibrium. Finally, concentrations of Ca^{2+} and Mg^{2+} are increased and the *SAR* is decreased. Adding gypsum will directly increase the concentration of Ca^{2+} and directly decrease *SAR*.

Although adding sulfuric acid and gypsum can alleviate soil sodicity, they also have the unfortunate effect of increasing the salinity level of the drainage water. Thus, the trade-off should be evaluated. In the model, if the user wants to evaluate the impact of adding gypsum, the amount of gypsum (in mole / litter) to be added must be noted. More gypsum will lead to lower *SAR* values but higher *EC* in the soil solute.

3.2.2 Modeling Procedure

The procedure of simulating soil water salinity in irrigated fields is illustrated by the flow chart shown in figure 3.4. For this study, A C/C++ program has been designed for this procedure.

As shown in figure 3.4, the first step is to read in the irrigation water data. The data includes quantity and quality of irrigation water for each month. A pre-calculation is necessary if there is rainfall or more than one irrigation water source for any particular month. The second step is to calculate the leaching fraction for each month. As previously mentioned, this calculation needs to incorporate the IDSCU model for simulating crop consumptive use. The third step is to set up soil treatments based on the farmer's planting practice. After setting up input data and parameters, soil water composition is simulated from the soil surface to the bottom of the root zone for each month of the simulation. Initial soil mineral condition must be added at the beginning of the simulation because it can affect ionic strength of the soil water. Among all chemical reactions, dissolution of CO_2 is simulated first because it affects the precipitation / dissolution of calcite and megesite. Based on equations 3.2, 3.3, 3.5 and 3.7, concentrations of CO_3^{2-} ,HCO_3^-, Ca^{2+} and Mg^{2+} are obtained by using numerical methods. After that, precipitation / dissolution of Gypsum is simulated based on equation

3.6. Then, the formation of various ion pairs in the soil water is simulated based on equations 3.8 to 3.17. Since the ion pair formation will affect dissolution of major ions, the equilibrium must be re-calculated. The sum of electrical charges in the soil water must be zero. If not, the numerical computations must continue until it is zero. Finally, concentrations of all chemical species are obtained. Mineral precipitation in a month is the initial soil mineral condition in the next month. Concentrations of dissolved ions are used to calculate salinity parameters including *EC* and *SAR*.

In the C/C++ program designed for simulating soil water salinity, the numerical method used to calculate ion concentration is based on a steady state soil water composition simulation model, WATSUIT (Rhoades, 1990). The WATSUIT was originally written in Fortran and is re-written in C/C++ for use in this application. A major improvement of this program over the WATSUIT is to allow dynamic state simulation. This improvement is achieved by dividing the simulation period into many time steps and by executing the model sequentially from the first time step to the last time step. The amount of minerals including calcite, magnesite and gypsum precipitated in a time step is used as the initial mineral condition in the soil for soil water composition simulation in the next time step.

Table 3.1 is an example showing the results of the soil water quality simulation of an irrigated field. The simulation period for this example is five months. Each month has different settings for leaching fractions and soil amendment applications. Concentrations of the major ions (*Ca, Mg, Na, K, Cl*, CO_3, HCO_3 and SO_4), salinity index (*EC* and *SAR*) and mineral precipitation (*magnesite, calcite* and *gypsum*) are simulated for the soil

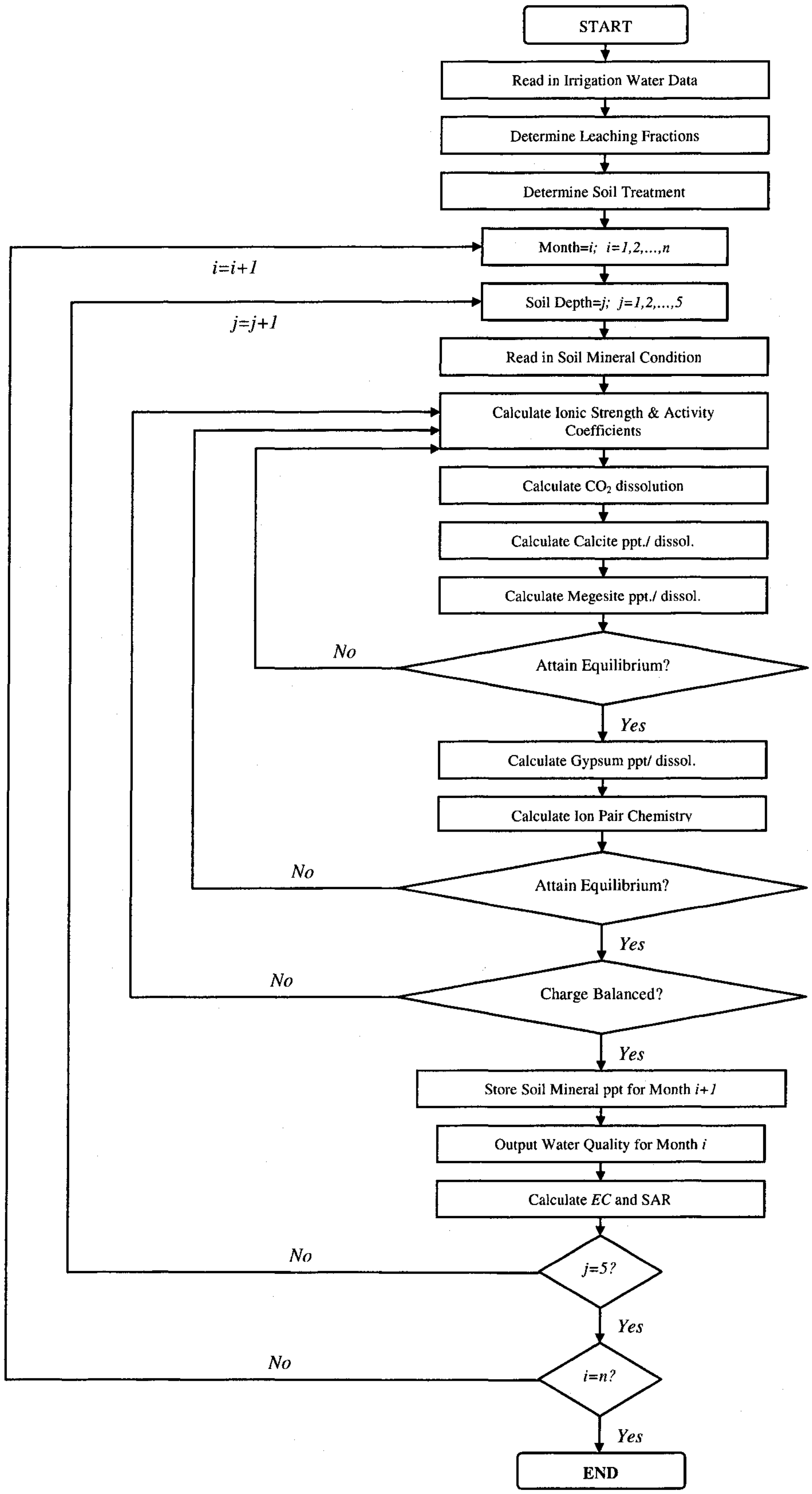


Figure 3.4: Flow Chart of Soil Water Salinity Simulation

surface, ¼, ½, ¾ and bottom of the root zone soil. As shown in the table, as soil depth increases, concentrations of the major ions, *EC* and *SAR* increase significantly. In month_1, due to the low leaching fraction (0.1), *EC* increases from 0.67 to 4.3, *SAR* increases from 1.77 to 7.55 from the soil surface to the bottom of the root zone, and, magnesite and calcite precipitated in the soil. Based on the quantity of irrigation water, the total amount of precipitated mineral can be calculated in moles. In month_2, the leaching fraction is 0.3. In this month, 2 meq/l H_2SO_4 is added to the irrigation water in order to prevent a high *SAR*. The results show that the *SAR* is well controlled in month_2 since the *SAR* is 2.93 in the bottom of the root zone. The other benefit of adding H_2SO_4 is that soil calcite accumulated in month_1 is totally dissolved. This is also helpful in controlling *EC* in future months. In contrast, if H_2SO_4 is not added in month_2, the *SAR* would be 3.21 in the bottom of the root zone and there would be 3.07 mmole/m^3 calcite precipitated at the bottom of the root zone. In month_3, due to simulated rainfall, leaching fraction is increased (0.5) and irrigation water is less saline. Compared to month_1 and month_2, concentrations of *Ca, Mg, Na, K, Cl,* CO_3, HCO_3 and SO_4 are much lower in month_3. Also, there is no mineral precipitated in the soil during this month. However, higher leaching fraction means that more water percolates into the saturated zone and can potentially cause high ground water table problems in irrigated areas. In month_4, the irrigation water contains large amount of salts. The simulation result shows higher concentrations of major ions and large amounts of mineral precipitations in the soil (magnesite and calcite). This results in the white salt deposits that can usually be found in the soil profile in areas suffering from salinity problems. In month_5, there are 6 meq/l $CaSO_4$ added to the irrigation water in order to control *SAR.*

The simulation results show that the *SAR* is controlled at 3.86. If $CaSO_4$ were not added the *SAR* would be 4.97. Magnesite, which accumulated in the previous month, is dissolved. However, even more calcite is precipitated this month, and gypsum is precipitated in the bottom of the root profile.

3.3 River Salinity Simulation in an Irrigated Basin

This section will illustrate how river quantity and quality in an irrigated river basin is simulated. A return flow model is constructed that integrates the soil water salinity simulation model developed in section 3.2.

3.3.1 Components in River Salinity Simulation

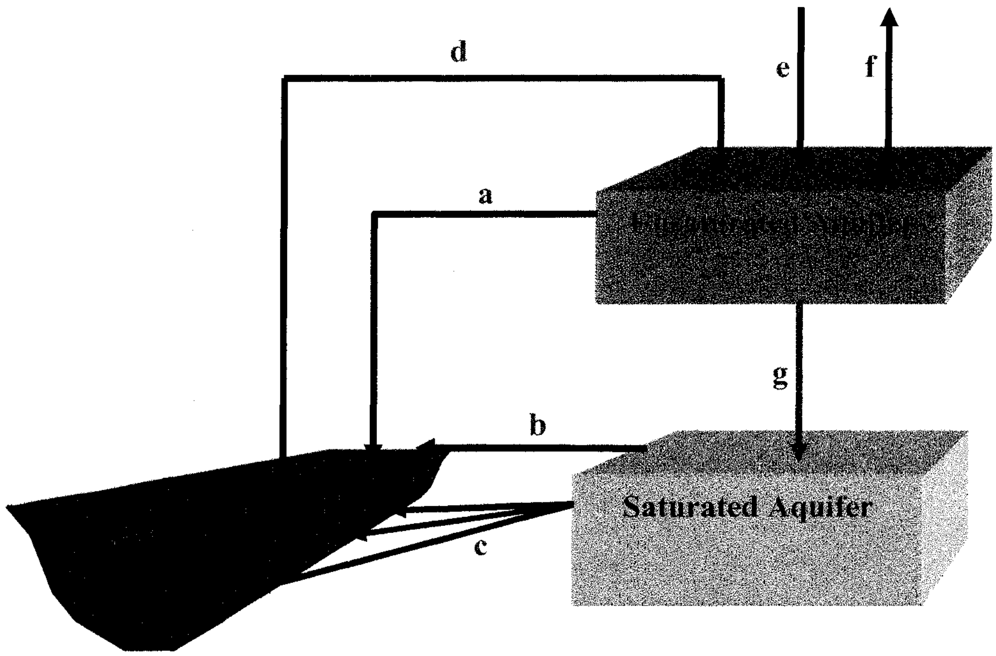
The methodology for the river basin salinity simulation is based on the soil water salinity simulation with the addition of a computation of irrigation return flows. Irrigation return flows are created because of irrigation activity. Figure 3.5 briefly illustrates how return flows are created in an irrigated river basin and how they return to the river. As shown in the figure, there are four kinds of return flows: tail waters, surface runoff, subsurface drainage flows and ground water return flows. Tail waters are created when irrigation water overflows fields and returns to ditches/ canals. Surface runoff is created when excess rainfall occurs. Subsurface drainage flows are created when percolating flows are intercepted by drainage devices (tile drains) underneath irrigated fields. Ground water return flows are created by percolation from the unsaturated aquifer to the saturated aquifer. Tail water and surface runoff return to the river via open ditch systems. Subsurface drainage flows are first transmitted from subsurface drainage pipes to open ditches before returning to the river. Ground water return flows return to the river via

Table 3.1: Output of Soil Water Composition Simulation

	Ca	*Mg*	*Na*	*K*	*Cl*	CO_3	HCO_3	SO_4	*pH*	*EC*	*SAR*	*Megne-site*	*Cal-cite*	*Gyp-sum*
Month 1: Leaching Fraction=0.1; Amount of Irrigated Water=10 acre-ft														
Irrigation water quality	2.52	1.37	2.67	0.13	0.62	0	2.62	3.55						
Soil Surface	2.32	2.15	2.65	0.13	0.61	0.72	2.39	3.52	8.31	0.67	1.77	0.2	0.18	0
¼ root zone depth	3.91	3.67	4.14	0.2	0.96	0.63	4.82	5.5	7.74	1	2.13	0	0	0
½ root zone depth	5.94	6.31	7.15	0.35	1.66	0.7	7.88	9.51	7.46	1.58	2.89	0.04	0.82	0
¾ root zone depth	8.32	8.88	13.94	0.68	3.24	0.69	9.35	18.53	7.33	2.52	4.75	3.49	4.84	0
Bottom root zone depth	11.96	12.71	26.51	1.29	6.15	0.69	10.38	35.24	7.24	4.3	7.55	10.81	13.07	0
Month 2: Leaching Fraction=0.30; Amount of Irrigated Water=15 acre-ft; Adding 2 Meq/l H_2SO_4														
Irrigation water quality	2.52	1.37	2.67	0.13	0.62	0	2.62	3.55						
Soil Surface	2.62	1.49	2.65	0.13	0.61	0.07	0.67	5.52	7.75	0.67	1.85	0	0	0
¼ root zone depth	3.47	1.89	3.68	0.18	0.85	0.01	0.68	7.67	6.9	0.85	2.25	0	0	0
½ root zone depth	5.3	2.69	5.19	0.25	1.21	0.02	1.37	10.83	6.71	1.17	2.6	0	0	0
¾ root zone depth	8.42	4.87	7.16	0.35	1.66	0.16	4.05	14.93	6.98	1.66	2.78	0	0	0
Bottom root zone depth	11.25	6.94	8.84	0.43	2.05	0.37	6.6	18.43	7.07	2.11	2.93	0	0	0
Month 3: Leaching Fraction=0.50; Amount of Irrigated Water=20 acre-ft; 5 cm rainfall														
Irrigation water quality	1.85	0.95	1.89	0.1	0.5	0.32	1.59	2.38	8.14	0.48	1.6	0	0	0
Soil Surface	2.31	1.19	2.37	0.12	0.62	0.12	2.28	2.98	7.44	0.57	1.79	0	0	0
¼ root zone depth	2.84	1.47	2.92	0.15	0.76	0.07	2.88	3.66	7.05	0.67	1.99	0	0	0
½ root zone depth	3.36	1.73	3.45	0.18	0.9	0.07	3.41	4.33	6.94	0.77	2.16	0	0	0
¾ root zone depth	3.7	1.91	3.79	0.2	0.99	0.07	3.77	4.77	6.86	0.84	2.27	0	0	0
Bottom root zone depth	1.85	0.95	1.89	0.1	0.5	0.32	1.59	2.38	8.14	0.48	1.6	0	0	0
Month 4: Leaching Fraction=0.3; Amount of Irrigated Water=20 acre-ft;														
Irrigation water quality	5.81	3.72	7.62	0.15	1.4	0	3.38	10.28						
Soil Surface	3.03	2.93	7.55	0.15	1.39	0.73	2.35	9.19	8.28	1.22	4.37	0.76	2.73	0
¼ root zone depth	4.88	5.04	10.49	0.21	1.93	0.71	5.21	12.77	7.76	1.72	4.71	0.08	3.12	0
½ root zone depth	7.17	7.23	14.81	0.29	2.72	0.7	8.04	18.04	7.46	2.39	5.52	0	4.13	0
¾ root zone depth	8.83	9.36	20.42	0.4	3.75	0.7	9.69	24.87	7.34	3.17	6.77	0.61	6.75	0
Bottom root zone depth	10.28	10.89	25.2	0.5	4.63	0.7	10.84	30.69	7.26	3.82	7.75	1.42	8.94	0
Month 5: Leaching Fraction=0.25; Amount of Irrigated Water=15 acre-ft; Adding 6 Meq/l $CaSO_4$														
Irrigation water quality	2.52	1.37	2.67	0.13	0.62	0	2.62	3.55						
Soil Surface	8.03	3.36	3.64	0.13	1.6	0.65	1.4	11.5	8.05	1.23	1.52	0	5.08	0
¼ root zone depth	13.92	3.56	5.35	0.19	2.36	0.64	3.11	16.91	7.52	1.78	1.81	0	4.42	0
½ root zone depth	21.4	5.34	8.27	0.29	3.65	0.63	4.87	26.15	7.22	2.65	2.26	0	6.53	0
¾ root zone depth	30.31	9.47	13	0.46	5.74	0.64	5.76	41.11	7.08	3.95	2.92	0	15.46	0
Bottom root zone depth	29.92	14.6	18.21	0.65	8.04	0.65	6.97	47.72	7.04	4.78	3.86	0	25.57	9.87

Note: Unit for ions and minerals is mmole/m^3 (Meq/l), Unit for *EC* is dS/m, *pH* and *SAR* are unitless

porous media in the saturated aquifer. Because return flows will carry salts back to the river, the main task in simulating river salinity is to address the quantity and quality of each kind of return flow.



a = irrigation tail waters and runoff
b = subsurface drainage flows
c = groundwater return flows
d = surface water diversion
e = rainfall
f = evaporation, transpiration
g = aquifer recharge

Figure 3.5: Return Flow in an Irrigated River Basin

In section 3.1, the methodology for simulating the quantity and quality of deep percolating water in an irrigated area is described. The percolating water will recharge into the saturated aquifer, flow through the aquifer and return to the river. As it flows through the aquifer, its chemical composition will be affected by the aquifer material. Dissolution of minerals can have a major influence on groundwater quality. To simulate the dissolution process, detailed information about the sort and the amount of the soluble minerals in the aquifer is required. Since this study is focused on a basin-scale study area instead of a small scale area, it is very difficult if not impossible to collect complete geological and hydro-chemical data. Therefore, a method which classifies the groundwater return flow into "short term" (less than 12 months) or "long term" (equal or more than 12 months) groundwater return flow is used in this study because the amount of mineral dissolved by the groundwater is dependent on how long the groundwater stays in the aquifer (Freeze and Cherry, 1979). Since the influence of mineral dissolution on groundwater that remains in the aquifer only for the short term is not significant, the chemical composition of the short-term groundwater is assigned to be equal to the composition of the percolating water. The chemical composition of the groundwater that stays in long term is assigned to be the same as composition of the observed groundwater in the deep aquifer zone. This is because when groundwater stays underground for a long time, its composition will be affected by the aquifer material, and its composition should be close to groundwater in the deep groundwater zone. As long as observed groundwater data from pumping wells are available, the quality of the "long term" groundwater return flow can be simulated.

Simulation of return flows in an irrigated river basin follows three steps: the first step is to divide the entire river into several reaches based on the locations of canal intakes. Each reach has a canal intake at the beginning of the reach. Also, irrigated fields that use the same canal for water delivery are integrated together as a "canal area". Figure 3.6 illustrates this strategy diagrammatically. The second step is to construct response functions for each kind of return flow created in each canal area in each month. The response function represents the spatial and temporal distribution of the return flows. The third step is to calculate each kind of return flow based on the response functions and amount of irrigation water applied to canal areas.

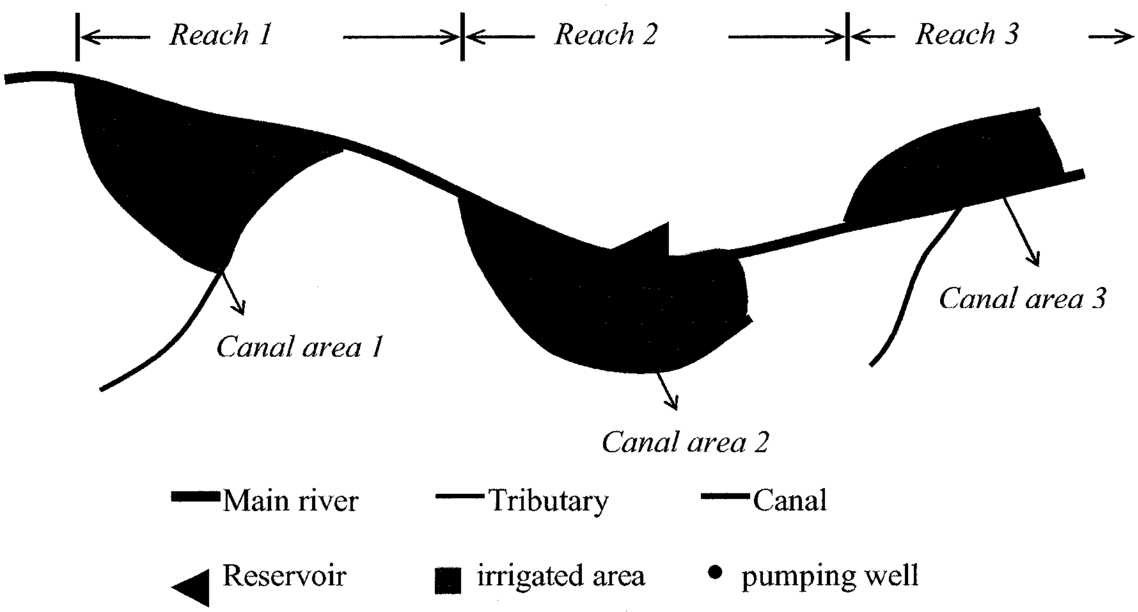


Figure 3.6: Strategy to Divide an Irrigated River Basin

In the calculation of return flows, the river basin is assumed to be a "linear system" (Chow, 1988). Under this assumption, response functions are independent of the amount of water applied to each canal service area. This assumption also implies that different kinds of return flows can be superimposed spatially and temporally. Therefore, the total amount of return flows received by a river reach in a certain month is the summation of all the kinds of return flows flowing to that reach in that month.

Depending on the hydrological characteristics of the basin to be simulated, there could be other kinds of return flows. If there is leakage from reservoirs, or the bottom of the river channel / canal is permeable, return flows will occur due to the leakage. Instead of return flows to the river, depletion from the river could occur if there is ground water pumping in the basin. Depletion due to pumping is calculated as a "negative return flow." To explain how return flows / depletion are calculated and how river salinity is simulated, consider a river basin that contains n canals and the simulation periods are p months. The river system therefore is divided into n reaches with n canal areas in each reach. Let $DIV(i,t)$ represent waters diverted from i^{th} river reach in t^{th} month. By deducting canal transit loss, the net amount can be represented as $NDIV(i,t)$. Therefore, the amount of surface water applied to the i^{th} canal area in the t^{th} month, denoted as $SIR(i,t)$, is:

$$SIR(i,t) = NDIV(i,t) + PPT(i,t) \tag{3.31}$$

where $PPT(i,t)$ represents rainfall applied to i^{th} canal area in the t^{th} month. Chemical composition of $SIR(i,t)$ is the mixture of $DIV(i,t)$ and $PPT(i,t)$. The mixture is calculated as follows: if X_1 units of $DIV(i,t)$ containing Y_1 units of Ca^{2+} mix with X_2 units of $PPT(i,t)$ containing Y_2 units of Ca^{2+}, the mixture will result in X_1+X_2 units of $SIR(i,t)$ with Y_1+Y_2 units of Ca^{2+}. The same calculation is applied to Mg^{2+}, Na^+, K^+, SO_4^{2-}, Cl^- and *alkalinity*. It should be noted that *alkalinity* is the sum of $HCO3^-$ and $CO3^{2-}$. Although $HCO3^-$ and $CO3^{2-}$ will exchange with each other during mixture, their sum remains the same.

The canal transit loss will percolate to the saturated zone and return to the downstream river reaches. Return flows caused by canal transit loss can be expressed as follows:

$$CLS(i,t) = C\_CLS(i,t) \cdot DIV(i,t) = DIV(i,t) - NDIV(i,t) \tag{3.32}$$

$$CRET(i,t,j,s) = C\_CRET(i,j,s) \cdot CLS(i,t) \qquad j \geq i, \quad s \geq t \tag{3.33}$$

$$\sum_{s \geq t} \sum_{j \geq i} C\_CRET(i,j,s) = 1 \tag{3.34}$$

where $C\_CLS(i,t)$ represents the portion of diverted flows which turns into canal transit loss. $CRET(i,t,j,s)$ represents return flows due to the transit loss. $C\_CRET(i,j,s)$ is the response function for canal transit loss return flows. Subscript i represents the canal in which transit loss occurs. Subscript t represents the month in which transit loss occurs.

Subscript j represents the river reach that transit loss return flows return to. Subscript s represents the month when transit loss return flows return. $j \geq i$ and $s \geq t$ because return flows go from upstream to downstream and will last for several months after transit loss occurs at time t.

When using surface water for irrigation, a certain portion of it will be lost due to irrigation tail waters or open ditch drains. In addition, excess rainfall during wet seasons can result in surface runoff. Let $TA(i,t)$ represent the tail water, ditch drain and surface runoff created in i^{th} canal area during t^{th} month, $TA(i,t)$ can be expressed as:

$$TA(i,t) = C\_TA(i,t) \cdot SIR(i,t) \tag{3.35}$$

where $C\_TA(i,t)$, referred to as tail water ratio here, represents the portion of $SIR(i,t)$ which turns into $TA(i,t)$. The chemical composition of $TA(i,t)$ is the same as $SIR(i,t)$. In this study, $TA(i,t)$ is surface return flow and $C\_TA(i,t)$ is tail water ratio.

By subtracting $TA(i,t)$, the amount of net infiltration, $ISIR(i,t)$, can be expressed as:

$$ISIR(i,t) = SIR(i,t) - TA(i,t) \tag{3.36}$$

As water leaches down from the surface of the soil to the bottom of the root zone, chemical concentration in the water increases due to the crop's consumptive use. Section 3.2 has illustrated how to calculate the water quantity / quality change in a single field. The same methodology is applied here. The only difference is that a canal service area, which involves hundreds of fields, is treated as a "big" field. By using crop consumptive use data, the value of the leaching fraction, $LF(i,t)$, for the i^{th} canal area and in the t^{th} month can be calculated. Therefore, the amount of water percolating from the unsaturated zone to the saturated zone, denoted as $PSIR(i,t)$, can be expressed as:

$$PSIR(i,t) = LF(i,t) \cdot SIR(i,t) \tag{3.37}$$

The percolating water will return to the river system by subsurface drainage or by ground water flows. The former returns quickly to the nearby river reach while the latter returns slowly to downstream river reaches. When combined together, they can be expressed as the following equations:

$$SRET(i,t,j,s) = C\_SRET(i,j,s) \cdot PSIR(i,t) \qquad j \geq i, \quad s \geq t \tag{3.38}$$

$$\sum_{s\geq t}\sum_{j\geq i} C\_SRET(i,j,s) = 1 \tag{3.39}$$

where *SRET(i,t,j,s)* represents return flows due to aquifer recharge as a result of surface water irrigation. *C_SRET(i,j,s)* is the response function for return flow due to surface water irrigation. Subscript i represents the canal area in which aquifer recharge occurs. Subscript t represents the month in which aquifer recharge occurs. Subscript j represents the river reach that return flows return to. Subscript s represents the month when return flows return. $j \geq i$ and $s \geq t$ because return flows go from upstream to downstream and will last for several months after time t.

In addition to surface water, ground water can be pumped for irrigation use. Since ground water used for irrigation is usually applied with sprinkler systems, there is minimal tail water or surface runoff generated during irrigation. Let *GIR(i,t)* represent the amount of ground water pumped from the i^{th} canal area in the t^{th} month. Due to crop's consumptive use, the amount of water percolating below the root zone, denoted as *PGIR(i,t)*, is:

$$PGIR(i,t) = LF(i,t) \cdot GIR(i,t) \tag{3.40}$$

The chemical composition of $PGIR(i,t)$ can be calculated by the methodology described in Section 3.2. The corresponding return flows can be expressed by the following equations.

$$GRET(i,t,j,s) = C\_GRET(i,j,s) \cdot PGIR(i,t) \qquad j \geq i, \quad s \geq t \qquad (3.41)$$

$$\sum_{s \geq t} \sum_{j \geq i} C\_GRET(i,j,s) = 1 \qquad (3.42)$$

where $GRET(i,t,j,s)$ represents return flows due to aquifer recharge resulting from ground water pumping for irrigation. $C\_GRET(i,j,s)$ is the response function for return flow due to ground water pumping. Subscript i represents the canal area in which aquifer recharge occurs. Subscript t represents the month in which aquifer recharge occurs. Subscript j represents the river reach that return flows return to. Subscript s represents the month when return flows return. $j \geq i$ and $s \geq t$ because return flows go from upstream to downstream and will last for several months after time t.

Ground water pumping will result in stream depletions to downstream river reaches and last for several months. The depletions can be expressed as follows:

$$GDEP(i,t,j,s) = C\_GDEP(i,j,s) \cdot GIR(i,t) \qquad j \geq i, \quad s \geq t \qquad (3.43)$$

$$\sum_{s \geq t} \sum_{j \geq i} C\_GDEP(i,j,s) = 1 \qquad (3.44)$$

where $GDEP(i,t,j,s)$ represents depletions caused by the pumping, $GIR(i,t)$. $C\_GDEP(i,j,s)$ is the depletion response function for $GDEP(i,j,s)$. Subscript i represents the canal area in which ground water pumping occurs. Subscript t represents the month in which ground water pumping occurs. Subscript j represents the river reach that is depleted by ground water pumping. Subscript s represents the month in which reach j is depleted. $j \geq i$ because stream depletion as a result of ground water pumping effects downstream river reaches. $s \geq t$ because stream depletion effects will continue for several months after pumping occurs at time t.

Other than irrigation that can create significant amounts of return flow to the river, phreatophytes and reservoir operation also have substantial influence in river basin water quantity and quality. Phreatophytes refer to the kinds of natural vegetation, usually trees or shrubs, which can consume large amounts of water from the hydrological system. Since they usually are located near river banks, the depletion effect to the river can not be ignored. Reservoirs, due to their vast surface area, can result in significant evaporation loss and leakage. The leakage will percolate to the ground water system and return to downstream river reaches. If there is a reservoir located in reach i, the reservoir leakage and its return flow can be expressed as:

$$LRET(i,t,j,s) = C\_LRET(i,j,s) \cdot LEK(i,t) \qquad j \geq i, \quad s \geq t \tag{3.45}$$

$$\sum_{s \geq t} \sum_{j \geq i} C\_LRET(i,j,s) = 1 \tag{3.46}$$

where $LEK(i,t)$ represents the amount of leakage. $LRET(i,t,j,s)$ represents return flows due to the reservoir leakage. $C\_LRET(i,j,s)$ is the response function for $LRET(i,t,j,s)$. Subscript i represents the reach in which reservoir leakage occurs. Subscript t represents the month in which reservoir leakage occurs. Subscript j represents the river reach that reservoir leakage return flows return to. Subscript s represents the month in which the return flows return to reach j.

As river waters transit from upstream to downstream, a portion of the water is lost to river channel leakage. Let $RCLEK(i,t)$ represent the amount of river channel leakage in the i^{th} reach in the t^{th} month. $RCLEK(i,t)$ can be represented as follows:

$$RCLEK(i,t) = C\_RCLEK(i,t) \cdot RIVER(i,t) \tag{3.47}$$

where *RIVER(i,t)* is the total volume of river water in the i^{th} reach in the t^{th} month which can be obtained by calculating the upstream inflow and all return flow and river depletion items. *C_RCLEK(i,t)* is the portion of the river water which is lost due to river channel leakage.

River channel leakage will return to downstream river reaches. The return flows can be expressed as:

$$RCRET(i,j,s) = C\_RCRET(i,j,s) \cdot RCLEK(i,t) \qquad j \geq i, \quad s \geq t \tag{3.48}$$

$$\sum_{s \geq t} \sum_{j \geq i} C\_RCRET(i,j,s) = 1 \tag{3.49}$$

where *RCRET(i,j,s)* represents return flows due to river channel leakage, *RCLEK(i,j)*. *C_RCRET(i,j,s)* is the response function for *RCRET(i,j,s)*. Subscript i represents the reach in which river channel leakage occurs. Subscript t represents the month in which river channel leakage occurs. Subscript j represents the river reach that return flows return to. Subscript s represents the month in which return flows return to reach j. $j \geq i$ because the return flows go from upstream to downstream. $s \geq t$ because the return flows will continue for several months after river channel leakage occurs at time t.

By calculating the return flows, depletion and other input / output terms, the mass balance in the j^{th} river reach in the s^{th} month can be expressed as follows.

$$\begin{aligned}\Delta S = {} & RIN(j,s) - ROUT(j,s) + TRI(j,s) - DIV(j,s) - PHY(j,s) + TA(j,s) \\ & + \sum_{i \leq j} \sum_{t \leq s} CRET(i,t,j,s) + \sum_{i \leq j} \sum_{t \leq s} SRET(i,t,j,s) + \sum_{i \leq j} \sum_{t \leq s} GRET(i,t,j,s) \\ & - \sum_{i \leq j} \sum_{t \leq s} GDEP(i,t,j,s) - EVP(j,s) - LEK(j,s) + \sum_{i \leq j} \sum_{t \leq s} LRET(i,t,j,s) \\ & - RCLEK(j,s) + \sum_{i \leq j} \sum_{t \leq s} RCRET(i,t,j,s)\end{aligned} \tag{3.50}$$

where ΔS is the storage change in j^{th} reach within the month; *RIN(j,s)* and *ROUT(j,s)* represent the amount of river water flow in and out of reach i in month s; *TRI(j,s)* represents the total tributary inflows to the reach; *PHY(j,s)* represents phreatophyte consumption; $\sum_{i \le j} \sum_{t \le s} CRET(i,t,j,s)$ represents the summation of all return flows due to canal transit loss from upstream canals; $\sum_{i \le j} \sum_{t \le s} SRET(i,t,j,s)$ represents the summation of all return flows due to surface water irrigation, from upstream canal areas; $\sum_{i \le j} \sum_{t \le s} GRET(i,t,j,s)$ represents the summation of all return flows due to ground water irrigation from upstream canal areas; $\sum_{i \le j} \sum_{t \le s} GDEP(i,t,j,s)$ represents the accumulated effect of stream depletion in the i^{th} reach due to ground water pumping in the upstream canal areas. *EVP(j,s)* represents reservoir evaporation loss. $\sum_{i \le j} \sum_{t \le s} LRET(i,t,j,s)$ represents the summation of all return flows due to upstream reservoir leakage. $\sum_{i \le j} \sum_{t \le s} RCRET(i,t,j,s)$ represents the summation of all return flows due to upstream river channel loss.

In equation 3.50, ΔS and *ROUT(j,s)* are two unknown terms. For the reach that does not have a reservoir on it, ΔS can be ignored. *ROUT(j,s)* can be calculated since other terms in the equation are known. The chemical composition of *ROUT(j,s)* can be obtained by calculating the mixture of different types of inflows and return flows. For the reach that has a reservoir on it, information about reservoir releases is required to obtain ΔS. Then, *ROUT(j,s)* and the chemical composition of *ROUT(j,s)* can be calculated. As the computation repeats from the upstream reach to the downstream reach and from the first month to the last month, water salinity in the entire river system can be simulated over the entire simulation period.

The required input data for river water salinity simulation include: (a) surface water diversion records, (b) ground water pumping records, (c) rainfall records, (d) tributary inflow records, (e) chemical composition of rainfall, pumped water and tributary inflows, (f) crop and non-crop consumptive use data.

From equations 3.31 to 3.49, many response functions are defined to calculate different types of return flows and depletions. They are *C_CLS(i,t)* in equation 3.32, *C_CRET(i,j,s)* in equation 3.33, *C_TA(i,t)* in equation 3.35, *C_SRET(i,j,s)* in equation 3.38, *C_GRET(i,j,s)* in equation 3.41, *C_GDEP(i,j,s)* in equation 3.43, *C_LRET(i,j,s)* in equation 3.45. *C_RCLEK(i,t)* in equation 3.47 and *C_RCRET(i,j,s)* in equation 3.48. These functions are crucial for simulating river water salinity in an irrigated basin. The way in which these response functions are obtained will be illustrated in the next chapter.

3.3.2 Modeling Procedure

The procedure for simulating river salinity combines the simulation of irrigation return flows and the simulation of soil water composition (shown in section 3.2.2). A C/C++ program has been implemented for this procedure. The algorithm of this program is illustrated as a flowchart in figure 3.7.

As shown in figure 3.7, the first step in the flowchart is to read input data and parameters which describe water use and hydrological characteristics of the river basin. After processing the input data into the appropriate format, iterations start from the upstream canal area to the downstream canal area ($i=1,2,..,p$), and from the first month to the last month ($t=1,2,..,n$). Within each iteration, return flows (or depletion effect) due to different "hydrologic stresses" are simulated. There are seven kinds of hydrologic stresses. Following the simulation order, they are: canal transit losses, irrigation tail water / surface runoff, surface water irrigation, groundwater irrigation, phreatophyte consumption, river channel leakage and reservoir leakage. The quantity and chemical composition of the return flows are simulated based on input data, response functions and the soil water salinity program developed in section 3.2.2. Each time the return flows of a certain hydrologic stress are simulated, water quantity and quality in certain river reaches in certain months are updated by mixing old river flows with the new return flows. As the iteration continues, water quantity and quality are obtained sequentially from the first month to last month and from upstream reaches to downstream reaches. Figures 3.8 to 3.11 show the results of a river salinity simulation for an example problem. In this example problem, the simulation period is twelve months. The watershed has five canals. The irrigation area for these five canal areas are 20,746 acre-ft, 4,985 acre-ft, 3,004 acre-ft, 17,925 acre-ft and 13,692 acre-ft. The canal lengths of these five canals are 65.1 miles, 14.3 miles, 38.1 miles, 20.6 miles and 23.8 miles. The growing season is from April to October. During this season the diverted surface water from these five canals are 100,945 acre-ft, 30,942 acre-ft, 10,122 acre-ft, 112,023 acre-ft and 65,638 acre-ft. Annual pumping for these five canals are 7,937 acre-ft, 2,869 acre-ft, 792 acre-ft, 5,075 acre-ft and 2,941 acre-ft. Annual rainfall on the service areas of these five canals are 21,331 acre-ft, 4,343 acre-ft, 522 acre-ft, 15,554 acre-ft and 16,928 acre-ft. The river diversion diagram of this example is shown in figure 3.8. Figure 3.9 shows return flows in each river reach. Figure 3.10 is the corresponding EC values for the return flows. Figure 3.11 is the summation of the stream depletion, including stream depletions caused by ground

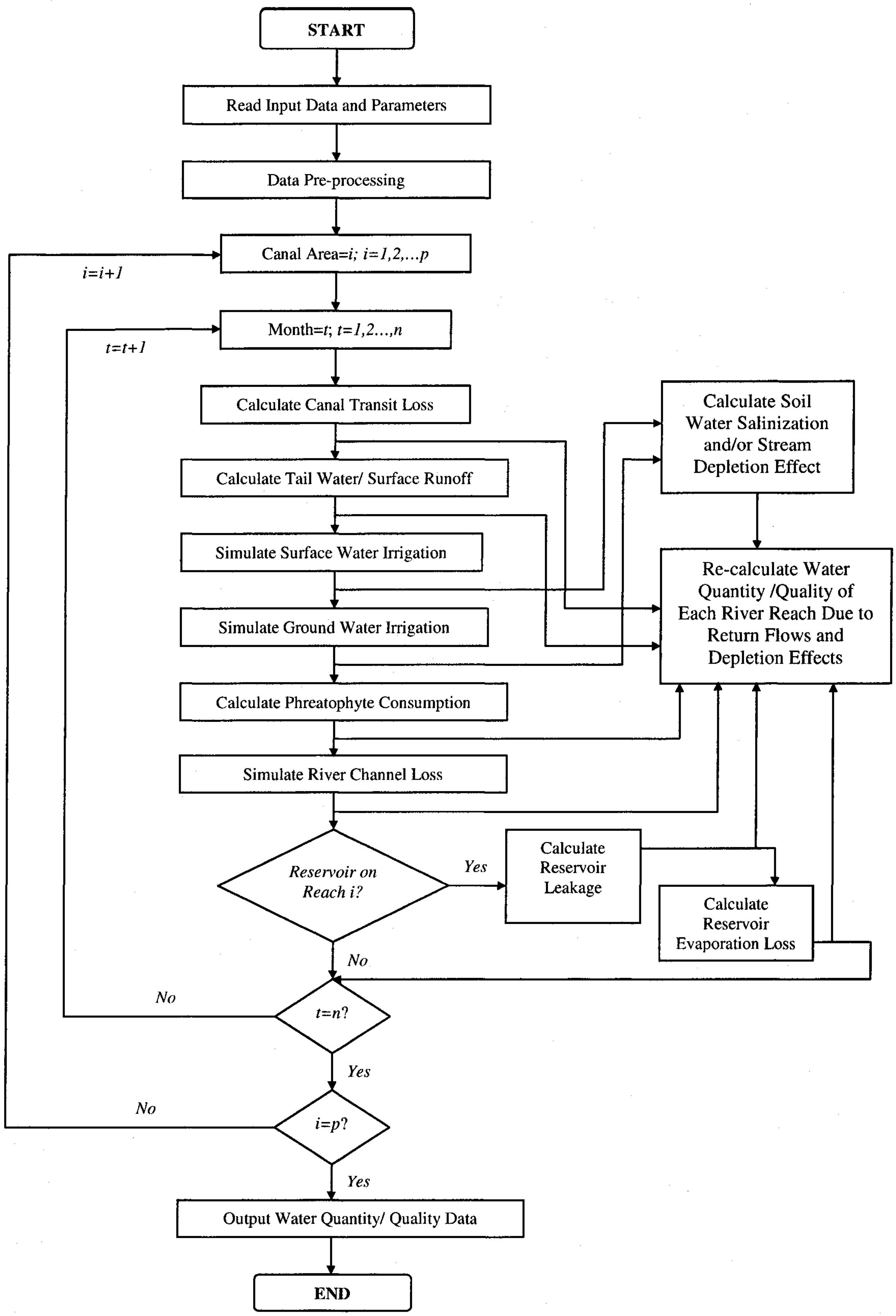


Figure 3.7: Flow Chart of River Salinity Simulation

water pumping and phreatophyte consumption. Figure 3.12 shows the simulated river salinity by calculating return flows and river depletion.

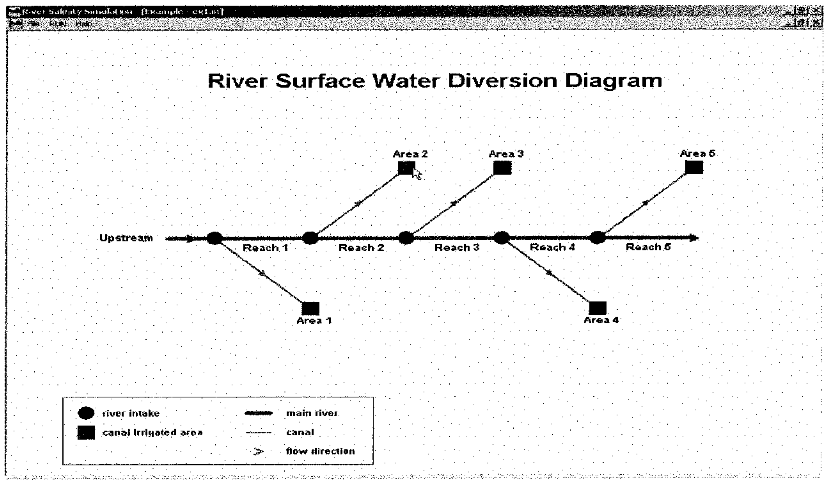
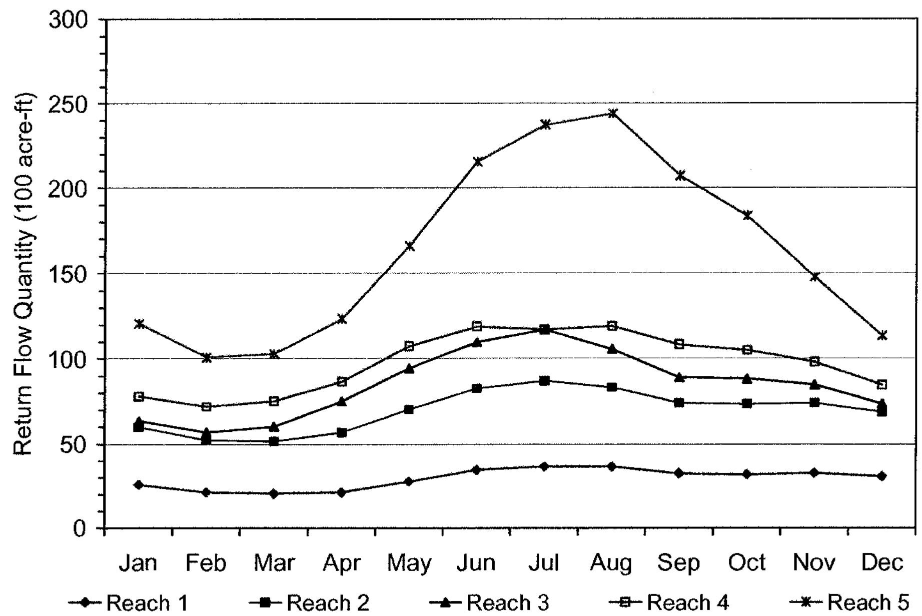


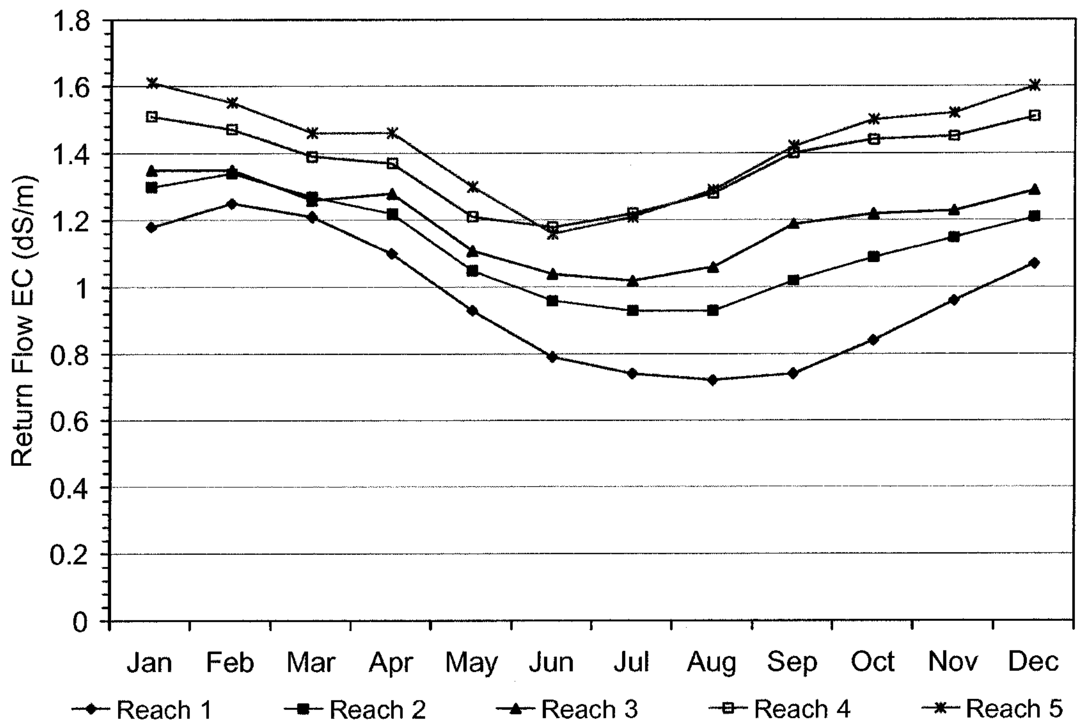
Figure 3.8: River Diversion Diagram



Units: 100 acre-ft

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Reach 1	25.7	21.2	20.4	21.1	27.5	34.5	36.5	36.3	32.4	31.8	32.7	30.7
Reach 2	60.1	52.3	51.7	56.7	70.5	82.8	87.3	83.5	74.2	73.8	74.4	69.0
Reach 3	63.8	57.0	60.4	75.3	94.7	110.0	117.2	105.9	89.3	88.5	85.1	73.8
Reach 4	78.1	72.1	75.1	86.6	107.6	118.9	117.2	119.2	108.5	105.1	98.1	84.7
Reach 5	120.8	101.0	103.0	123.4	165.9	215.6	237.5	244.0	207.3	183.6	147.8	113.6

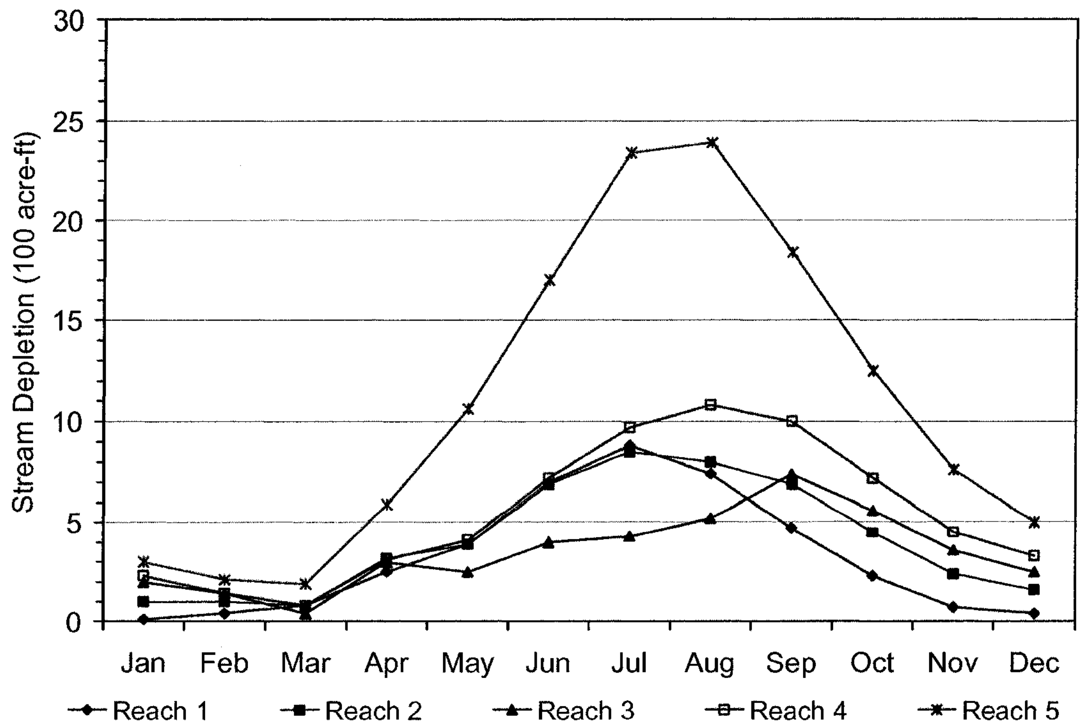
Figure 3.9: Simulation of the Quantity of Return Flow (12 months)



Units: dS/m

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Reach 1	1.18	1.25	1.21	1.10	0.93	0.79	0.74	0.72	0.74	0.84	0.96	1.07
Reach 2	1.30	1.34	1.27	1.22	1.05	0.96	0.93	0.93	1.02	1.09	1.15	1.21
Reach 3	1.35	1.35	1.26	1.28	1.11	1.04	1.02	1.06	1.19	1.22	1.23	1.29
Reach 4	1.51	1.47	1.39	1.37	1.21	1.18	1.22	1.28	1.40	1.44	1.45	1.51
Reach 5	1.61	1.55	1.46	1.46	1.30	1.16	1.21	1.29	1.42	1.50	1.52	1.60

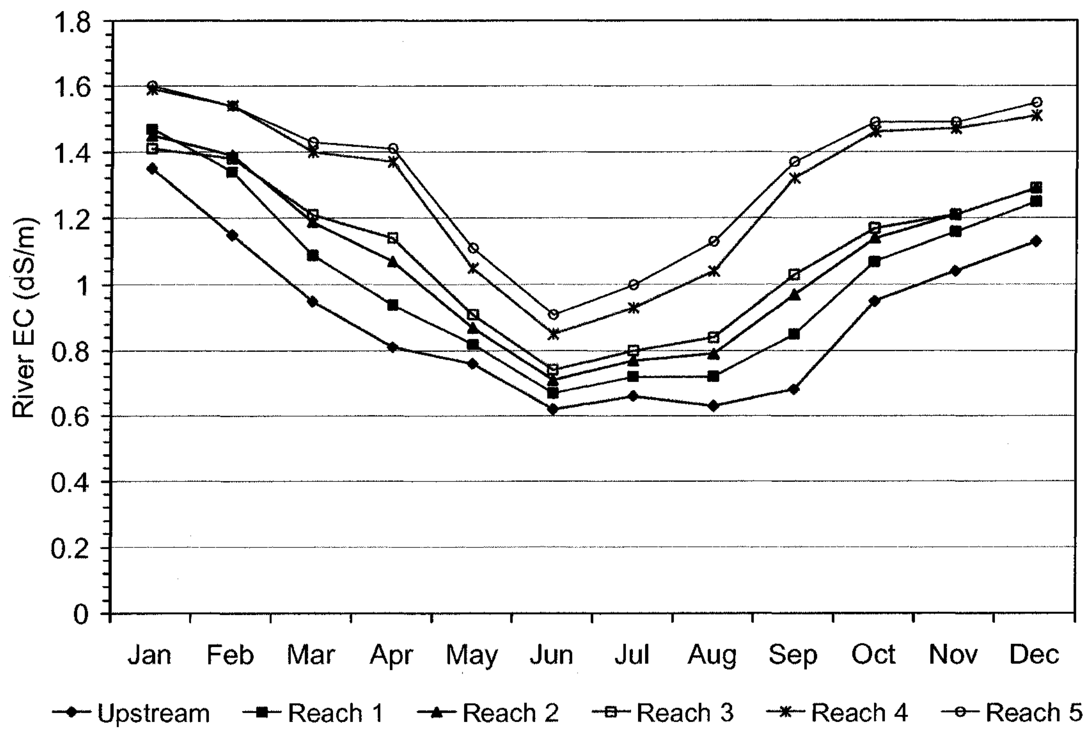
Figure 3.10: Simulation of the EC of Return Flow (12 months)



Units: 100 acre-ft

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Reach 1	0.1	0.4	0.8	2.5	3.9	7.0	8.8	7.4	4.7	2.3	0.7	0.4
Reach 2	1.0	1.0	0.8	3.2	3.9	6.9	8.5	8.0	6.9	4.5	2.4	1.6
Reach 3	2.0	1.4	0.4	3.0	2.5	4.0	4.3	5.2	7.4	5.6	3.6	2.5
Reach 4	2.3	1.4	0.8	3.1	4.1	7.2	9.7	10.8	10.0	7.2	4.5	3.3
Reach 5	3.0	2.1	1.9	5.9	10.6	17.0	23.4	23.9	18.4	12.5	7.6	5.0

Figure 3.11: Simulation of the Stream Depletion (12 months)



Units: dS/cm

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Upstream	1.35	1.15	0.95	0.81	0.76	0.62	0.66	0.63	0.68	0.95	1.04	1.13
Reach 1	1.47	1.34	1.09	0.94	0.82	0.67	0.72	0.72	0.85	1.07	1.16	1.25
Reach 2	1.45	1.39	1.19	1.07	0.87	0.71	0.77	0.79	0.97	1.14	1.21	1.29
Reach 3	1.41	1.38	1.21	1.14	0.91	0.74	0.80	0.84	1.03	1.17	1.21	1.29
Reach 4	1.59	1.54	1.40	1.37	1.05	0.85	0.93	1.04	1.32	1.46	1.47	1.51
Reach 5	1.60	1.54	1.43	1.41	1.11	0.91	1.00	1.13	1.37	1.49	1.49	1.55

Note: The EC values represent salinity in the end of each river reach.

Figure 3.12: Simulation of the River EC (12 months)

The quantity of the return flow shown in figure 3.9 is the summation of the different kinds of return flows in each reach. For example, the return flow in June in reach 4 is a total of 11,890 acre-ft. This amount consists of: (a) 1,090 acre-ft of canal transit loss from canals 1, 3 and 4 from March to June; (b) 2,110 acre-ft of irrigation tail water from canal areas 1, 3 and 4 in June; (c) 2,530 acre-ft of ground water return flow due to surface water irrigation from canal areas 1, 3 and 4 from March to June; (d) 120 acre-ft of ground water return flow due to ground water irrigation from canal areas 1, 3 and 4 from January to June; (e) 6,030 acre-ft of river channel loss from river reaches 2, 3 and 4 from January to June.

The quality of the return flow shown in figure 3.10 is calculated based on a mixture of the different types of return flows. Concentrations of *Ca, Mg, Na, K, Cl, Alkalinity* and SO_4 are re-calculated for the mixture. Then, *EC* of the return flow is calculated based on this new chemical composition.

As shown in figure 3.9, return flows vary by month. There are more return flows during the irrigation season (April to October). This is mainly because of irrigation tail water which only occurs during the irrigation season. Since the salinity level of the irrigation tail water is lower than other kinds of return flows, it also makes the total return flows less saline during the irrigation season as shown in figure 3.10. During the non-irrigation season (November to March), the return flow mainly consists of groundwater flows making return flows significantly more saline during the non-irrigation season.

The amount of return flow for each reach is based on irrigation practices and watershed characteristics. The former determines how much excess agricultural water is created while the latter determines how the excess water returns to each river reach spatially and temporally. In this particular example, a significant amount of excess water is created in upstream areas which flows to downstream reaches. This makes the return flow in reach 5 higher than the return flows in other reaches.

As shown in figure 3.11, the amount of stream depletion gradually increases from a low in January. It attains its maximum value in July, then gradually declines until December. The results reflect the consumptive use of phreatophytes and the depletion effect of ground water pumping in the saturated aquifer, which is hydraulically connected with the river system. Figure 3.12 shows river salinity after calculating return flows and depletion. As shown in the figure, river salinity is lower during the irrigation season and higher during the off season. And, due to salts carried by return flows, downstream reaches have higher salinity levels than upstream reaches.

3.4 Computer Program for River Salinity Simulation

3.4.1 Programming Concept

Programming tasks for river salinity simulation include developing code for salinity computation and for a graphical user's interface (GUI). The two parts are developed separately and then integrated together. C/C++ was selected as the programming language because it allows memory address manipulation, supports object oriented programming and can utilize Microsoft Foundation Classes as Windows interface tools. The importance of these features are discussed below.

Object oriented programming (OOP) is a programming principle which combines structured programming with the concepts of encapsulation, polymorphism and inheritance (Schildt, 1998a). These attributes are designed for managing larger and more complex programs. Under OOP principles, "classes" are created when writing a program. A class is a user-defined type that links functions and data. Then, "objects" for these classes are declared to operate the functions and data. Different classes are independent to each other. The river salinity simulation program is designed using the OOP principle. There are different classes created to perform different types of tasks such as simulating salinity, displaying the river network, inputting data and displaying river salinity. Whenever a specific class needs to be modified or a new class needs to be added, other classes remain unchanged. This greatly reduces the effort involved in maintaining this program. Furthermore, since all functions and data are localized within classes, they won't be accidentally altered or misused. The OOP design also increases a program's run time efficiency. This is because data transfer between different objects is carried out internally by passing the memory addresses of the data. There is no need to create intermediate files to do the read in / print out functions which are not secure.

Microsoft Foundation Classes (MFC) is a system of C++ classes designed to facilitate Windows programming. When designing a Windows program, programmers need to know how to use the Win32 API functions to make an operating system perform a task such as drawing a line or receive input from the screen. The advantage of using MFC is that it organizes many of these functions into over 200 classes so that programmers can use API functions via MFC, which makes Windows programming much easier. For example, if a programmer wants to create a dialog box in the program, the first thing to do is to create a class inherited from MFC's "CDialog" class since CDialog class encapsulates all API functions related to dialog box drawing. Then, based on the specific work this dialog box is being designed for, customized functions are added on and / or the CDialog's functions can be overridden (Schildt, 1998a).

Figure 3.12 briefly illustrates how the salinity simulation program is executed in a Windows environment. Like all Windows programs, the river salinity simulation program is executed on an event-driven basis (Prosise, 1999). An event could be a keystroke on a dialog box, a mouse click on a button or a command to repaint the window. When the operating system senses an event, it will send a "Windows message" such as WM_KEYDOWN, WM_LBUTTONDOWN to the program. The program will first put the message in a message loop and then dispatch it to an appropriate message handler

(Prosise, 1999). A message handler is a function designed to instruct the operating system to perform a task. For example, in this salinity simulation program, a message handler could ask the operating system to show output data on the screen, or to calculate river salinity. A message handler may need to incorporate other functions and data in order to complete its work. After the operating system receives instructions from the message handlers, it will perform the task accordingly. Then, the windows and menus shown on the screen will be re-drawn and users can take the next action based on the updated information shown on the screen.

3.4.2 Major Features of the Graphical User's Interface

A graphical user's interface is developed to provide a user-friendly environment for river salinity simulation. Figure 3.13 illustrates how users execute the simulation program through the user's interface in a Windows environment. Major features of the graphical user's interface include: (1) diagrammatic illustration of the river basin network, (2) dialog-based data input, and (3) graphical output display. These features are described below.

After the program is started, the user needs to open a project. By opening a project, input data will be loaded to the program. Then, a diagram which shows the river network will be displayed on the screen. An example of a five-canal river basin network is shown in figure 3.8. In the figure, irrigated areas, river reaches, river intakes and canals are all symbolized as links and nodes to facilitate further data editing.

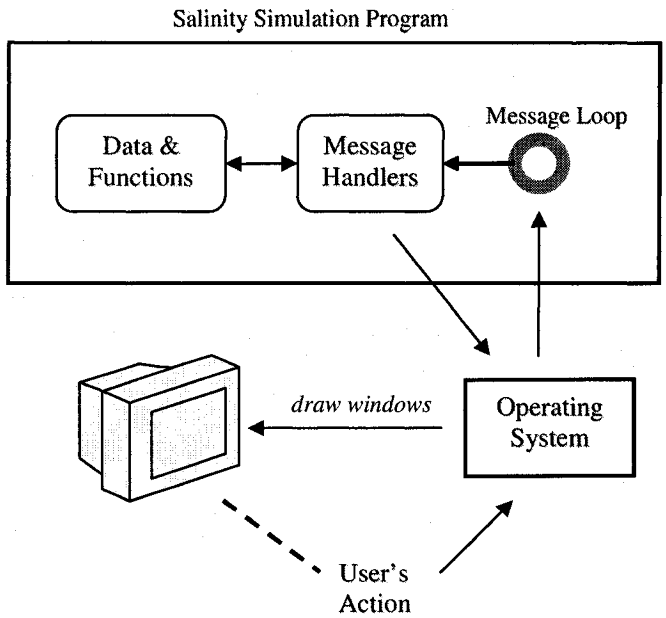


Figure 3.13: Execution of the Salinity Simulation Model in Windows Environment

The program organizes the input data into different "property sheets" (Schildt, 1998b). A property sheet is a virtual page containing several dialog boxes in which input data are displayed. Property sheets will pop-up when the mouse is clicked on the river basin network. If it is clicked on a canal area (red squares), all input data related to irrigation in this canal area will be shown. Likewise, corresponding input data will be shown when the mouse click is on a river reach (thick dark green lines), a canal (thin light green lines) or a river intake node (blue dots). Figure 3.14 is the property sheet for a canal service area. On that property sheet, there are seven dialog boxes: precipitation, GW pumping, crop consumptive use, soil treatment, GW return flows, SW return flows and stream depletion. Each dialog box could be activated for data editing.

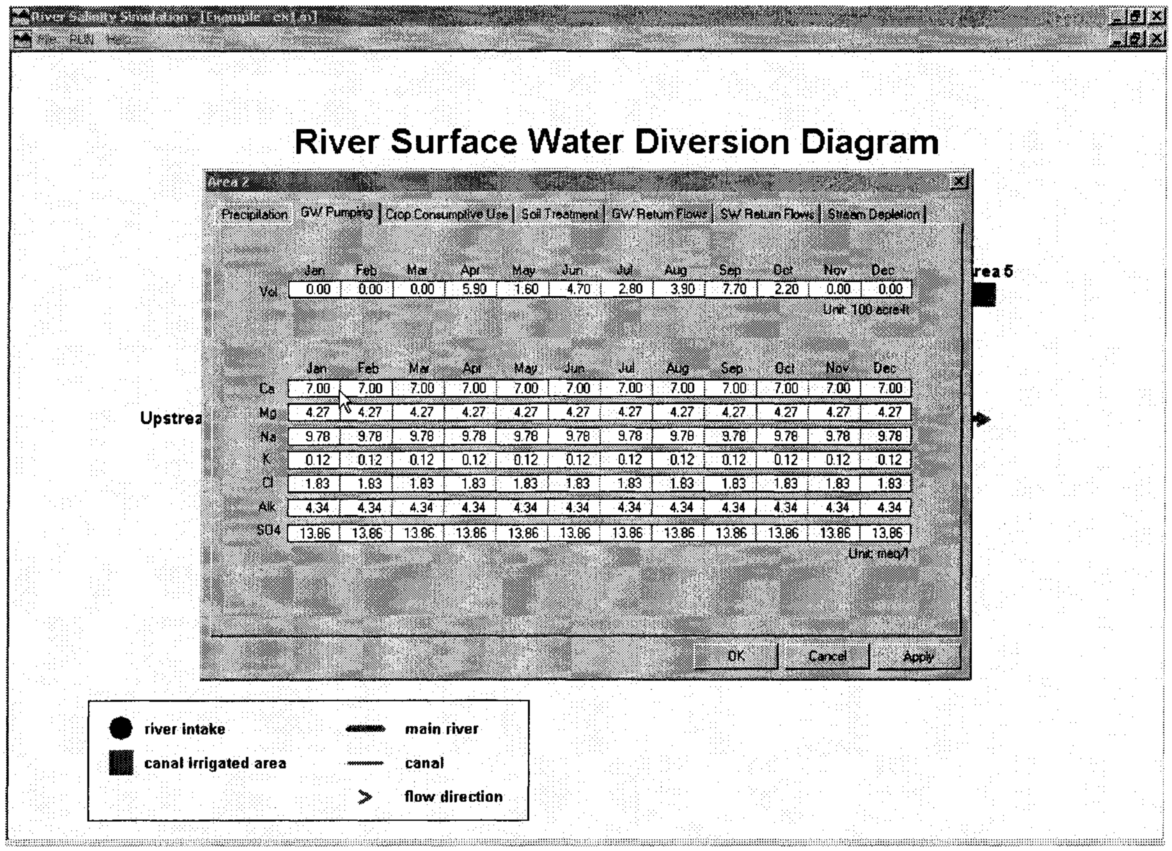


Figure 3.14: Input Data Editing Using Dialog Boxes

After editing of input data is completed, users can click on the "RUN" icon, located in the upper menu bar to execute the river salinity simulation. Depending on the size of the irrigated basin and the length of the simulation period, execution time varies. For a basin with five canals on a twelve-month simulation, it takes approximately 5 seconds to finish the simulation on a computer equipped with AMD Duron 600 MHz processor. After the river salinity simulation is completed, an output diagram for the simulated river basin will be displayed. By clicking on the diagram, users can view the simulation results. If the click is on a canal area (magenta squares), soil water quality of that canal area will be displayed. If the click is on a river location (magenta dots), river salinity at that location will be displayed. An alternative way to view the simulation results is to click on two items in the upper menu bar, "View Soil WQ" and "View River

Salinity. Figure 3.15 is an example to show an output diagram after the salinity simulation is executed.

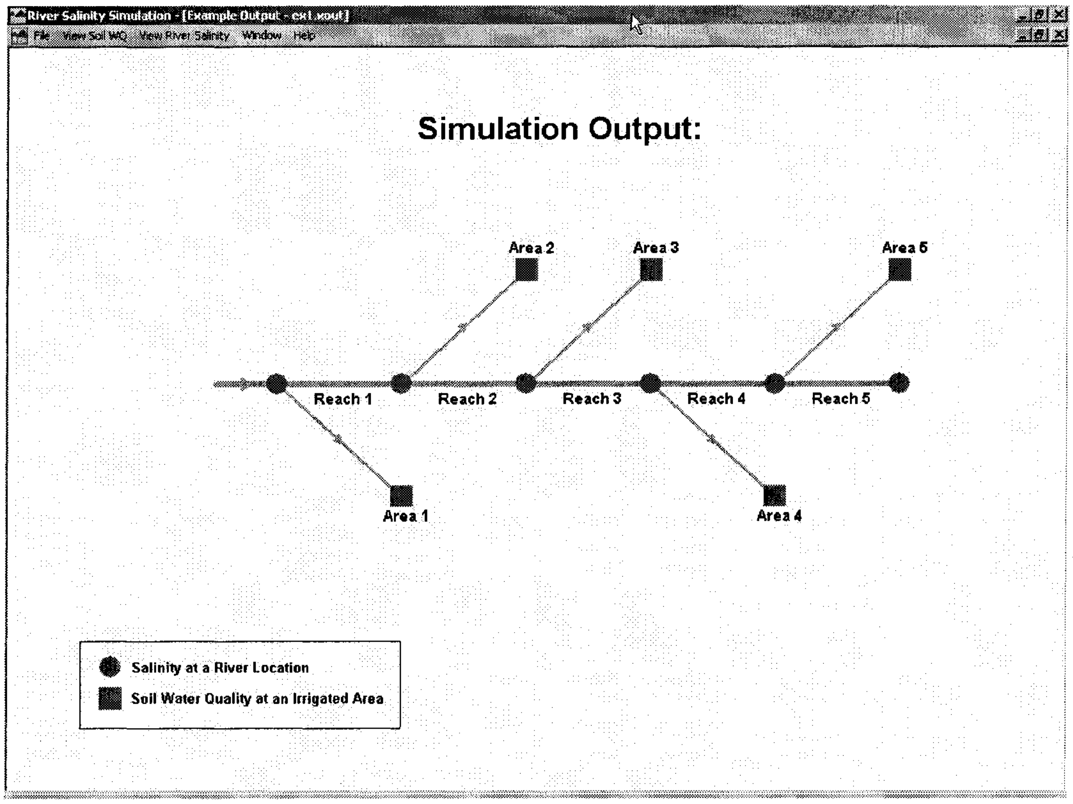


Figure 3.15: Output Data Display from the River Network

Figure 3.16 illustrates how soil water quality is displayed by the program. There are sixteen graphs which show different water quality parameters versus five soil depths. The parameters include: (a) concentrations of major cations and anions, (b) composite water quality indexes, and (c) major soil minerals. By making a selection from the list box shown in upper left corner, users can view soil water quality in a specific month. In addition, users can push the "Detailed Data" button to view detailed data used for plotting these graphs. The "Time Series" button is designed to display soil salinity from the first simulation month to the last simulation month continuously. With this feature, users can observe the seasonal fluctuation of soil water quality.

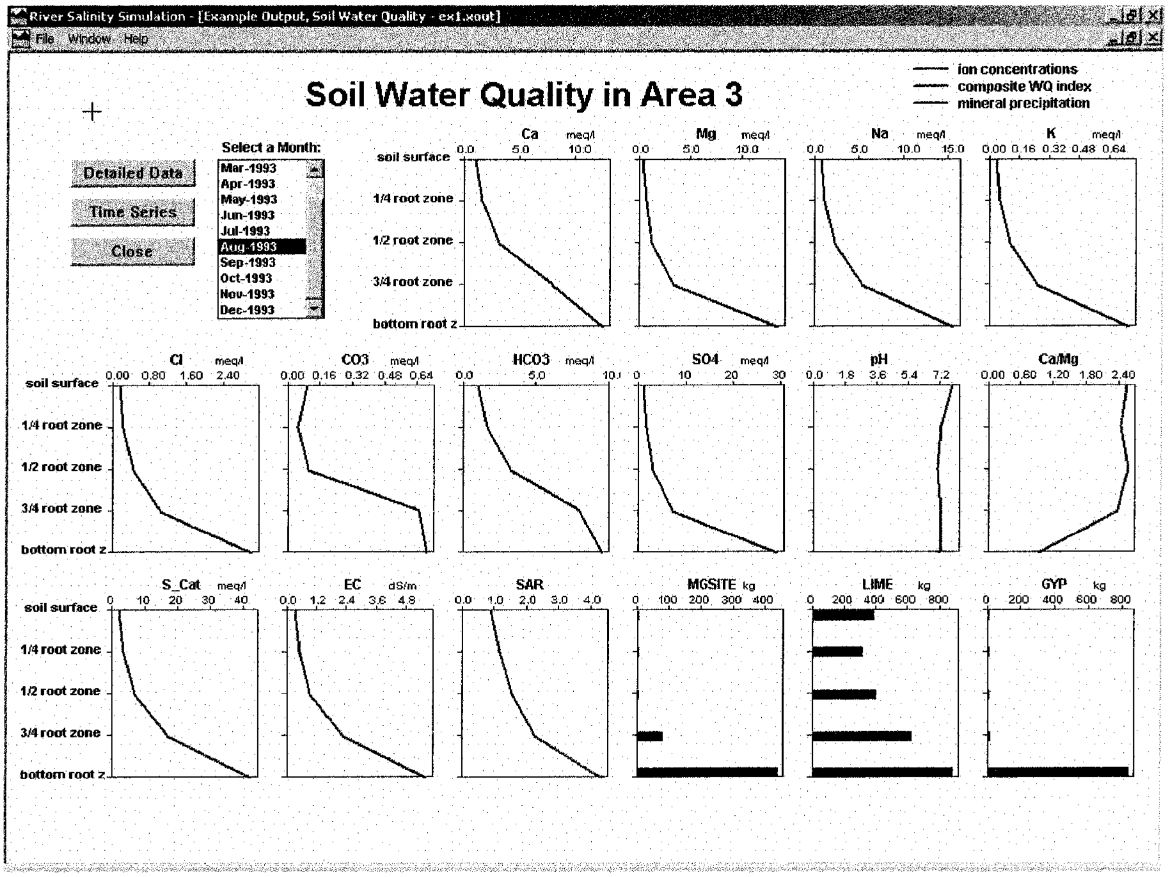


Figure 3.16: Soil Water Quality in a Canal Area

Figure 3.17 illustrates how river salinity is displayed by the program. There are nine graphs which show river quantity, EC, concentrations of Ca, Mg, Na, K, Cl, Alkalinity and SO_4 for the entire simulation period. In this example, the simulation period is from January 1993 to December 1993. This graph allows users to view seasonal fluctuations of river salinity at a certain river location. Users can push the "Detailed Data" button to view detailed data used for plotting these graphs. The "Entire River" button is designed to show river salinity at different river miles. When this button is pushed, figure 3.18 will be displayed. This figure allows users to observe river quantity and salinity changes from upstream to downstream. Also, users can view river salinity in a specific month or view river salinity as a time series.

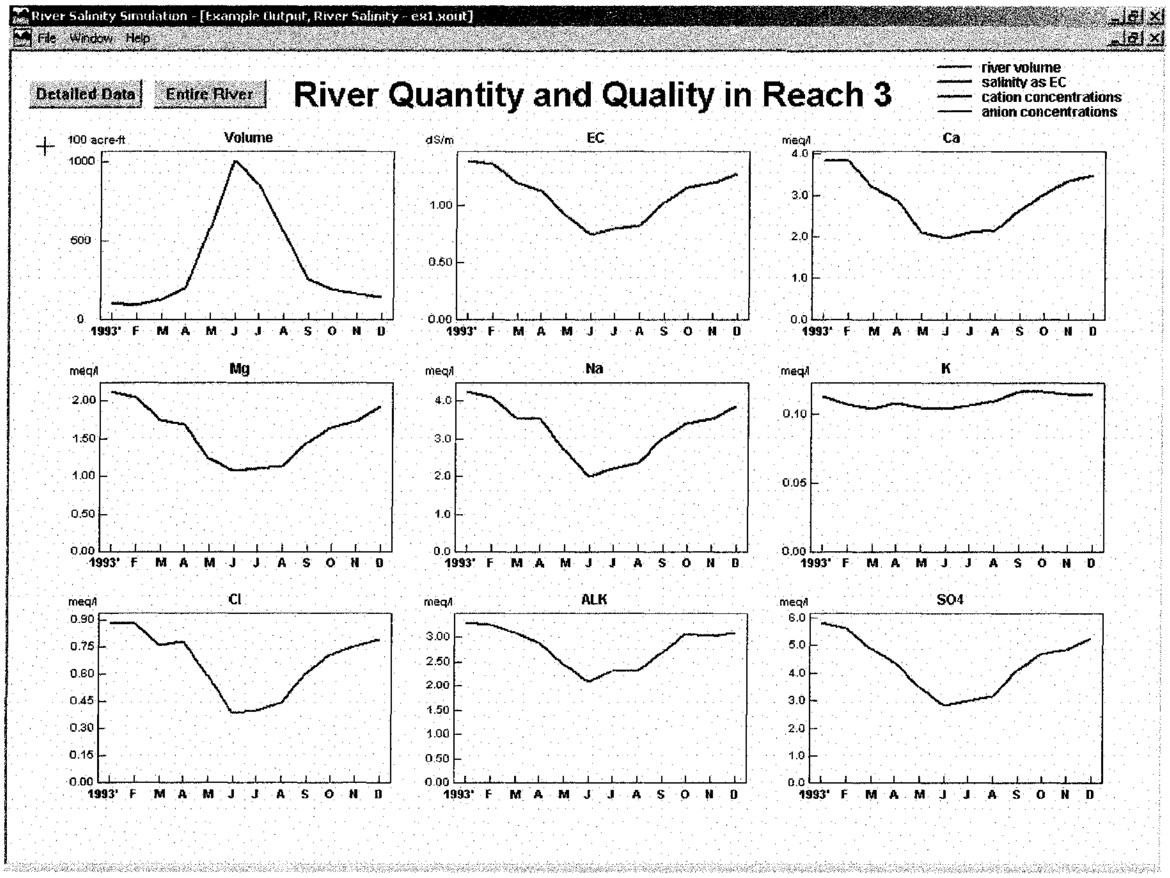


Figure 3.17: River Salinity in the End of a Reach

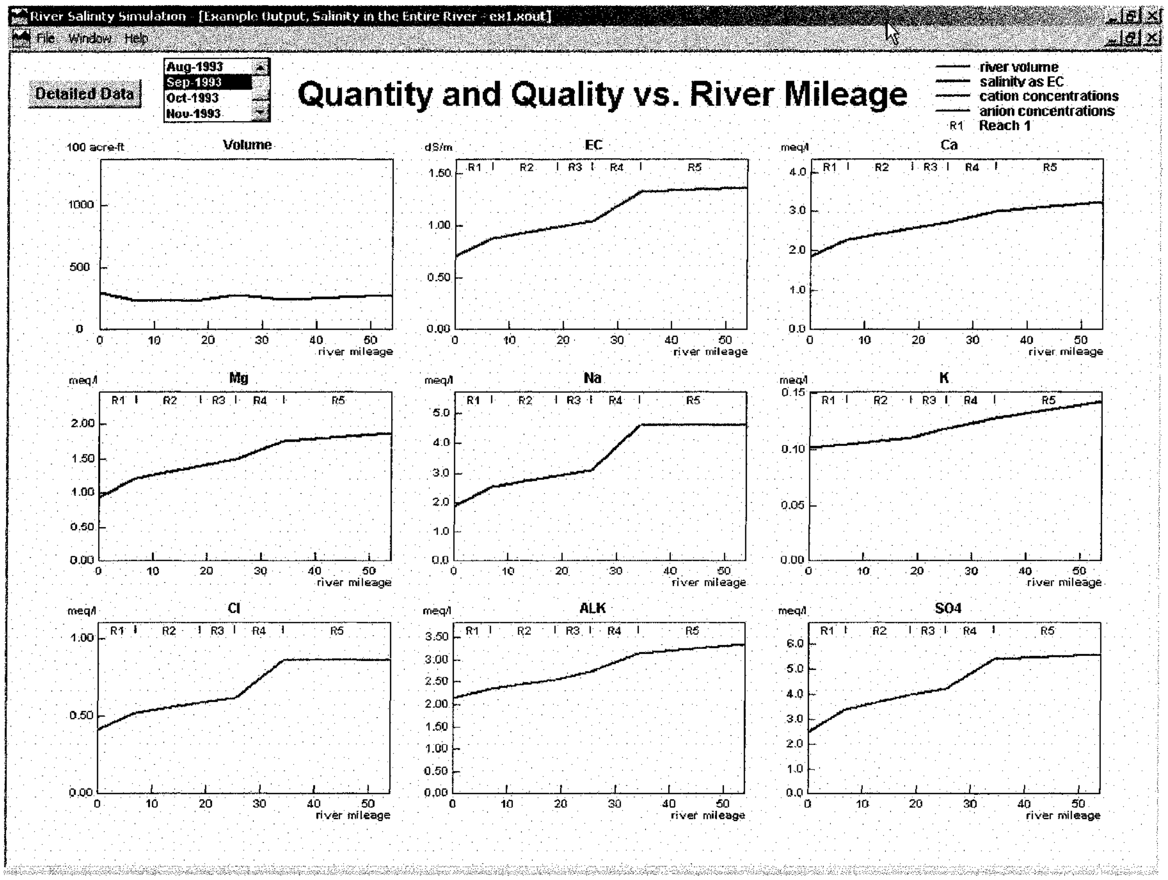


Figure 3.18: Salinity vs. River Miles

CHAPTER 4

IRRIGATION RETURN FLOW SIMULATION

In the previous chapter, irrigation return flows are expressed as response functions multiplied by volume of the hydrologic stress applied on the basin. In this chapter, the methodology for developing response functions is described.

4.1 Governing Equations and Assumptions

As described in the previous chapter, excess irrigation water will return to the river through either surface water or groundwater. The former is referred to as irrigation tail water which flows back to the river quickly through surface drains. The latter is referred to as deep percolation due to on-farm irrigation, canal transit loss or reservoir leakage. The deep percolated water turns into ground water and flows back to the river underground through porous media. This chapter will focus on the simulation of ground water return flows through porous media. The goal is to determine the spatial and temporal distribution of the ground water return flows.

Based on Darcy's Law (Bear, 1972), three-dimensional groundwater flows in isotropic porous media can be described as follows:

$$q_x = -K \cdot \frac{\partial \varphi}{\partial x} \quad (4.1)$$

$$q_y = -K \cdot \frac{\partial \varphi}{\partial y} \quad (4.2)$$

$$q_z = -K \cdot \frac{\partial \varphi}{\partial z} \quad (4.3)$$

where q_x, q_y and q_z are specific velocity of ground water (m/day) in the directions of the Cartsian *x, y* and *z* coordinates; *K* is hydraulic conductivity (m/day) that $K = K(x, y, z)$; φ is potential energy of water (m) that:

$$\varphi = \frac{p}{\gamma} + z + c \quad (4.4)$$

where p is hydrostatic pressure; γ is specific weight; z is elevation above a datum; c is any constant.

Vertical specific velocity, q_z, can be neglected if it is much smaller than the two horizontal specific velocities, q_x and q_y. This usually occurs in an aquifer that has a much larger lateral extent than aquifer thickness, such as a lateral extent that can be measured in kilometers versus a thickness that can be measured in tens of meters (Charbeneau, 2000). Since this study is focused on ground water flows in unconfined aquifers, the potential energy, φ, in an unconfined aquifer can be expressed as ground water table level, *h*. Therefore, $\bar{s}$ represents the direction of ground water flows that is: $\bar{s} = \bar{x} + \bar{y}$. The two-dimensional groundwater flows can be re-written as:

$$q_s = -K \cdot \frac{\partial h}{\partial s} \quad (4.5)$$

where q_s is specific discharge in the direction of $\bar{s}$.

The actual velocity of groundwater can be determined by the specific discharge divided by porosity of the media (Chow, 1988, p39), $\eta(x, y)$, that is:

$$v_s = \frac{q_s}{\eta} \tag{4.6}$$

Based on equation (4.5) and (4.6), with information on K, h and η, groundwater flow directions and flow velocity in the aquifer can be determined. Flow directions form flow paths. Therefore, when a unit hydrological stress is placed on a unit area in the aquifer, its return flow can be traced by the flow path in the aquifer. This return flow is a spatial and temporal function and is referred to as a "unit response function" in this study. For a stress covering a large area (more than just a unit area), its response function can be obtained by superimposing unit response functions covering this large area. Then, by multiplying this response function with the amount of the stress, a return flow function due to this specific stress can be simulated. With this methodology, return flow functions resulting from any stress such as surface water irrigation or canal transit loss can be simulated. Since the irrigation process is dynamic, the amount of water applied on each canal service area is different from month to month, as a result, the simulated return flow functions are also spatially and temporally varied.

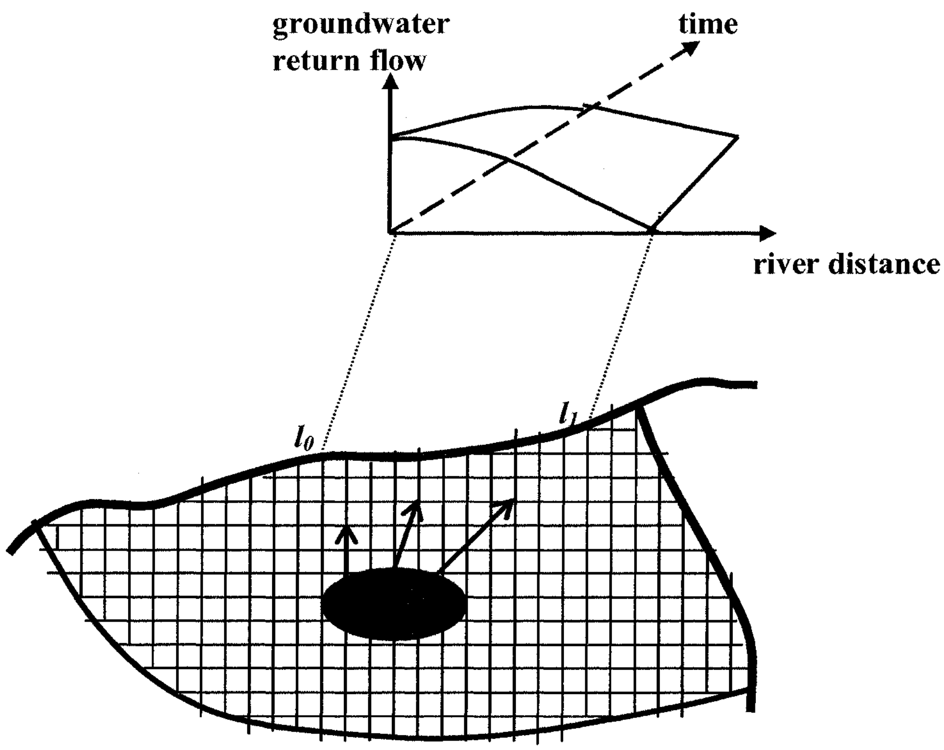
A detail procedure for generating response functions is demonstrated in the next section.

4.2 Response Function Generation

The procedure of applying two-dimensional groundwater flow to generate response functions for a stress is accomplished through the following steps: (a) discretize the aquifer into two-dimensional cells; (b) simulate groundwater flow paths; (c) simulate travel time for groundwater flow using a probability approach; and (d) superimpose individual unit response functions.

4.2.1 Groundwater Flow Path Simulation

Figure 4.1 illustrates how an aquifer is discretized for generating response functions. In the figure, the shaded area represents locations where a unit hydrological stress is placed. The stress covers several aquifer cells. As groundwater from these cells flows towards the river, it is intercepted by the river from an upstream location l_0 to a downstream location l_1. Groundwater from different cells returns to the river at different time steps. As a result, the response function can be represented as functions of "time" and "river distance".



▬ River; ━ Canal; ■ Stress area; □ Aquifer cells

→ Direction of groundwater return flow

Figure 4.1: Aquifer Discretization for Generating Response Functions

As shown in figure 4.1, a flow path for each cell covered by the stress should be simulated. As described in equation (4.5), the flow path for a "point" in the porous media is directed by the hydraulic gradient, $\frac{\partial h}{\partial s}$, at that point. Physically, $\frac{\partial h}{\partial s}$ represents the steepest slope. As the aquifer is discretized into cells, the steepest slope will then be substituted by the maximum slope surrounding a cell. Therefore, $\frac{\partial h}{\partial s}$ for the cell at the i^{th} row and the j^{th} column can be represented as:

$$\left.\frac{\partial h}{\partial s}\right|_{i,j} = \max\left(\frac{h_{i,j}-h_{i-1,j}}{\delta}, \frac{h_{i,j}-h_{i,j-1}}{\delta}, \frac{h_{i,j}-h_{i+1,j}}{\delta}, \frac{h_{i,j}-h_{i,j+1}}{\delta}, \frac{h_{i,j}-h_{i-1,j+1}}{\sqrt{2}\delta}, \frac{h_{i,j}-h_{i-1,j-1}}{\sqrt{2}\delta}, \frac{h_{i,j}-h_{i+1,j+1}}{\sqrt{2}\delta}, \frac{h_{i,j}-h_{i+1,j-1}}{\sqrt{2}\delta} \right) \quad (4.7)$$

where $h_{i,j}$ represents the average groundwater table elevation for cell *(i,j)*; δ represents the length (and also the width) of a cell. The average distance between two cells is δ for two adjacent cells and is $\sqrt{2}\delta$ for two diagonal cells.

After the hydraulic gradient is determined for each cell, the flow path of each cell can be simulated. Figure 4.2 illustrates how the flow path is simulated. As shown in the figure, a stress is placed on a cell and it results in groundwater flow towards the river. The flow path can be simulated by seven aquifer cells. Theoretically, if the aquifer is discretized into an infinite amount of cells, the simulated flow path should converge to the actual flow path.

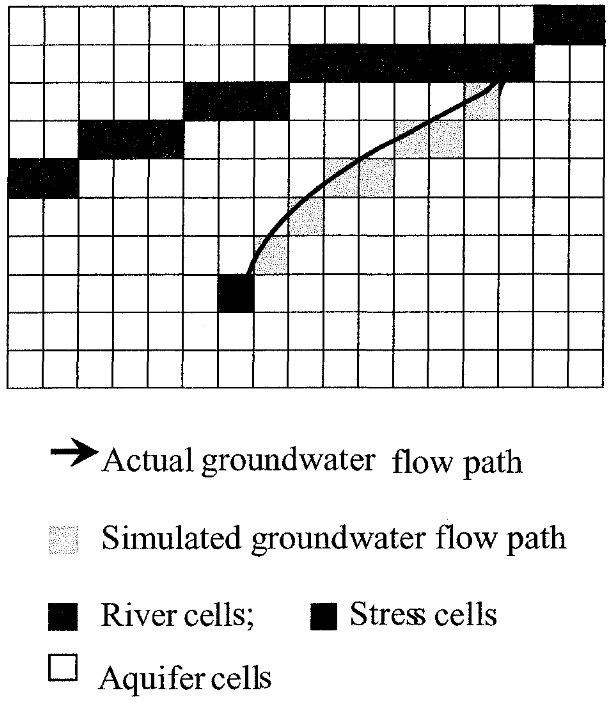


Figure 4.2: Actual vs. Simulated Groundwater Flow Path

4.2.2 Groundwater Travel Time Simulation with a Probabilistic Approach

In the previous section, methodology for simulating groundwater flow paths was illustrated. This section is focused on simulating groundwater travel time for a groundwater flow path. The approach used in this research is to calculate travel time between any two cells along a flow path, then, accumulate the travel time from the starting cell to the ending cell. The travel time between two cells can be obtained by dividing the distance between them by their flow velocity. The distance between two cells is δ for two adjacent cells and is $\sqrt{2\delta}$ for two diagonal cells as shown in equation (4.7). The velocity for flow between two cells can be approximated by equation (4.6).

This approach is valid for the situation where groundwater always stays in the porous media and never turns into surface flow on its way towards the river. However, in an irrigated region, farmers usually install drainage devices on their farms to control seepage. This is especially popular in areas affected by waterlogging. Drainage devices shorten the travel time for groundwater return flows. This is because a portion of the groundwater return flow is intercepted by drainage devices such as tile drains or drainage ditches. This portion of the water is transported at a much faster velocity in a drainage device than in the porous media. For this common situation, a probability distribution is derived for this study. This probability distribution simulates travel time of groundwater return flow.

It is assumed that groundwater drained by drainage devices in a field will recharge into the aquifer again. This assumption is based on the fact that farmers only install drainage devices for their own farms. The drained water is usually released down slope in the vicinity of their farms. Therefore, the drained water will recharge into the aquifer, join the groundwater system again and continue to flow towards the river. Then, a portion of the drained water may be intercepted by other drainage devices on the way to the river. To simulate this process, groundwater flowing through a cell is divided into two portions: drained and not drained. For the portion of groundwater that is drained, the travel time through this cell is assumed to be 0, and this portion of water is placed on the next cell in the flow path of the groundwater. For the portion that is not drained, the travel time is calculated by using equation (4.6). By using this approach on all the cells in a flow path, travel time for groundwater return flow can be simulated. As a result, travel time for groundwater flow though a cell can be expressed as a probability distribution.

Consider a unit stress placed on the aquifer. The groundwater caused by this stress flows through n cells to the river. Let α_i represent the percentage of the groundwater removed by drainage devices in the i^{th} cell, $0 \le \alpha_i \le 1$, i represents the order of the cell in the flow path, i=0,1,2,…,n. Let travel time for the portion of groundwater not drained by a drainage device in the i^{th} cell be t_i. For groundwater flows from the 1^{st} cell to n^{th} cell, the travel time can be expressed as the following multi-variable probability distribution:

$$f(X) = \begin{cases} \left|\prod_{i=1}^{n}(\alpha_i - k_i)\right|, & for \quad x_i = t_i \cdot k_i, \quad k_i = 0,1, \\ 0, & otherwise \end{cases} \tag{4.8}$$

$$T = \sum_{i=1}^{n} x_i \tag{4.9}$$

In equation (4.8), X is a random vector. $X' = [x_1, x_2, ..., x_n]$. x_i is the groundwater travel time in the i^{th} cell. X represents different outcomes of the distribution. $f(X)$ is the probability distribution for X. k_i is a parameter to indicate the outcome of X. While k_i equals 0, x_i equals 0 which represents the groundwater that is drained in the i^{th} cell. While k_i equals 1, x_i equals t_i which represents the groundwater that is not drained in the i^{th} cell. T represents the total travel time for groundwater flowing from the 1^{st} cell to the n^{th} cell.

By setting the parameter α_i in equation (4.8), the influence of drainage devices on groundwater can be simulated. For the cell where no drainage pipes or drainage ditch is present, α_i can be set as 0. Table 4.1 illustrates how to calculate groundwater travel time with this probability approach with an example. In this example, groundwater flows through three cells (n=3). Drainage parameters are: $\alpha_1 = 0.3$, $\alpha_2 = 0.4$, $\alpha_3 = 0.2$. Travel time parameters are: $t_1 = 25$, $t_2 = 15$, $t_3 = 10$ (units: days). As shown in table 4.1, with different combinations of k_i, eight outcomes are generated with different probabilities. In this example, a stress placed on the specific cell will result in 33.6%, 8.4%, 5.6%, 14.4%, 3.6%, 9.6% and 2.4% of its water returning to the river in 50, 40, 35, 25, 25, 15, 10 and 0 days, respectively. To illustrate how $f(x_1, x_2, x_3)$ is calculated in table 4.1 in more detail, consider the probability of the outcome that groundwater flow is drained in the 1^{st} cell and the 3^{rd} cell but not drained in the 2^{nd} cell. This makes $k_1=0$, $k_2=1$ and $k_3=0$. Then, the travel time x_i for the three cells will be $x_1=0$, $x_2=t_2=15(days)$ and $x_3=0$ according to equation (4.8). So, the total groundwater travel time can be calculated as: $T=x_1+x_2+x_3=15$ *(days).* The probability for $x_1=0$, $x_2=15$ and $x_3=0$ can be calculated as: $f(0,1,0)=|\alpha_1 \cdot (\alpha_2 - k_2) \cdot \alpha_3| = |0.3 \cdot (0.4-1) \cdot 0.2| = 0.036$.

Table 4.1: Probability of Groundwater Travel Time

k_1	1	1	1	1	0	0	0	0
k_2	1	1	0	0	1	1	0	0
k_3	1	0	1	0	1	0	1	0
x_1	25	25	25	25	0	0	0	0
x_2	15	15	0	0	15	15	0	0
x_3	10	0	10	0	10	0	10	0
$f(x_1, x_2, x_3)$	0.336	0.084	0.224	0.056	0.144	0.036	0.096	0.024
GW Travel Time, T	***50***	***40***	***35***	***25***	***25***	***15***	***10***	***0***
GW Percentage	***33.6***	***8.4***	***22.4***	***5.6***	***14.4***	***3.6***	***9.6***	***2.4***

Units for x_1, x_2, x_3 and GW travel time are days

4.2.3 Response Function Superimposition and Computer Programming

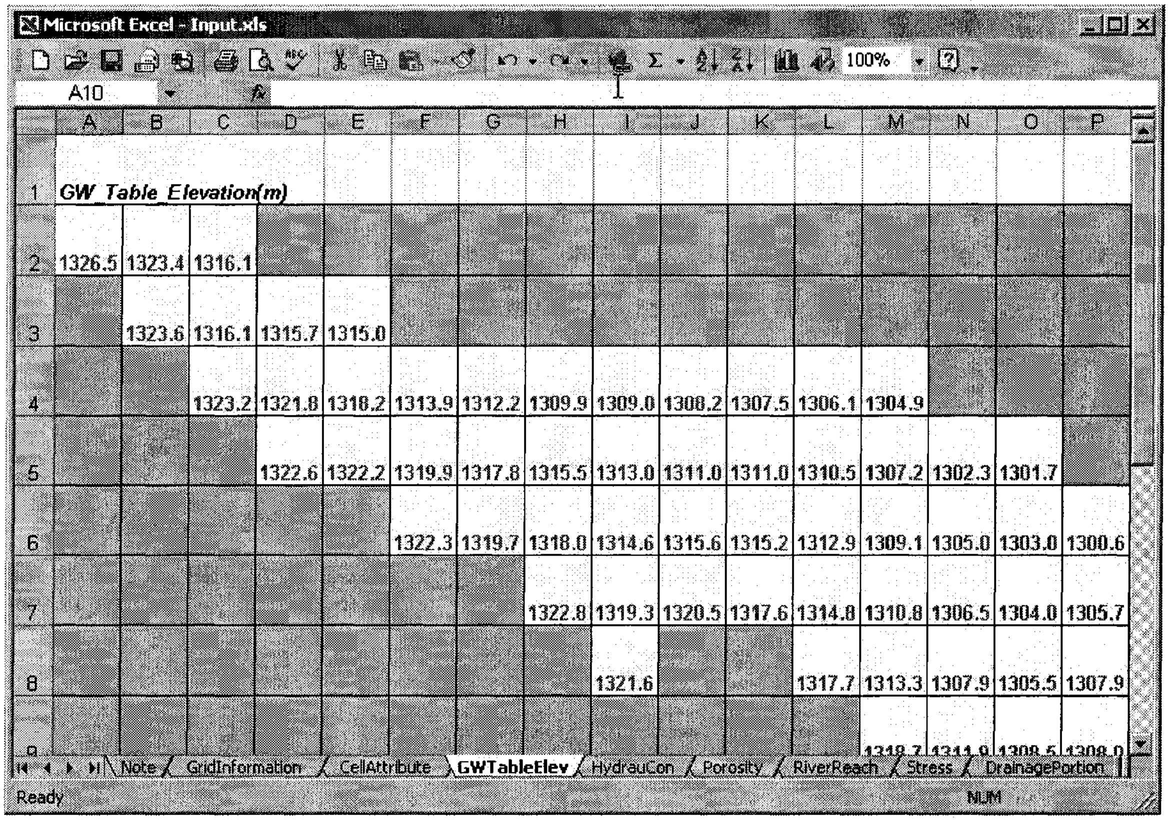
By using equation (4.8) and (4.9), groundwater travel time distribution for a stress placed on an aquifer cell can be simulated. Then, this distribution needs to be re-written in the form of a response function, which is the return flow percentage in different time steps and in different river reaches. The general form of a response function can be described as $g(j,s)$ that:

$$\sum_j \sum_s g(s,j) = 1 \tag{4.10}$$

where s and *j* represent the time period and the river reach this return flow occurs in. For example, the return flow shown in table 4.1 is to be re-written as 64.4% in month 1 for reach 1 and 35.6% in month 2 for reach 1.

Different response functions can be superimposed and form a new response function. This skill is applied for generating response functions for stresses which cover vast areas. With this skill, response functions for stresses such as irrigation deep percolation, canal leakage, and reservoir leakage can be generated. By multiplying the amount of the stresses with the response function, spatial and temporal distribution of groundwater return flows can be determined. A similar methodology is applied for calculating river depletions due to well pumping. The only difference is that the spatial and temporal distribution for river depletions is considered as negative return flow.

A computer program using the C/C++ language was implemented for simulating response functions based on the aforementioned concept and algorithm. The program is shown in Appendix F. The aquifer must be discretized into two-dimensional cells. Each cell has multiple data including: groundwater elevation, hydraulic conductivity, aquifer porosity and drainage parameters. These data are edited in a spreadsheet (as shown in figure 4.3) then are imported into the program. By using this program, response functions for irrigation percolation, canal leakage, reservoir leakage and well pumping can be generated.



	A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P
1	*GW_Table_Elevation(m)*															
2	1326.5	1323.4	1316.1													
3		1323.6	1316.1	1315.7	1315.0											
4			1323.2	1321.8	1318.2	1313.9	1312.2	1309.9	1309.0	1308.2	1307.5	1306.1	1304.9			
5				1322.6	1322.2	1319.9	1317.8	1315.5	1313.0	1311.0	1311.0	1310.5	1307.2	1302.3	1301.7	
6						1322.3	1319.7	1318.0	1314.6	1315.6	1315.2	1312.9	1309.1	1305.0	1303.0	1300.6
7								1322.8	1319.3	1320.5	1317.6	1314.8	1310.8	1306.5	1304.0	1305.7
8									1321.6			1317.7	1313.3	1307.9	1305.5	1307.9
9													1318.7	1311.9	1308.5	1308.0

Figure 4.3: Data Pre-Processing in a Spreadsheet for Response Function Generation

CHAPTER 5

RIVER SALINITY MODEL CALIBRATION

In this chapter, the salinity model described in previous chapters is applied to the Arkansas River Basin in Colorado. Input data and parameters will be set up in the model. Then, river salinity records from 1986 to 1990 will be used to calibrate the model. Finally, this model will be validated for its accuracy.

5.1 Description of the Study Area

5.1.1 Water Distribution System

The location of the study area, the Arkansas River Basin in Colorado, is shown in figure 5.1. The study area covers all agricultural lands which rely on the Arkansas River for irrigation. The upstream boundary of the study area is Pueblo Reservoir and the downstream boundary is the Kansas state line. Five counties are involved in this study area: Pueblo, Crowley, Otero, Bent, and Prowers.

There are twenty-one major canals in this study area which divert water to feed a total of 313,867 acres of irrigated fields (1985 data). Figure 5.2 shows the relative locations of the major canals. Water allocation between the canals is based on the water right priority doctrine (Grigg, 1996). Table 5.1 lists the water right decrees owned by each canal.

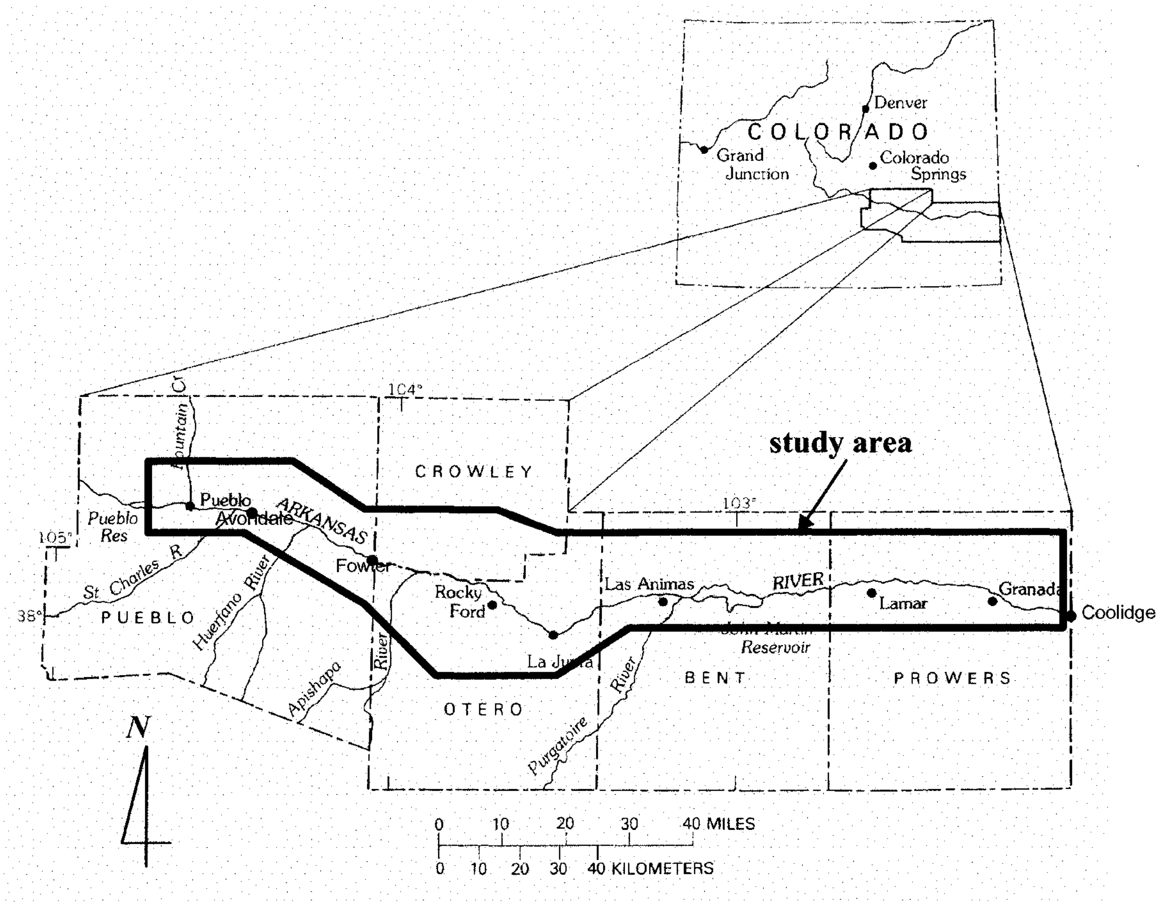


Figure 5.1: Location of the Study Area

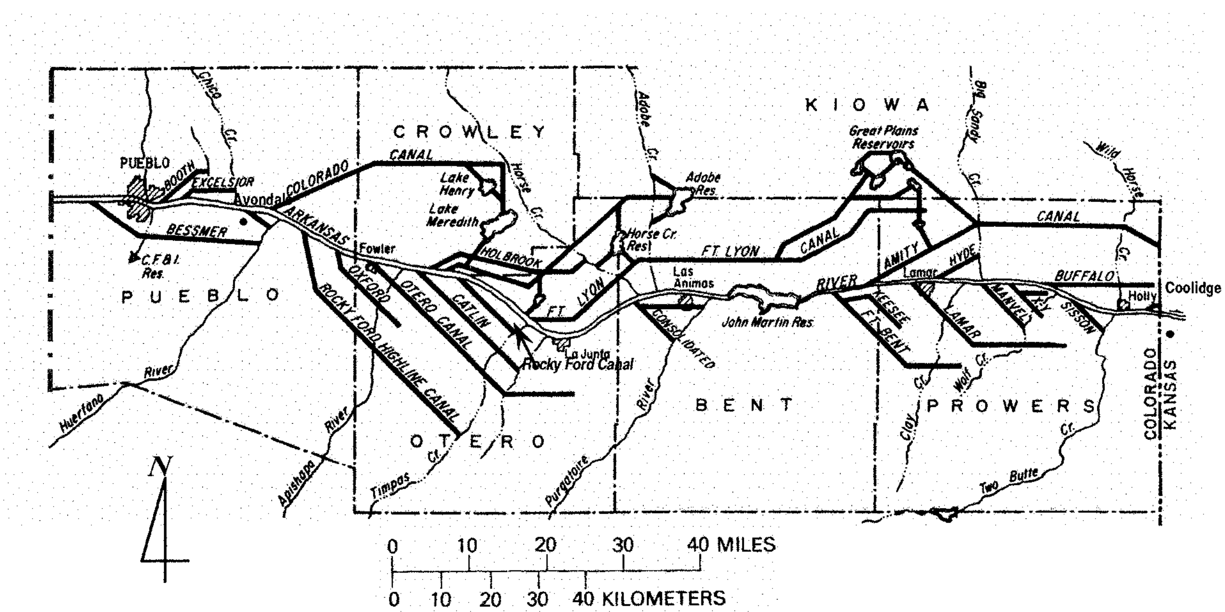


Figure 5.2: Canal System in the Study Area (from Moulder and Jenkins, 1969)

Table 5.1 Canal Length and Water Rights in the Study Area (summarized from Abbott, 1985)

Canal Name	Canal length (100 feet)	Diversion Right (cfs), Decree Date	Total Right (cfs)
Bessemer Canal	1500	2.0, 30APR1861; 20, 31DEC1861; 3.74, 31MAY1864; 3.0, 30JUN1866; 2.5, 08JAN1867; 5.13, 31MAY1867; 1.47, 30NOV1870; 3.4, 31DEC1870; 2.0, 18SEP1873; 3.0, 31DEC1876; 0.41, 31DEC1876; 14, 31DEC1878; 2.0, 20JUN1881; 8.0, 31MAR1882; 322, 01MAY1887	392.55
Booth Orchard Canal	500	7.0, 01APR1861; 8.0, 01APR1864; 1.0, 01JAN1871; 2.0, 01JAN1881; 2.1, 30MAR1888; 3.2, 15APR1893; 7.0, 31DEC1894	82.36
Excelsior Canal	417	20, 01MAY1887; 40, 06JAN1890	60.00
Collier Canal	250	4.0, 10MAR1887; 22.0, 01MAY1887	26.00
Colorado Canal	2083	756.28, 09JUN1890	756.28
Rocky Ford Highline Canal	3417	40.0, 31DEC1861; 0.6, 01SEP1867; 16.0, 01JUL1869; 30.0, 30JUN1885; 2.0, 11MAR1886; 2.5, 06JAN1890; 378, 06JAN1890	501.60
Oxford Canal	750	13.4, 01SEP1867; 116, 26FEB1887	129.40
Otero Canal	2000	123, 03MAR1890; 334.92, 02FEB1903	457.92
Catlin Canal	1083	248, 03DEC1884; 97.0, 14NOV1887	345.00
Fort Lyon Canal	4500	164.64, 15APR1884; 597.16, 01MAR1887; 171.2, 31AUG1893	933.00
Rocky Ford Canal	667	111.76, 15MAY1874; 96.54, 06MAY1890	208.30
Holbrook Canal	1250	155, 25SEP1889; 445, 30AUG1893	600.00
Las Animas Consolidated Canal	833	38, 07MAR1884; 44.3, 03DEC1884; 80, 13MAR1888; 44.8, 15APR1909	129.50
Fort Bent Canal	1250	27.77, 01APR1886; 32.77, 10MAR1889; 11.7, 11SEP1889; 26.77, 12AUG1890; 50, 01JAN1893; 80, 31DEC1900; 85, 31DEC1980	217.31
Keese Canal	333	9, 31MAR1871; 4.5, 31DEC1883; 15, 03SEP1893; 18, 31DEC1980	28.50
Amity Canal	3583	283.5, 21FEB1887; 500, 01APR1893; 395, 31DEC1980	783.50
Lamar Canal	1000	15.75, 30NOV1875; 72.09, 04NOV1886; 13.64, 16APR1887; 184.27, 16JUL1890	297.45
Hyde Canal	333	23.44, 10MAY1887; 8, 31DEC1980	
Manvel Canal	500	1, 14OCT1890	54.00
XY Graham Canal	750	69, 22JUL1889; 61, 24AUG1891; 38, 31DEC1980	130.00
Buffalo Canal	917	67.5, 29JAN1885; 80, 31DEC1980	67.50
Sisson Canal	250	18, 01DEC1890; 7.2, 01DEC1895; 10, 31DEC1980	32.74

There is a major reservoir, John Martin Reservoir, located in Bent County on the Arkansas River (figure 5.1). This reservoir was built in 1943 for water users in Colorado and Kansas. Its storage capacity is 655,000 acre-ft. Operation of John Martin is based on an agreement between Colorado and Kansas referred to as the "Arkansas River Compact". The Compact stipulates how the stream inflow should be stored and be diverted to each state (Abbott, 1985).

In addition to John Martin Reservoir, there are 7 off-stream reservoirs in the Arkansas River Basin. These reservoirs are owned by canal companies and are used for off-season storage. Lake Henry, Lake Meredith, the Holbrook Reservoir, and Dye Reservoir are owned by the Holbrook Canal. Horse Creek Reservoir and Adobe Reservoir are owned by the Fort Lyon Canal, and the Great Plains Reservoir is owned by the Amity Canal. Lake Meredith, Dye Reservoir, and Holbrook Reservoir are topographically too low to divert water to their canals. Therefore, water exchange with downstream users is necessary for owners to utilize water in these reservoirs.

In addition to surface water, groundwater is also used for irrigation. Before the 1950's, groundwater pumping was less than 38,000 acre-ft per year. After the 1950's, due to a growing demand for water, groundwater pumping increased to nearly 120,000 acre-ft per year (Abbott, 1985). The groundwater is pumped primarily from the high-yield alluvial aquifer. Due to the fact that the aquifer and the river system are hydraulically connected, groundwater pumping has actually depleted stream flow and injured senior surface water rights in both Colorado and Kansas. As a result, Colorado State Water Court curtails groundwater pumping in the Arkansas River unless water users have water rights prior to 1948 (District Court Water Diversion No.2 State of Colorado, 1994).

5.1.2 Geologic and Hydro-salinity Characteristics

The geologic units in the study area include unconsolidated deposits and several kinds of shale and limestone. Their formation age, thickness, physical and hydrologic characteristics are summarized in table 5.2. As shown in the table, the valley-fill deposits (alluvium) contain a mixture of unconsolidated gravel, sand, silt and clay and are a major porous media for groundwater flow.

Table 5.2: Characteristics of the Geologic Units in the Arkansas River Valley (from Major et al. 1970)

Series	Geologic unit	Thickness (feet)	Physical character	Hydrologic character
Holocene and Pleistocene	Dune sand	0-100	Very fine to coarse, poorly sorted sand.	Commonly not saturated, but transmits water readily from the surface to underlying aquifers. Source of water for a few domestic and stock wells.
	Valley-fill deposits	0-300	Boulders, cobbles, gravel, sand, silt, and clay. Generally grades from fine sand near the surface to coarse sand and gravel at the base.	Principal source of water for irrigation, public supply, and industrial wells. Irrigation-well yields are as much as 3,150 gpm (gallons per minute) and average 650 gpm. Aquifer furnishes water to 1,348 irrigation wells.
Upper Cretaceous	Pierre Shale	0-2,200	Shale and sandy shale.	Low-permeability confining bed; acts as a barrier to vertical movement of ground water. Not known to yield water to wells.
	Niobrara Formation	0-700	Chalky and marly limestone and calcareous shale.	Low-permeability confining bed; acts as a barrier to vertical movement of ground water. A few stock wells tapping fractured limestone yield less than 5 gpm.
	Carlile Shale	0-200	Calcareous shale, limestone, and sandstone.	Low-permeability confining bed; acts as a barrier to vertical movement of ground water. Not known to yield water to wells.
	Greenhorn Limestone	0-150	Limestone and chalky shale.	Low-permeability confining bed; acts as a barrier to vertical movement of ground water. A few stock wells tapping fractured limestone yield less than 5 gpm.
	Graneros Shale	0-200	Gypsiferous shale and sandstone.	Low-permeability confining bed; acts as a barrier to vertical movement of ground water. Not known to yield water to wells.
Lower Cretaceous	Dakota Sandstone	75-235	Sandstone, sandy shale, siltstone, and shale.	Important source of water for domestic, stock, and public supply wells. Restricts vertical movement of water to and from the valley-fill deposits. Wells yield as much as 100 gpm and average 20 gpm.

The extent of the alluvial aquifer, based on a U.S. Geological Survey using pumping wells (USGS 1989a, b and c), is a strip along the river ranging between 1 and 14 miles wide. Boyle Engineering Corporation (1990) claims that the extent of the alluvial aquifer should also include the shallower "surfacial aquifer," which extends from the strip along the river to the outermost boundaries of the major canal systems. Generally, the alluvial aquifer forms a U-shaped trough above the low-permeability bedrock. Figure 5.3 illustrates the profile of the alluvial aquifer in Otero County near the city of La Junta. This figure is based on the soil survey of Otero County (USDA, 1972). Since the permeability of bedrock beneath the alluvial aquifer is low, groundwater is confined in the alluvium.

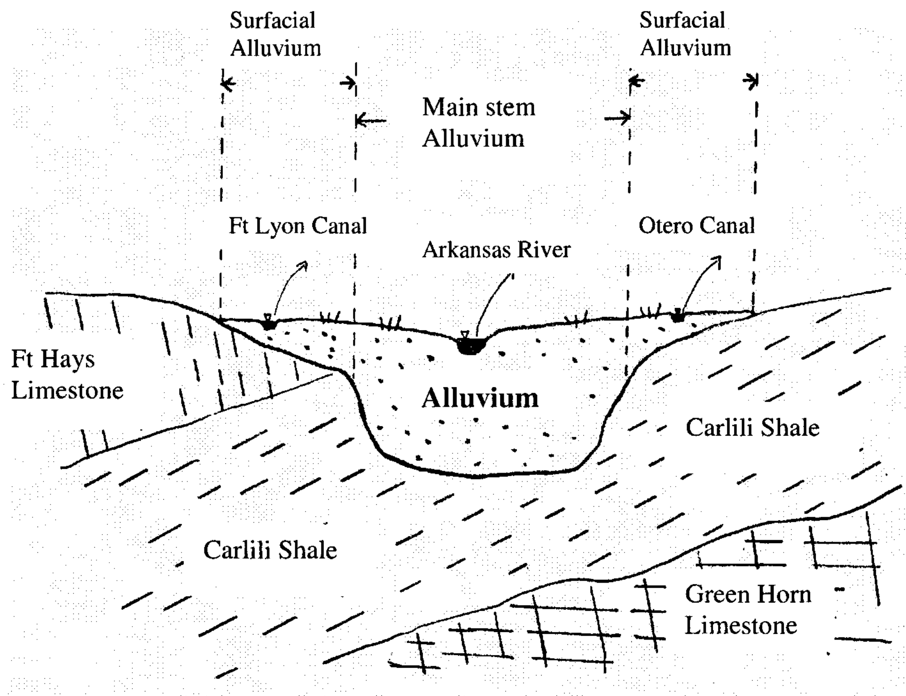


Figure 5.3: Geologic Profile in the Arkansas River near the City of La Junta

Transmissivity of the alluvium varies spatially. Wilson's research (1965) from 55 locations showed that transmissivity ranges from 517 to 15,333 gal/day/ft^2. Gates, et al. (2002) measured hydraulic conductivity, another index of aquifer transmissivity, in Otero County. Their results showed that hydraulic conductivity ranges from 0.003 to 10.24 m/day.

The alluvial aquifer is abundant in minerals. As minerals are dissolved, groundwater salinity increases. Based on a survey conducted by the U.S. Geological Survey using a total of 231 samples taken between 1964 and 1997, groundwater EC ranges from 0.17 to 9.56 dS/m and averages 2.36 dS/m throughout the entire alluvium. Generally, samples which are taken near rivers or canals tend to show lower EC values. This may be due to dilution from river leakage or canal leakage. In some samples, EC is especially high, which may indicate salt pick-up from the aquifer or heavy evapo-concentration on the ground. Figure 5.3 compares the chemical composition of groundwater in five counties. As compared to other counties, Crowley County's groundwater is high in Na^+ and low in Ca^{++} and Mg^{++}. In Bent County and Prowers County the groundwater composition is high in SO_4^{2-}.

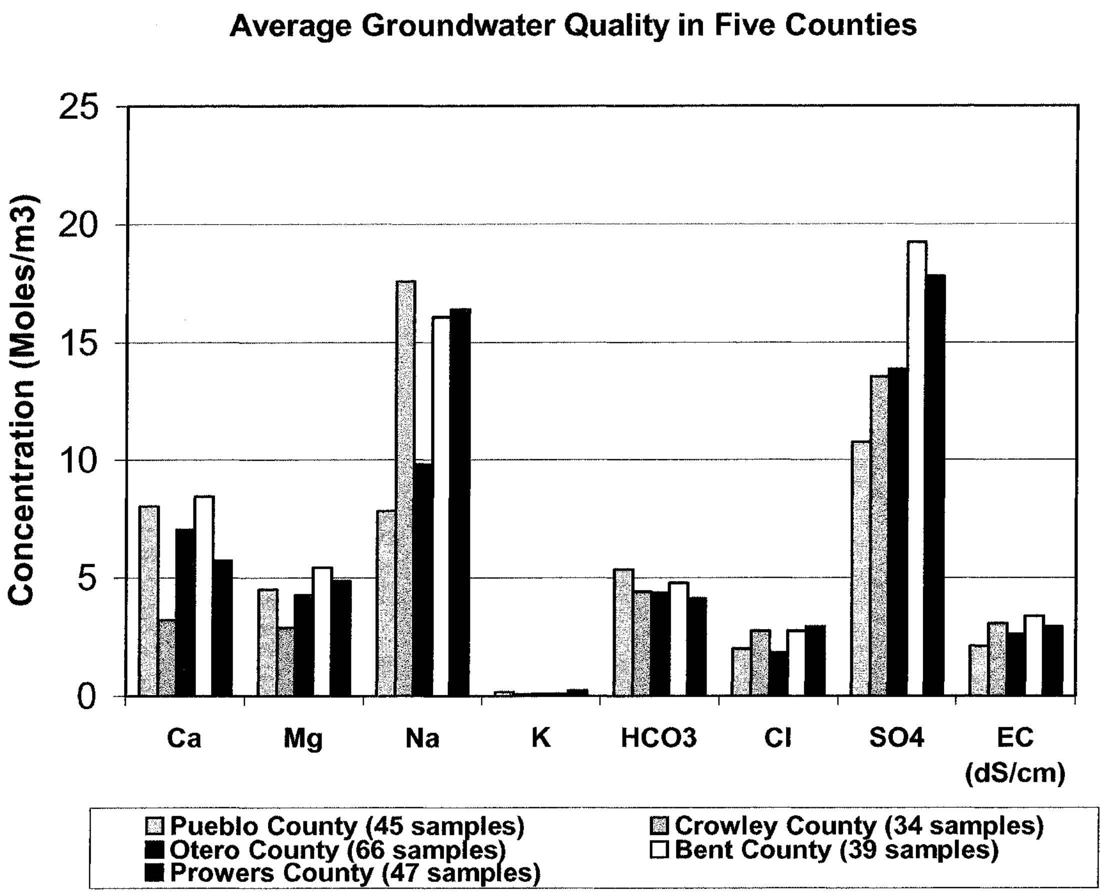


Figure 5.4: Groundwater Composition in Five Counties in the Arkansas Valley

In the process of irrigation, a certain portion of the irrigation water will turn into tail water. Tail water rapidly returns to the river system through surface drainage ditches. Tail water substantially influences both the quantity and quality of surface water (Cain, 1985). Figure 5.5 uses Timpas Creek as an example to illustrate how tail water affects river water quantity and quality. Timpas Creek is a major tributary of the Arkansas River located in southern Otero County. Since this tributary is adjacent to four major irrigated areas (Rocky Ford, Catlin, Otero and Highline Canal irrigated areas), it receives significant amounts of tail water from these areas during the irrigation season. As shown in figure 5.5, the stream flow rate is low and stable during the off-season when irrigation is not being practiced (November, December and January). On the other hand, the stream flow rate is high and fluctuating during the irrigation season (February through October). When no irrigation is taking place, the stream flow is recharged only by groundwater base flow. Since base flow recharge is steady, analysis of 1990 to 1995 values shows that the stream flow rate during the period from November to January is maintained at about 15 cfs with an EC of about 3.3 dS/m. As the irrigation season begins in March, tail water is created whenever farmers apply water to their fields. The amount of tail water fluctuates from month to month because the crop's needs change over the season. As a result, stream flows fluctuate from March to October. Tail water contains much less salt than groundwater. Therefore, it dilutes stream flow and lowers stream EC. As shown in figure 5.5, stream flow EC and stream flow rate show a negative correlation. When stream flow is high, the corresponding EC is low and vice versa.

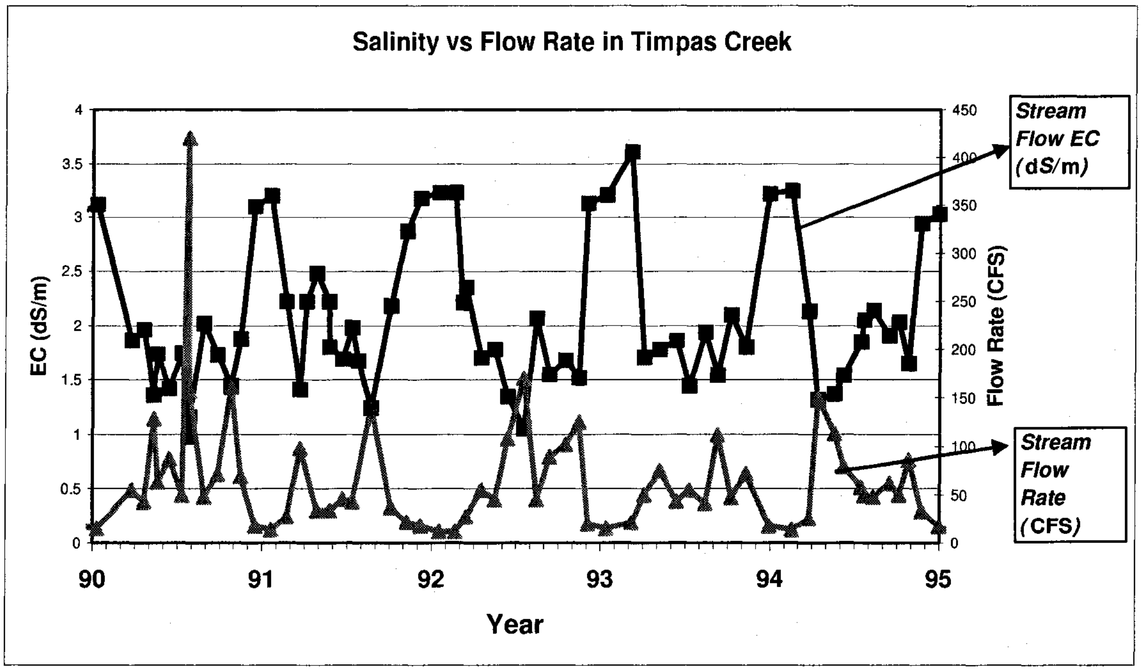


Figure 5.5: Salinity vs. Flow Rate in Timpas Creek

Due to the influence of return flows, the Arkansas River shows a significant increase in salinity from upstream to downstream. Based on water quality records published by the U.S. Geological Survey, EC in the upstream Arkansas River near Pueblo is 0.52 dS/m on average. EC gradually increases over each mile of the river. At the Kansas state line, EC reaches 3.55 dS/m. Figure 5.6 show average EC at seven major gaging stations using data collected from 1961 to 2002.

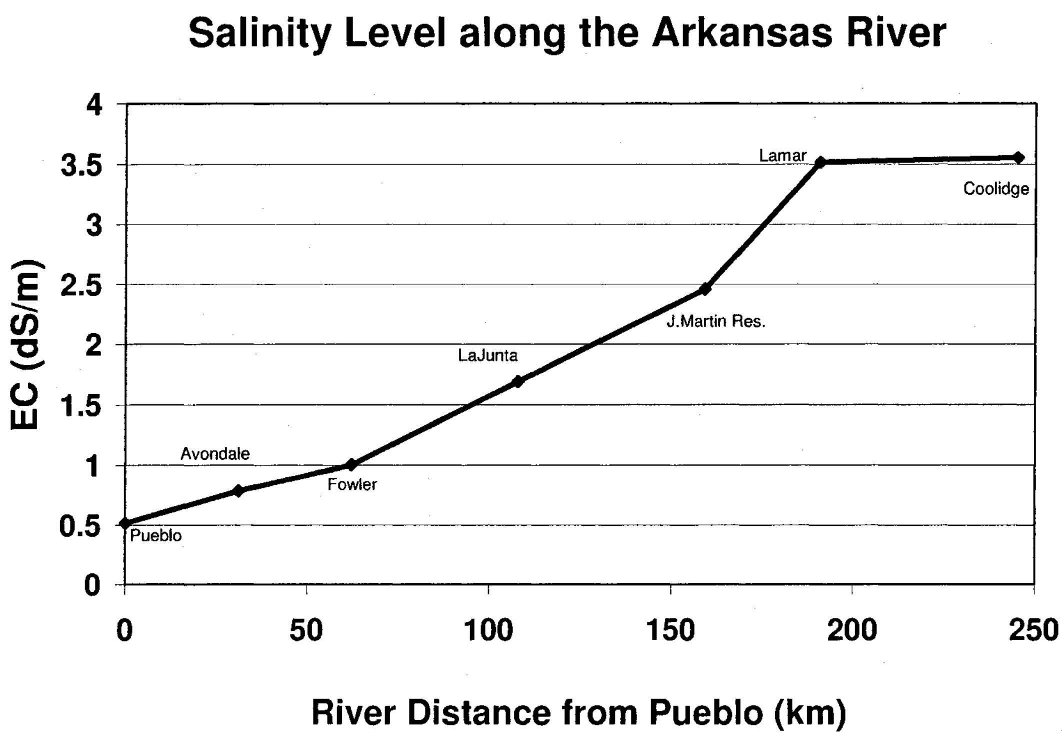


Figure 5.6: Increase in River Salinity from Pueblo to the Kansas State Line

5.2 Calibration Procedure

The salinity model is calibrated for the Arkansas River Basin. The calibration period is from January, 1986 to December, 1990, a total of 60 months. The calibration procedure consists of three major steps: database construction, response function generation and parameter adjustment.

5.2.1 Database Construction

Database construction entails preparing all input data and parameters required in the salinity simulation model. Figure 5.7 shows a graphical interface designed to facilitate this process. As shown in the diagram, the irrigation network in the Arkansas River is symbolized as components of river intakes, canal service areas, reservoirs, river reaches and canals. There are a total of 24 reaches, 21 canal areas, and seven reservoirs in the study area.

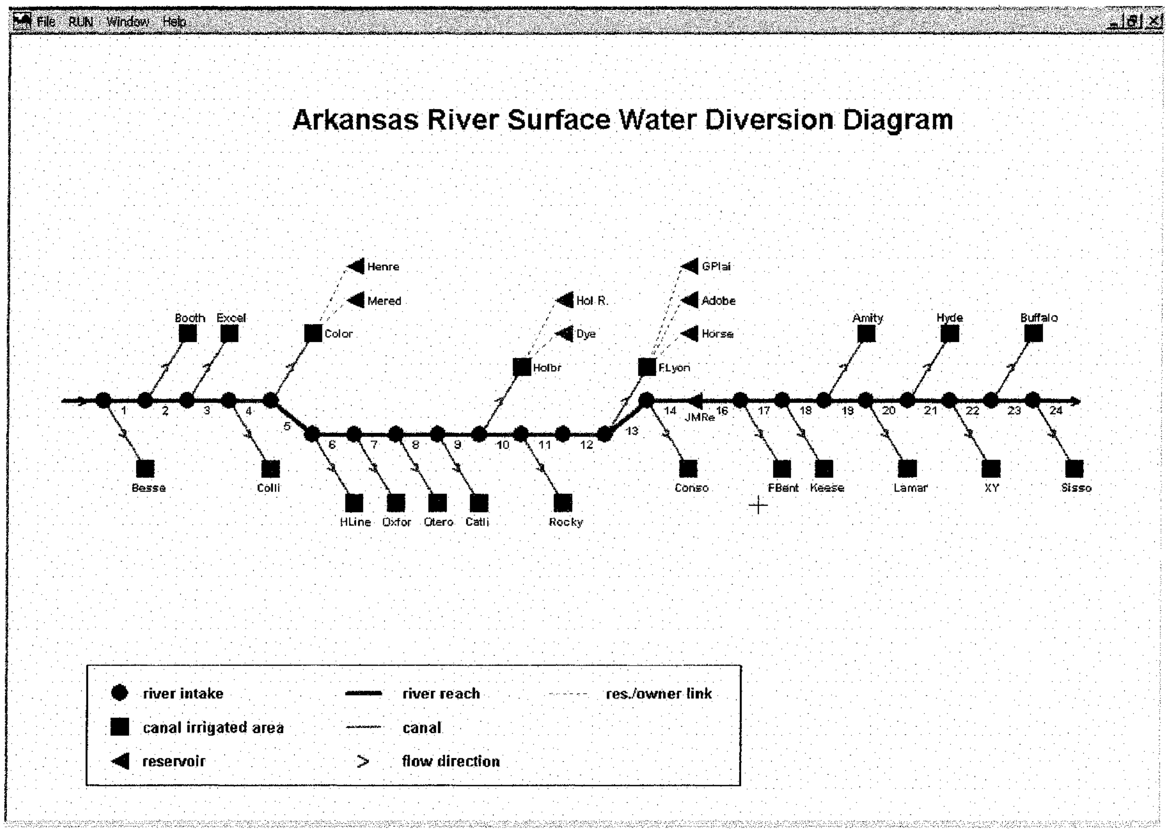


Figure 5.7: Irrigation Network for the Arkansas River

The input data for river intakes (the circles in figure 5.7) are monthly diversions for each canal. This data is based on canal diversion records provided by the Colorado State Engineer.

The input data for canal service areas (the squares in figure 5.7) include precipitation, groundwater pumping, crop consumptive use, surface water return flow parameters, groundwater return flow parameters and stream depletion parameters. The precipitation numbers are obtained by multiplying monthly rainfall in inches with the acreage of each canal service area. Groundwater pumping data includes quantity and quality of the pumping water. The quantity is based on historical monthly pumping records for each canal service area. The quality of the pumping water is based on historical groundwater composition. The locations of the pumping wells are shown in figure 5.9 which is based on Major et al. (1970). For each canal service area, it is assumed that the amount of pumping in each service area is uniformly distributed over the wells. This means that if there are n wells in a canal service area which pump a total of X units of water in a month, the amount of pumping in each well in this month is X/n units.

The quality of the pumping water is based on historical groundwater composition. Crop consumptive use (CU) is calculated using the Blaney-Criddle method (Viessman, et al., 1977) using a computer program developed by Garcia (2000). The required data for calculating crop consumptive use includes temperature, percent daylight, and crop types for each canal service area. Temperature and daylight data are obtained from the Colorado Agricultural Meteorological Network (COAGMET). Crop type data are obtained from the Colorado State Engineer. Table 5.3 shows crop types and the corresponding irrigated acreage for each canal service area.

Table 5.3: Acreage of Major Crop Types in Each Canal Area in 1998 (from Colorado State Engineer, 1999)

Canal Name	Winter Wheat (acre)	Corn (acre)	Grain Sorghum (acre)	Dry Beans (acre)	Alfalfa (acre)	Grass & Other Hay (acre)	Vegetables (acre)	Total Acreage (acre)
Bessemer	0	6574	160	179	8391	2479	261	18044
Booth Orchard	0	7	12	0	278	204	0	502
Excelsior	0	312	0	0	1007	923	9	2251
Collier	0	250	0	0	202	80	0	532
Colorado	0	2095	48	0	14977	6709	75	23904
Rocky Ford Highline	0	5435	0	0	12318	2993	0	20746
Oxford	0	1155	0	0	3528	302	0	4985
Otero	0	359	0	35	1179	1431	0	3004
Catlin	49	6609	50	268	7874	3074	0	17925
Fort Lyon	8459	15359	455	176	57770	9091	0	91309
Rocky Ford	0	1509	0	74	1183	2820	69	5655
Holbrook Canal	0	1492	0	4	8927	3269	0	13692
Las Animas Consolidated	0	937	0	44	4368	1489	0	6838
Baldwin-Stubbs	0	215	0	0	996	181	0	1392
Fort Bent	601	1066	0	0	2494	402	0	4562
Keese	870	76	0	0	838	174	0	1958
Amity	5676	7246	0	0	23253	2904	0	39080
Lamar/Manvel	697	2584	0	0	2576	2326	0	8184
Hyde	274	147	0	0	797	70	0	1287
XY Graham	1018	1951	0	0	1060	510	0	4539
Buffalo	681	663	0	0	3018	732	0	5095
Sisson	0	199	0	0	0	0	0	199
Pumpers at Stateline	1026	1178	0	0	1967	831	0	6003
Total	19351	57418	726	781	160002	42993	415	281666

The calculation of crop consumptive use needs to be adjusted to groundwater table depth in the Arkansas River Valley. This is because when groundwater table is shallow, a portion of the plant's roots will be immersed. The portion of root that is immersed loses the ability to uptake soil water. This results in less crop ET. To appropriately adjust crop ET, water table depth information in the irrigation area is required. By using water table depth data from approximately 960 wells in the Arkansas River Valley (Major et al., 1970), water table depth surfaces are constructed. Figure 5.8

shows the generated surface for Otero County. In the figure, water table depth is measured in meters. Blue cells represent the river. Blank cells represent the aquifer boundary. With the surfaces and crop irrigation acreages shown in table 5.3, the acreage over which plant roots are immersed in groundwater for each crop type can be estimated. Plant water uptake is assumed to be 40%, 30%, 20% and 10% from the upper to lower quarters of the root zone. By calculating the length of the portion of the root immersed in the groundwater using the water table depth surface, the amount of the reduced crop ET can be calculated. Then, the adjusted crop consumptive use for each canal service area in each month can be obtained. Table 5.4 shows the percentage of reduction of crop EC for four major crop types in each canal service area due to shallow groundwater table problems in the Arkansas River Basin.

In addition to shallow groundwater table, salinity stress in soil is another factor that can result in reduced crop consumptive use. This is because salinity stress causes high osmotic pressure in soil water making it difficult for crop roots to uptake water. According to the FAO (2004), crop ET under salinity stress in soil can be calculated by the following two equations:

$$ET_{adj} = ET \cdot K_s \tag{5.1}$$

$$K_s = 1 - \frac{b}{K_y 100}(EC_e - EC_{e\ threshold}) \tag{5.2}$$

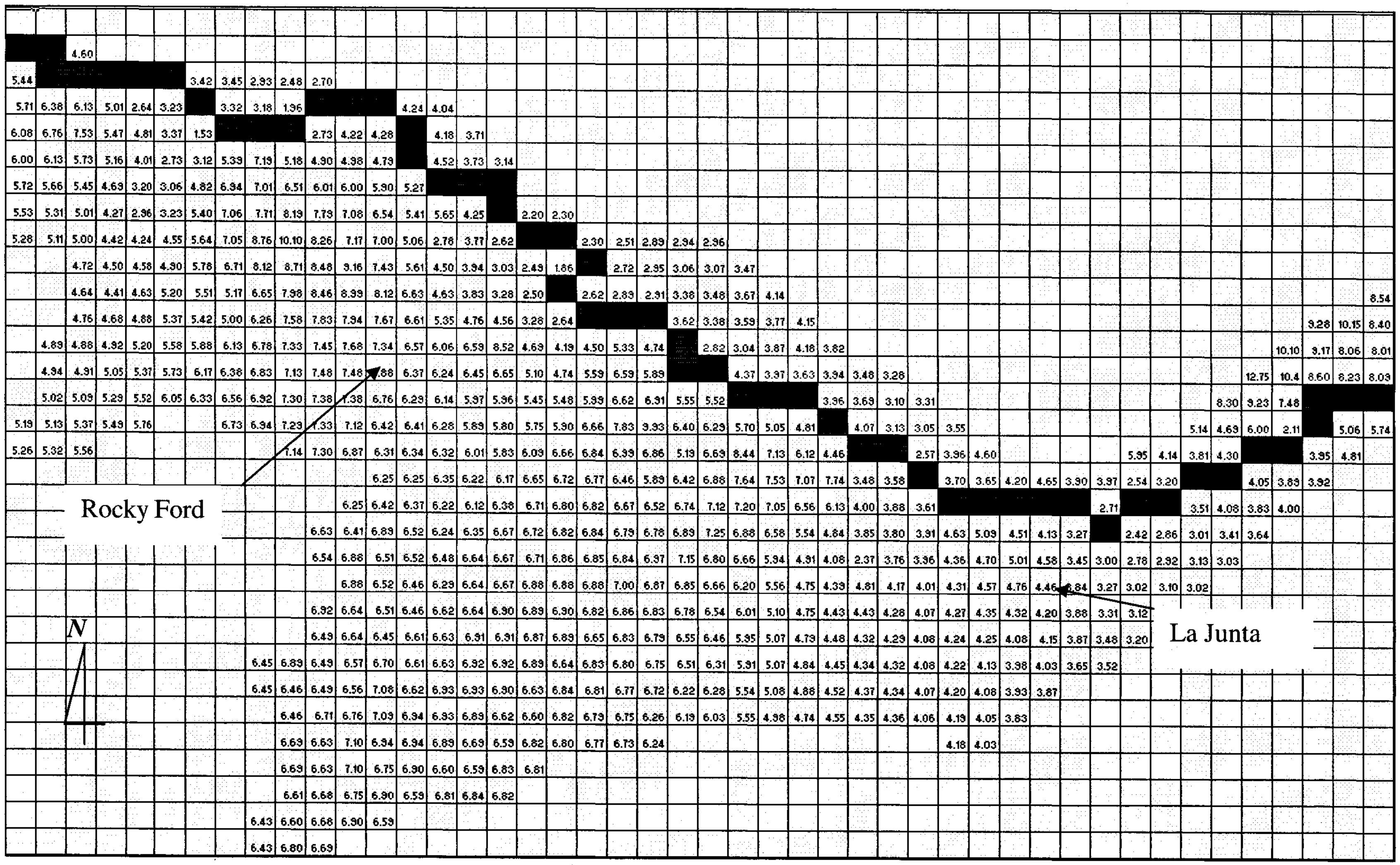


Figure 5.8 Generated Surface of Groundwater Table Depth in Otero County (800m x 800m cell size)

Table 5.4 Percentage of Crop EC Reduction due to Shallow Groundwater Table

Canal Service Area	Winter Wheat or Corn (Root Depth=4ft)	Grass & other hay (Root Depth=5ft)	Alfalfa (Root Depth=6ft)
Bessemer	0.00%	0.00%	0.00%
Booth	0.00%	0.00%	0.00%
Excelsior	0.00%	0.43%	0.87%
Collier	0.00%	2.50%	7.50%
Colorado	0.00%	0.10%	0.21%
Highline	0.00%	0.00%	0.00%
Oxford	0.00%	0.00%	0.00%
Otero	0.00%	0.00%	0.00%
Catlin	0.00%	0.00%	0.15%
Holbrook	0.00%	0.00%	0.00%
RockyFord	0.00%	0.00%	0.00%
FtLyon	0.00%	0.05%	0.11%
Consolidated	0.00%	0.18%	0.88%
FtBent	0.00%	0.67%	2.00%
Keese	0.00%	0.00%	0.71%
Amity	0.00%	0.00%	0.00%
Lamar_Manvel	0.00%	0.32%	0.97%
Hyde	0.00%	1.00%	1.00%
XY	0.00%	0.00%	0.00%
Buffalo	0.00%	0.00%	0.24%
Sisson	0.00%	1.67%	1.67%

In equation (5.1), ET_{adj} represents the adjusted crop evapo-transpiration (mm) of a certain crop type due to salinity stress in the soil. ET is the amount of evapo-transpiration when the salinity of the soil is not taken into consideration. K_s is the water stress coefficient for a certain crop type. $0 \leq K_s \leq 1$. $K_s=1$ means no salinity stress in the soil. As salinity stress increases, K_s decreases to 0. As shown in equation (5.2), K_s can be expressed as a function of b, EC_e, $EC_{e\ threshold}$ and K_y. b is the percentage reduction in crop yield per unit of increased salinity ($\%$ / dSm^{-1}) as measured by EC_e beyond a threshold value, $EC_{e\ threshold}$. EC_e is the average electrical conductivity of the saturation extract for the root zone (dS/m). $EC_{e\ threshold}$ is the electrical conductivity of the saturation extract at the threshold of EC_e at which the crop yield starts to decline. K_y is a factor that describes the reduction in relative yield according to the reduction in *ET* caused by soil water shortage. FAO has provided guidelines for the values of each parameter for each type of crop (Appendix D).

The EC_e in equation (5.2) can be estimated as the soil water salinity multiplied by 0.5 (Tanji, 1990). In the model developed in this study, soil water salinity is simulated in the root zone for each canal service area. The simulated soil water salinity represents an average value in the canal service area. Due to the fact that soil water salinity is spatially varied and therefore some fields have lower values and some have higher values than the average value, the soil water salinity should be described as a distribution instead of a deterministic value when calculating EC_e in a canal service area. Let $\overline{\mu}$ represents the average soil water salinity simulated from the model, the distribution of soil water salinity can be assumed to be as follows.

$$f(U) = \begin{cases} 0.5, & for \quad \overline{\mu} \geq U \\ 0.25, & for \quad 2\overline{\mu} \geq U > \overline{\mu} \\ 0.15, & for \quad 3\overline{\mu} \geq U > 2\overline{\mu} \\ 0.1, & for \quad U > 3\overline{\mu} \end{cases} \tag{5.3}$$

$$E[f(U)] = \overline{\mu} \tag{5.4}$$

where U is a random variable that represent soil water salinity in a canal service area; $f(U)$ represents the probability distribution of U; $E[f(U)]$ is the expected value of $f(U)$ which equals to the average soil water salinity $\overline{\mu}$.

By using the salinity simulation model to simulate $\overline{\mu}$ and applying equation (5.3) and (5.4), the distribution of EC_e in each canal service area in each month can be obtained. Then, K_s for each crop type can be calculated based equation (5.2). With information on crop acreages shown in table 5.3, the adjusted crop consumptive use for each canal service area in each month can be obtained. Table 5.5 shows the adjusted crop consumptive use in each canal service area from January, 1986 to December, 1990 and the percentage of crop consumptive use reduction due to high soil salinity in the Arkansas River Basin (detail data is shown in Appendix E).

Table 5.5 Percentage of Crop Consumptive Use Reduction due to High Soil Salinity

Canal Service Area	Adjusted Crop Consumptive Use (acre-ft)	Non-adjusted Crop Consumptive Use (acre-ft)	Percentage of Reduction
Bessemer	32243.3	32305.3	0.19%
Booth	2985.0	2993.2	0.27%
Excelsior	10460.9	10498.7	0.36%
Collier	801.6	804.5	0.36%
Colorado	109008.2	109441.6	0.40%
Highline	43533.6	43689.3	0.36%
Oxford	5526.9	5555.4	0.51%
Otero	18904.9	19067.7	0.85%
Catlin	39855.4	40271.4	1.03%
Holbrook	38063.9	38304.2	0.63%
Rocky Ford	39345.8	39969.2	1.56%
Ft Lyon	197770.9	200059.9	1.14%
Consolidated	19220.2	19732.3	2.60%
Ft Bent	6550.6	6900.4	5.07%
Keese	2059.7	2171.5	5.15%
Amity	58621.5	60894.0	3.73%
Lamar_Manvel	14102.6	15305.8	7.86%
Hyde	1306.7	1369.1	4.56%
XY	7317.8	7800.7	6.19%
Buffalo	11441.3	12346.6	7.33%
Sisson	33.2	33.4	0.60%

The input data for river reaches (the thick lines in figure 5.7) include tributary inflow, phreatophyte consumption, river channel loss parameters, and groundwater baseflow. For the data of tributary inflow, monthly inflow volume and its corresponding chemical composition (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, SO_4^{2-}, HCO_3^{-} and CO_3^{2-}) are required. The data can be acquired from the website of the U.S. Geological Survey (http://www.usgs.gov). There are some missing data for the simulation period. These missing data must be filled in, or the model cannot run. The methodologies for generating missing data are as follows: (a) For the missing data of monthly tributary inflow volume, seasonal average volume from other known data are used. For example, if the tributary volume in reach 3 in May/1988 is unknown, its substitute data will be the average of tributary volume in reach 3 in May/1986, May/1987, May/1989 and May/1990. (b) For the missing data of chemical components, if the concurrent EC data is available, these missing data will be generated based on the EC level. For example, for a specific month, Ca^{2+}, Mg^{2+}, Na^{+}, K^{+} and CO_3^{2-} are available but SO_4^{2-} and HCO_3^{-} are missing. If the EC in that month is 2.11 dS/m. Then, the substitute values of the missing SO_4^{2-} and HCO_3^{-} will be generated to make the chemical composition in that specific month attain an EC level of 2.11 dS/m. In doing this kind of data generation, computer codes were designed to do the iterative computation. The computation will make the error between the simulated and the observed EC be under 10^{-3} dS/m so that the generated data can be validated. (c) For a situation like (b) where no concurrent EC data is available, the substitute values for the missing data will be the seasonal average data taken from the known data.

Phreatophyte consumption represents river water consumed by phreatophyte trees in a river reach in a month. Data sets are based on calculations by a computer program used by the Kansas State Government. Input data of river channel loss parameters are monthly river channel loss ratio and response functions for channel loss. The monthly channel loss ratio is set up based on the research of Livingston (1978). Due to the lack of a relevant reference, the response functions for channel loss are first set up in a reasonable range. When the salinity model is calibrated, the values are adjusted. Input data for groundwater base flow will be discussed later.

Input data for canals (the thin lines in figure 5.7) include canal transit loss and transit loss return flow parameters for each canal. The amount of transit loss is computed as a ratio multiplied by monthly diversions. The ratio is simulated based on research of transit losses in the Arkansas River by Livingston (1978). Input data for transit loss return flow parameters will be discussed in the next section.

Input data for the seven reservoirs (the triangles in figure 5.7) include reservoir evaporation and reservoir leakage. For John Martin reservoir, input data also include monthly release data from the reservoir. The amount of reservoir evaporation is calculated by multiplying evaporation depth with the area of each reservoir. Based on a survey of mean annual evaporation conducted by the U.S. Department of Agriculture (Viessman, et al., 1977), the mean annual free water surface evaporation depth in Colorado is approximately 65 inches. Therefore, the monthly averaged free water evaporation depth is set to be 5.42 inch/month. Area of the reservoirs can be computed using a Geographic Information System. The amount of reservoir leakage is calculated by multiplying leakage depth with the area of each reservoir. The leakage depth is calculated based on Horton's equation (Chow et al., 1988). The equilibrium infiltration parameter in Horton's equation is first set to range from 0.01 inch / hour to 0.1 inch / hour. When the salinity model is calibrated, this parameter is adjusted to obtain an optimal value. Response functions for reservoir leakage are set in a range first. When the salinity model is calibrated, their values are adjusted.

5.2.2 Response Function Generation

In chapters 3 and 4, methodologies are developed to generate response functions for calculating return flows and stream depletions. In this section, the methodologies are applied to the Arkansas River Basin in Colorado. As described in Section 5.1.2, groundwater is confined to the alluvium, and the alluvium thickness is assumed to range from 0 to 300 ft (table 5.2), which is much less than the lateral dimension of the alluvial aquifer. Therefore, the two-dimensional flow assumption in Chapter 4 is satisfied. As shown in Chapter 4, calculating response functions for two-dimensional groundwater flows consists of three steps: constructing a groundwater table elevation, setting up stress cells, and assigning parameters. Some examples of the calculated response functions are shown in figures 5.29 and 5.30.

The groundwater table elevation is constructed using groundwater table data from 965 wells (Major et al., 1970). These wells penetrate into the alluvium. Figure 5.9 shows the location of these wells in the Arkansas River Basin. In addition to these wells, ten wells located in the surfacial aquifer in southern Otero County (Gates, et al. 2002) are also used. The Arkansas River Basin is discretized into 800m by 800m cells. Then, the water table elevation in each cell is interpolated based on the well data. A GIS program, ArcView3.2, is used for the interpolation. In ArcView3.2 the "spline" method is utilized. This method can minimize the curvature of the interpolated surface.

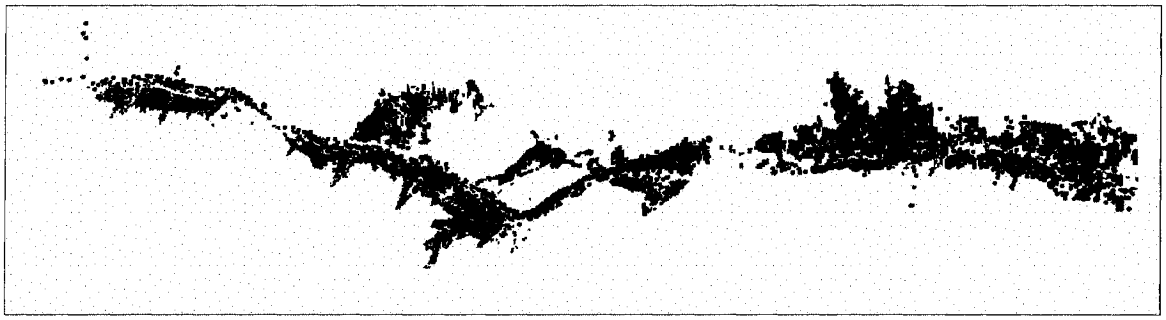


Figure 5.9: Locations of Pumping Wells (Dots) in Irrigated Fields in the Arkansas River Basin

After the groundwater table elevation in each cell is obtained, stress cells must be assigned. There are three kinds of stresses: irrigation deep percolation, canal leakage, and groundwater pumping. Figure 5.10 uses the upstream portion of the Arkansas River Basin to illustrate how stress cells are assigned. As shown in the figure, green cells are irrigated fields so they represent the stresses due to irrigation deep percolation. Light blue cells are canals so they represent stresses due to canal leakage. Red cells are pumping wells, representing stresses due to pumping. In figure 5.10 four canals and their service areas are displayed. These canals are: Bessemer, Booth Orchard, Excelsior and Collier. It should be noted that the cells representing irrigated fields can be overlaid with the cells representing a canal or the cells representing pumping wells.

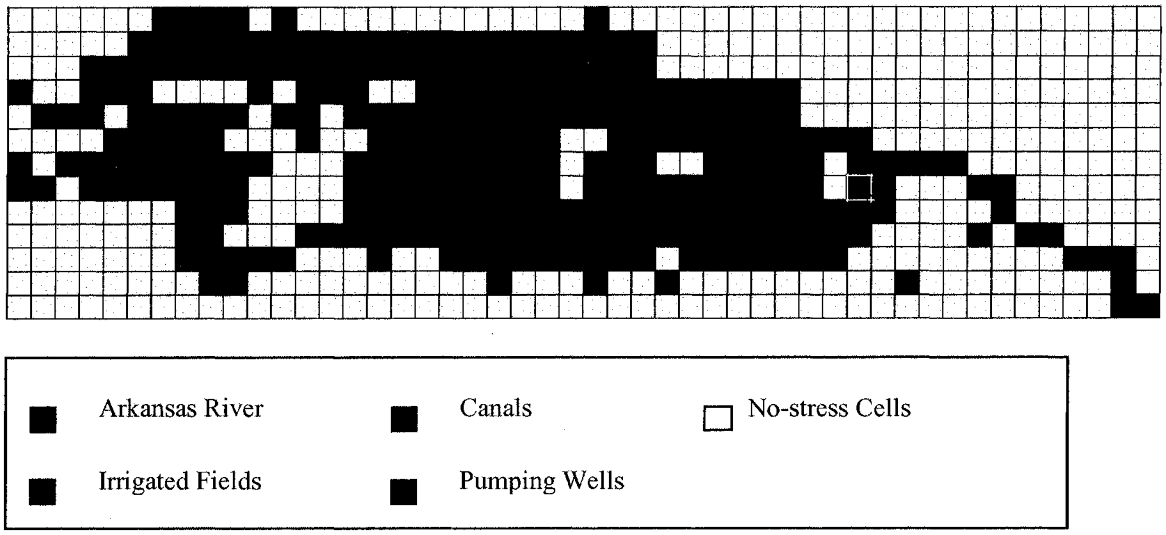


Figure 5.10: Assigning Stress Cells on the Arkansas River Basin

After all stress cells are set up for the 21 canal service areas, parameters for each aquifer cell must be set up. The parameters include hydraulic conductivity, porosity, and drainage portion. As mentioned in Section 5.1.2, Wilson (1965) surveyed the aquifer transmissivity in 56 locations. By dividing the transmissivity data with aquifer depth, hydraulic conductivity can be calculated. The result show that hydraulic conductivity ranges from 21.02 m/day to 623.06 m/day. Since data from 56 locations are not sufficient for estimating the entire basin, a simple yet reasonable strategy is applied in this study. For river cells and cells adjacent to river cells, hydraulic conductivity is assigned as 500m/day. A lower value, 380 m/day, is assigned to the rest of the cells in the basin. The value 380 m/day is approximately the average of the surveyed data. River cells are assigned to 500 m/day because groundwater usually flows faster in the vicinity of the river.

Porosity for alluvium material which contains a mixture of gravel, sand, silt, and clay could range from 0.25 to 0.7 (Freeze and Cherry, 1979). In this study, the porosity value is set to be 0.25 for each aquifer cell. As described in Chapter 4, the drainage portion is a parameter designed for irrigated fields that have some subsurface drainage device. This parameter must range from 0 to 1 for each cell. Zero means no drainage device in the cell. One means that all water leaching down through the root zone soil in this cell is drained by the drainage device. Based on field surveys of the Arkansas River Valley, it has been determined that some subsurface drains have been installed and that deep drainage ditches exist. However the drainage is not sufficient since parts of the valley suffer from waterlogging problems (Gates, et al., 2002). Therefore, drainage portions are assigned as 0.25 to each cell in this study.

After all parameters are set up, response functions for irrigation percolation, canal leakage, and pumping depletion in each canal area are ready to be generated. A C++ program was designed for the generation of the response functions (Appendix F). The general form of the response functions can be described as $g(s,t)$:

$$\sum_j \sum_s g(s,t) = 1 \tag{5.5}$$

where s and t represent the month and the river reach in which the return flow or stream depletion occurs.

By using a similar methodology, response functions for irrigation tail water can also be generated. The differences are that land surface elevation is used instead of groundwater table elevation to generate response functions and the drainage portion is set as 1. This allows tail water to return to the river system within a month since tail water does not flow through the underground porous media.

Theoretically, as all the response functions have been generated, irrigation return flows can be simulated for the entire basin. However, insufficient information about "initial conditions" can cause inaccuracies in the river salinity simulation. For example, assuming there is a response function which has an 120 month time lag (ten years). Then, if river salinity in 1986 is to be simulated, the program will require return flows data back to 1977 (ten years prior) since irrigation in 1977 would effect river quantity and quality in 1986. Due to this practical problem, an adjustment is made: the time lags of all response functions are limited to 12 months (1 year). For response functions with time lags greater than 12 months, the flow is considered as groundwater base flow. During calibration, groundwater base flow is estimated based on the original response functions and water use during the simulation period. Each river reach will be assigned a certain amount of groundwater base flow. With this adjustment, the influence of initial conditions can be minimized.

As described in section 3.3.1, groundwater return flow is classified as "short-term" and "long-term" flow. The chemical composition of the short-term flow is determined based on the result of a soil water quality simulation. The chemical composition of the long-term flow is based on observed ground water quality. In this study, groundwater return flow which stays underground for longer than 12 months will be classified as long-term flow. Groundwater return flow which remains underground for less than 12 months will be classified as short-term flow.

5.2.3 Parameter Adjustment and Results

After setting up all input data and parameters in the model, the salinity program is ready for execution. Each time the program is executed, it simulates monthly river quality and quantity at 24 locations along the Arkansas River as well as soil water composition in 21 irrigated areas from January 1986 to December 1990. The simulated values will be compared with observed values. Then, parameters will be adjusted to improve the simulation. The adjustment will continue until satisfactory results are obtained. Observed values are available from several gaging stations along the Arkansas River. For river quality data, stations at Avondale, John Martin Reservoir, Lamar and Coolidge have EC (dS/cm) data with a sampling frequency of approximately once a month by the U.S. Geological Survey. For river quantity data, stations at Avondale, Fowler, Lamar and Coolidge have monthly stream flow data (acre-ft) calculated by the Colorado State Engineer. Figure 5.11 shows the locations of these stations.

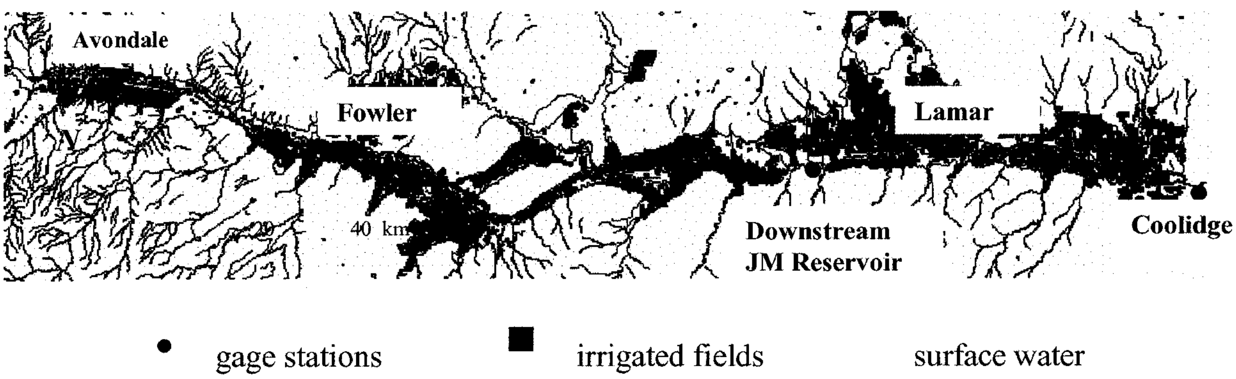


Figure 5.11: Location of Gaging Stations in the Arkansas River in Colorado

Four types of parameters are adjusted. They are tail water ratio, river channel loss response functions, leakage of John Martin Reservoir and response functions for leakage at the John Martin Reservoir. Table 5.4 shows the final values of these parameters.

Table 5.6: Final Values of the Adjusted Parameters

(a) Tail Water Ratio

Tail Water Ratio
0.32

(b) Response Functions for River Channel Loss

Time Lags	Downstream Reaches	
	+0	+1
+0	***0.60***	***0.00***
+1	***0.20***	***0.00***
+2	***0.20***	***0.00***

(c) Leakage in John Martin Reservoir

Leakage (inch/hour)
0.032

(d) Response Functions for John Martin Reservoir Leakage

Time Lags	Downstream Reaches												
	+0	+1	+2	+3	+4	+6	+7	+8	+9	+10	+11	+12	+13
+0	***0.00***	***0.02***	***.005***	***.005***	***.005***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***
+1	***0.00***	***.015***	***.005***	***.005***	***.005***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***	***0.01***
+2	***0.00***	***0.01***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***
+3	***0.00***	***.005***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***	***0.00***

In addition to the parameters shown in table 5.6, the amount of dissolution of the un-weathered calcite, gypsum and magnesite in the soil is also determined in the calibration process. There are significant amounts of un-weathered minerals in semi-arid areas like the Arkansas River Basin (Tanji, 1990). The un-weathered minerals can be dissolved as the soil water comes in contact with them. However, the dissolution rate is slow. Some factors that can affect the dissolution include: the salinity of the soil water, the leaching fraction in the irrigation fields, partial pressure of CO_2 in the soil and the geologic formation of the aquifer (Tanji, 1990). These factors are spatially-varied in an irrigated river basin. Butters and Cooper (2005) have analyzed the amount of minerals in soil samples taken from the Arkansas River Basin. The soil samples are repeatedly mixed with distilled water into saturated paste for chemical analyses. The results in a selected field show that the sum of Ca^{2+}, Mg^{2+} and Na^{+} is 3,650 mg/l in the first saturated paste, and the amount gradually decreases to 1,180 mg/l in the fourteenth saturated paste. However, due to the methodology of this analysis in which distilled water is used and thoroughly mixed to create the saturated soil paste for the analyses, the results can be far from what is expected in the fields. This is because distilled water has a much higher capacity for dissolving minerals due to its low salinity. Distilled water usually has a salinity of less than 0.1 dS/m, while actual soil water can have a salinity greater than 5 dS/m. Furthermore, thorough mixing can significantly increase the contact between the minerals and the water, and thus enhance the dissolution of mineral.

In this study, the amount of dissolution of the un-weathered minerals is addressed by adjusting the mineral dissolution capacity of the soil water. The soil root zone is divided into four layers. Each unit of soil water is assumed to have the capacity (10^{-3} mole / liter) to dissolve calcite, gypsum and magnesite as it filtrates down from the soil surface to the bottom of the root zone. Let d_1, d_2 and d_3 represent the amount of calcite, gypsum and magnesite dissolved by each unit volume of the soil water (*liter*) in each of the four soil layers. By adjusting d_1, d_2 and d_3 and checking the resulting river salinity, the final values of d_1, d_2 and d_3 can be obtained. The results are shown in table 5.7.

Table 5.7: Final Values of the Dissolution of the Un-weathered Minerals in Soil

Calcite (10^{-3} mole/ liter)	Gypsum (10^{-3} mole/ liter)	Magnesite (10^{-3} mole/ liter)
0.9	***1.2***	***0.3***

With the adjustment of the parameters, the salinity model is then calibrated. Figures 5.12 to 5.15 show simulated river EC versus observed river EC at four river locations. Figures 5.16 to 5.19 show simulated stream flow quantity versus observed stream flow quantity at four river locations.

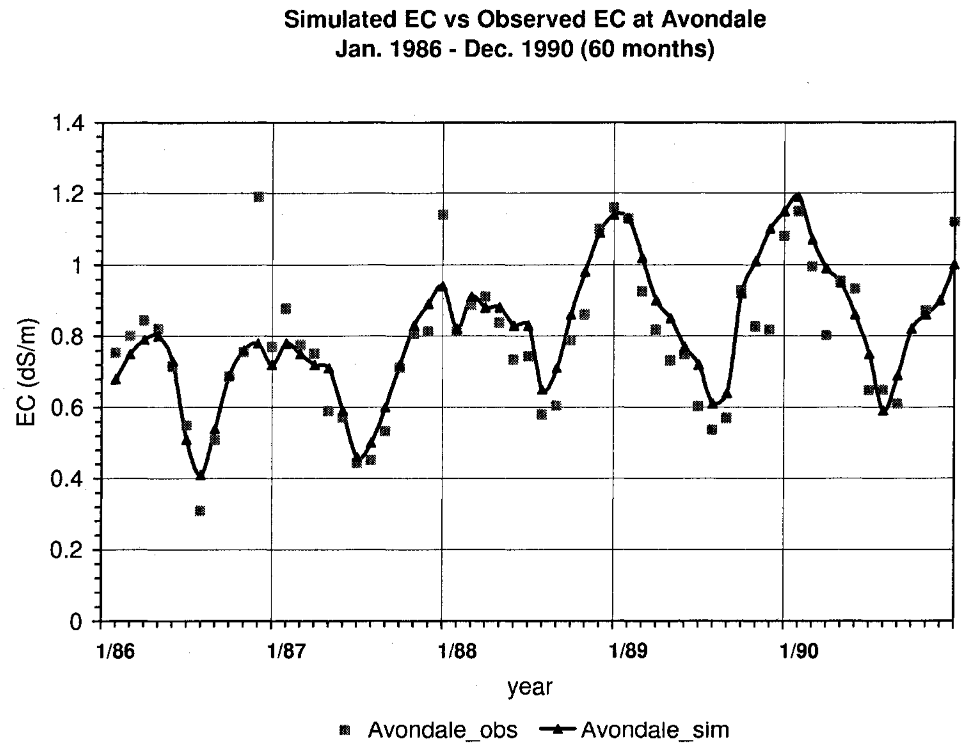


Figure 5.12: Simulated EC vs. Observed EC in the Arkansas River at Avondale (1986-1990, 60 Months)

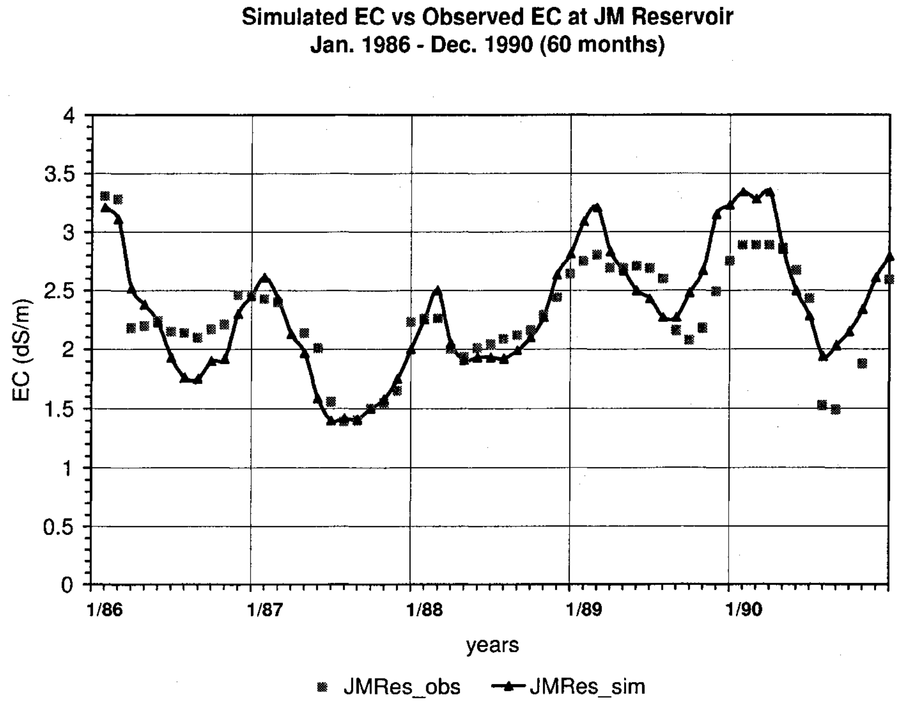


Figure 5.13: Simulated EC vs. Observed EC in the Arkansas River Downstream of John Martin Reservoir (1986-1990, 60 Months)

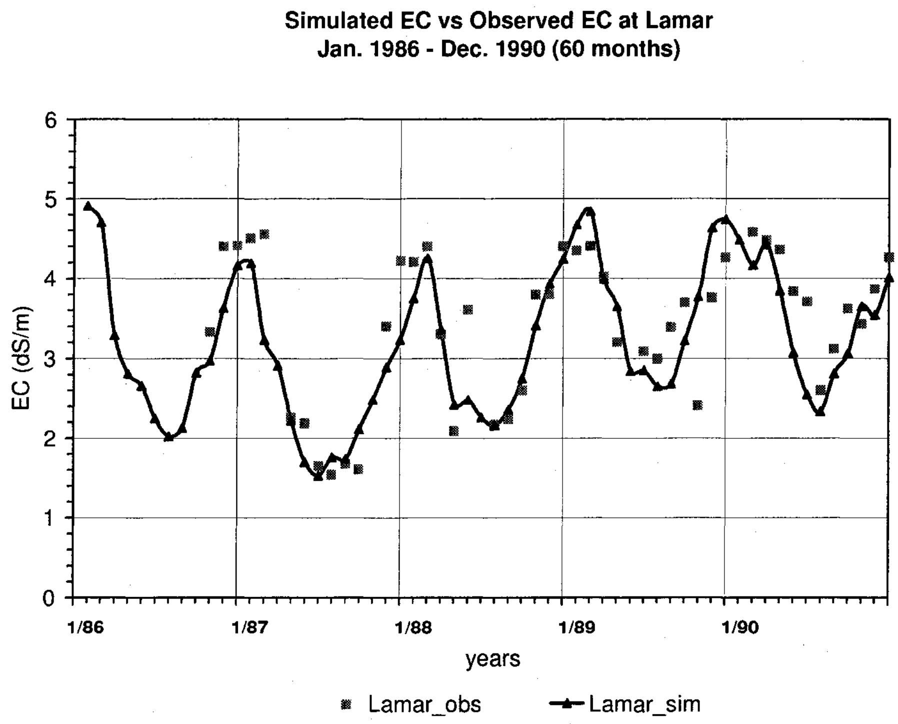


Figure 5.14: Simulated EC vs. Observed EC in the Arkansas River at Lamar (1986-1990, 60 Months)

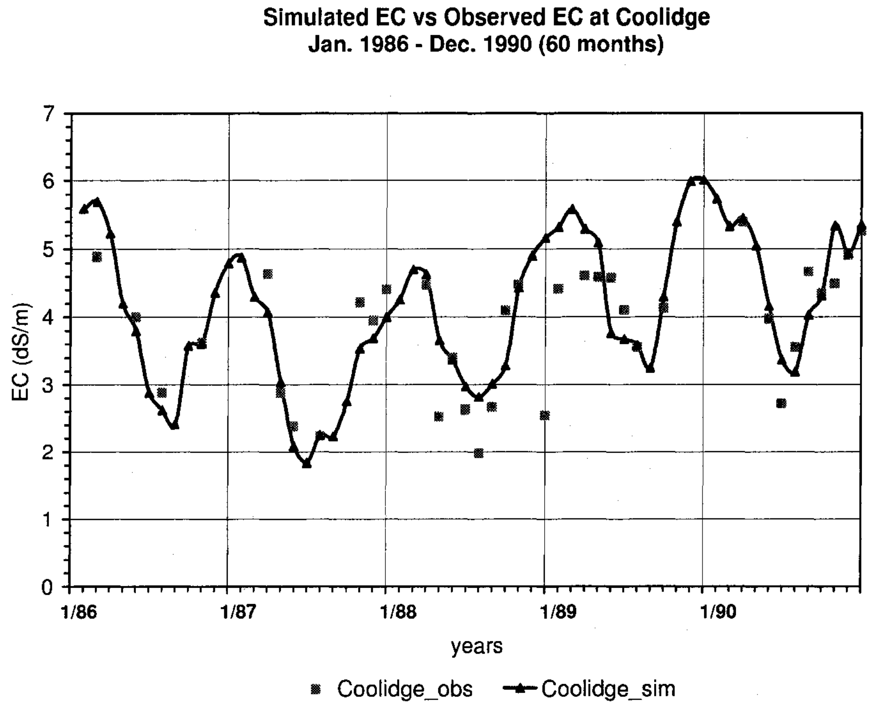


Figure 5.15: Simulated EC vs. Observed EC in the Arkansas River at Coolidge (1986-1990, 60 Months)

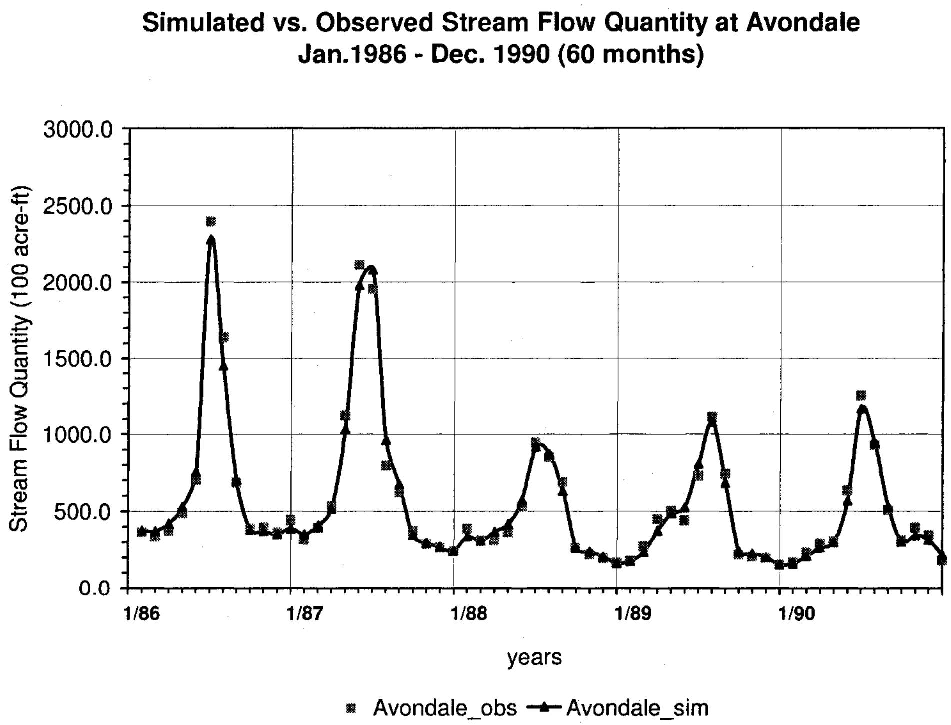


Figure 5.16: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Avondale (1986-1990, 60 Months)

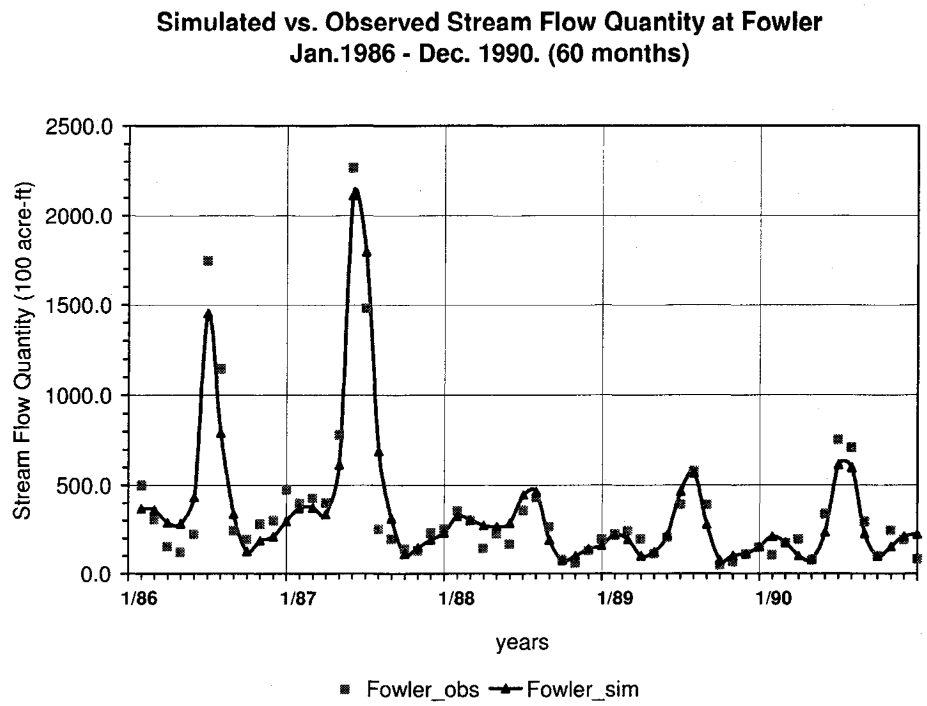


Figure 5.17: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Fowler (1986-1990, 60 Months)

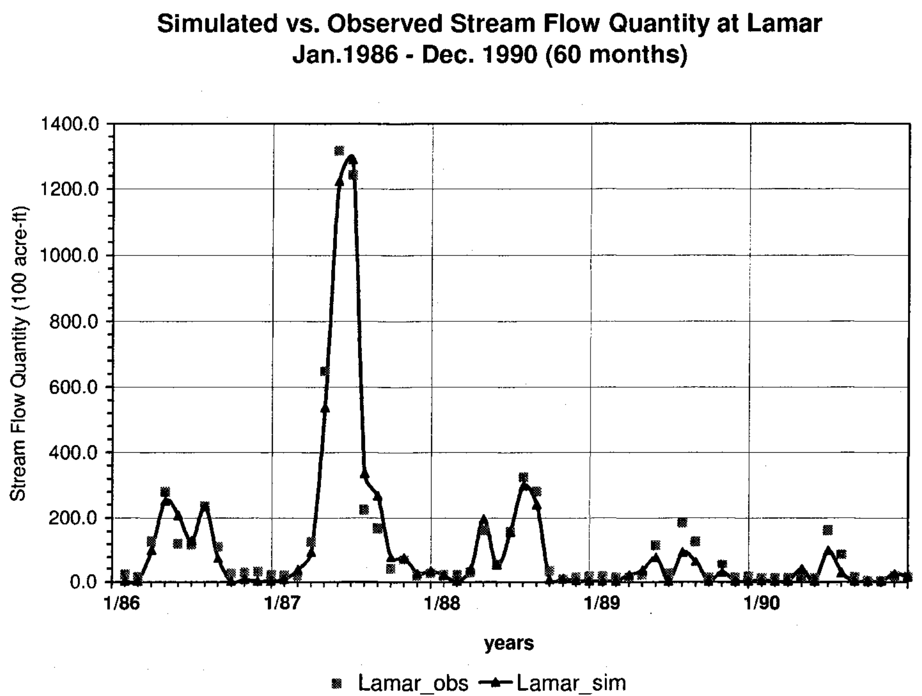


Figure 5.18: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Lamar (1986-1990, 60 Months)

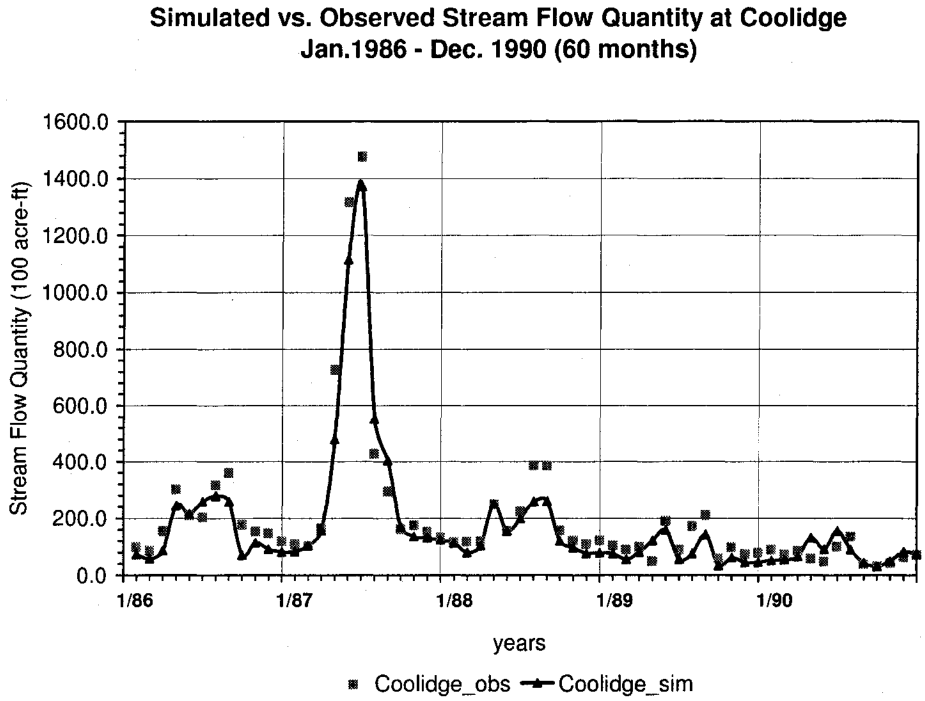


Figure 5.19: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Coolidge (1986-1990, 60 Months)

As shown in figures 5.12 to 5.19, the simulated EC and the simulated stream flow quantities match observed data well at different locations. The correlation coefficient (Hogg and Craig, 1989), r, is used to evaluate the accuracy of the simulation. r is defined as equation (5.6) as follows.

$$r = \frac{\frac{1}{n}\sum_{i=1}^{n}(X_i - \overline{X})(Y_i - \overline{Y})}{\sqrt{\frac{1}{n}\sum_{i=1}^{n}(X_i - \overline{X})^2}\sqrt{\frac{1}{n}\sum_{i=1}^{n}(Y_i - \overline{Y})^2}} \qquad (5.6)$$

where X_i represents the i^{th} observed data and Y_i represent i^{th} simulated data, $i=1,2,3,\dots,n$. n is the sample size of the data. $\overline{X}$ and $\overline{Y}$ are the means of X_i and Y_i respectively that

$\overline{X} = \frac{1}{n}\sum_{i=1}^{n} X_i$ and $Y = \frac{1}{n}\sum_{i=1}^{n} Y_i$. The better the simulation is, the higher the *r* value is and its upper limit is 1. The worse the simulation is, the more the *r* value is close to *0*.

By using correlation coefficient *r*, the accuracy of the simulation can be evaluated. The result is shown in table 5.8. In the table, all *r* values range between 0.75 and 0.99, indicating that the simulation is successful.

Table 5.8: Correlation Coefficients, *r*, for Simulation in Calibration Period

(a) *r* for Water Quality Simulation

	Avondale	JM Res	Lamar	Coolidge
r	***0.868***	***0.866***	***0.838***	***0.756***

(b) *r* for Water Quantity Simulation

	Avondale	Fowler	Lamar	Coolidge
r	***0.993***	***0.950***	***0.987***	***0.974***

5.3 Model Validation

After the salinity simulation model is calibrated for the period from 1986 to 1990, it is necessary to run the model in another time period. This is done to make sure that the parameters obtained in the calibration period (January 1986 to December 1990) still produce accurate result in other periods. This process is usually referred to as model validation. In this study, a 132-month period, from January 1991 to December 2001, is used for the validation. The hydrological record for this period is processed to create the input data required for the model. Then, the program is executed. The results are compared with observed values and are shown in figures 5.20 to 5.28. During the validation period, observed values of EC at Fowler are available while they are not available during the calibration period (January 1986 to December 1990). Therefore, there is one more set of data that can be used to validate the simulation model.

Figures 5.20 to 5.24 show simulated river EC versus observed river EC at five river locations. Figures 5.25 to 5.28 show simulated stream flow quantity versus observed stream flow quantity at four river locations.

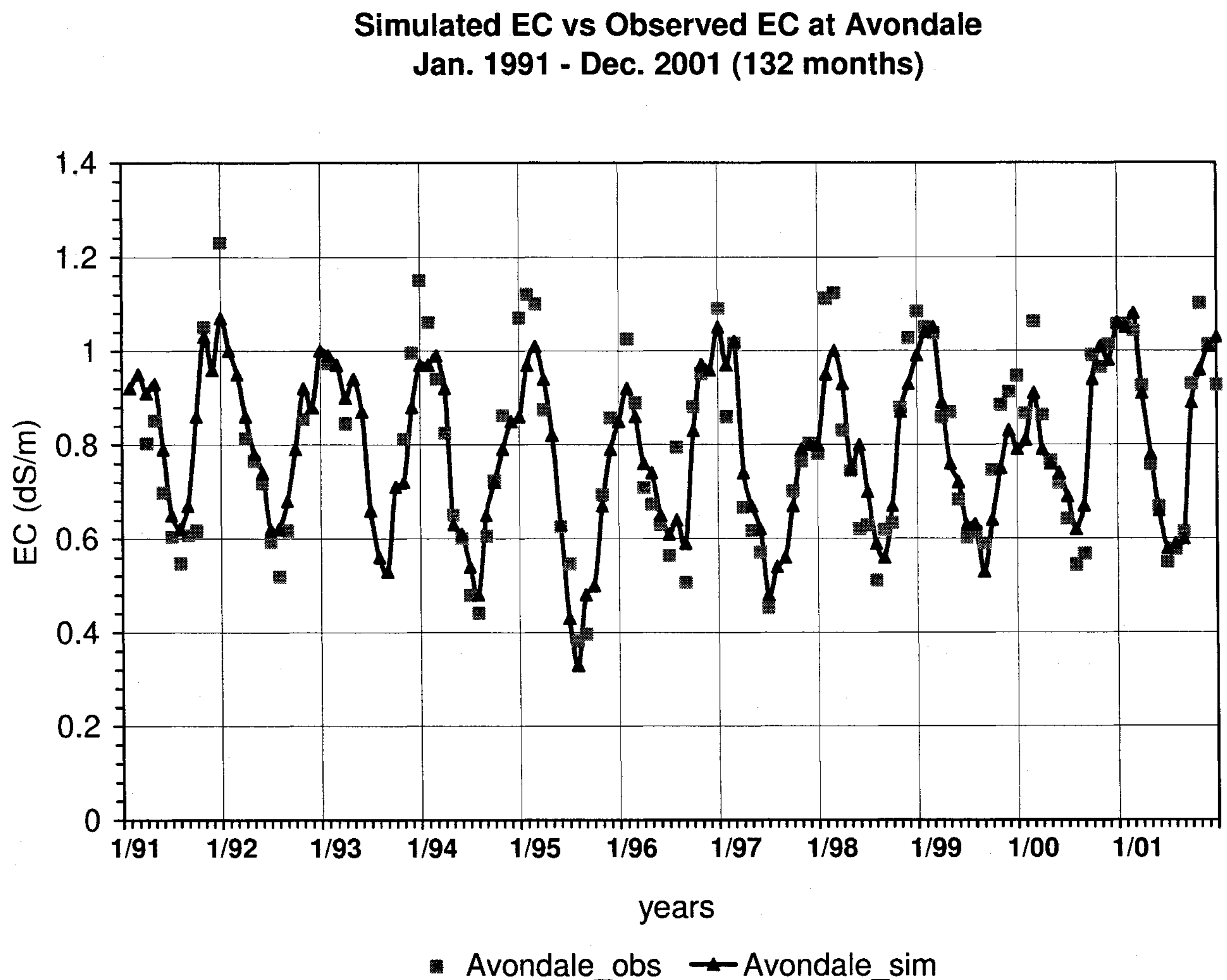


Figure 5.20: Simulated EC vs. Observed EC in the Arkansas River at Avondale (1991-2001, 132 Months)

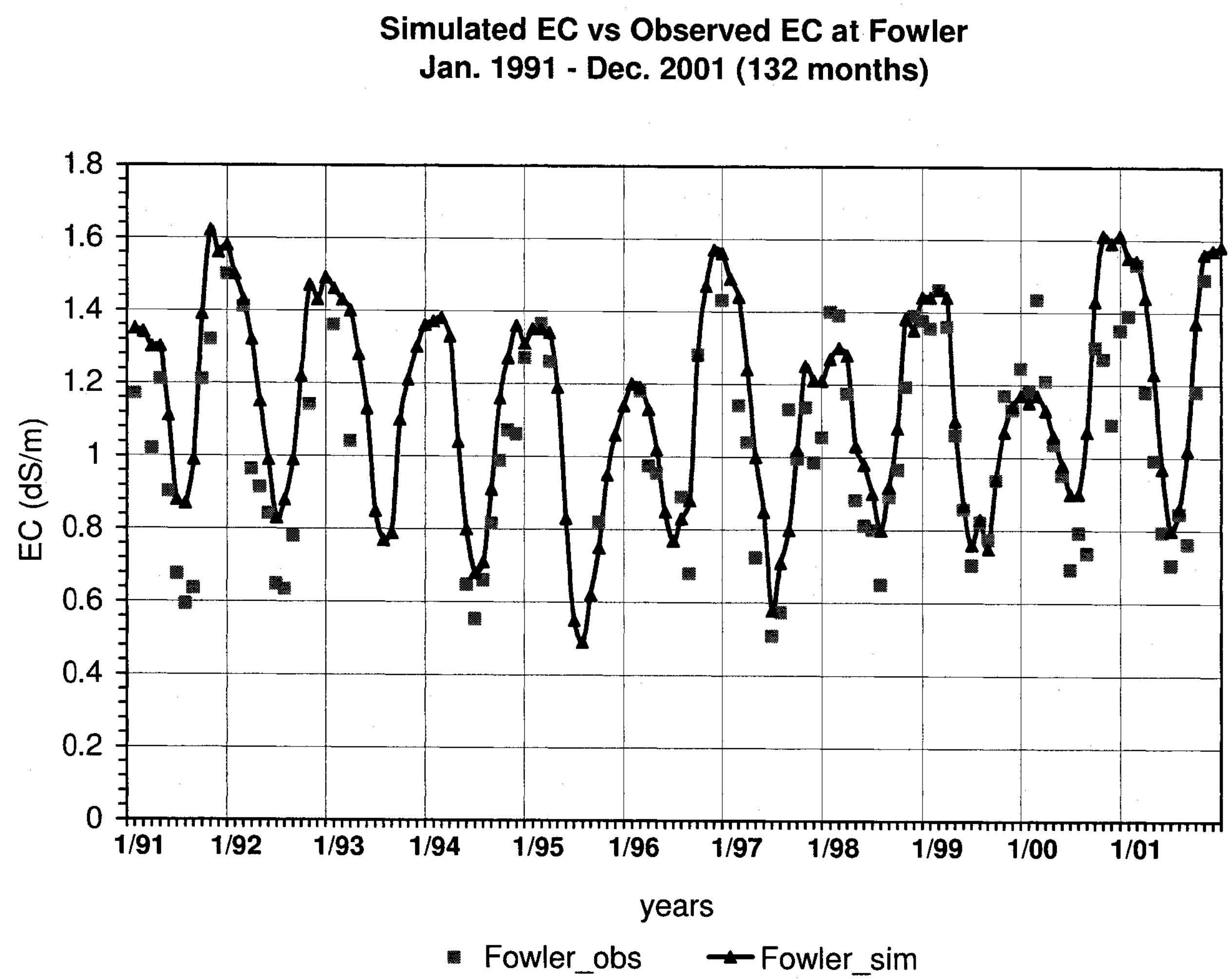


Figure 5.21: Simulated EC vs. Observed EC in the Arkansas River at Fowler (1991-2001, 132 Months)

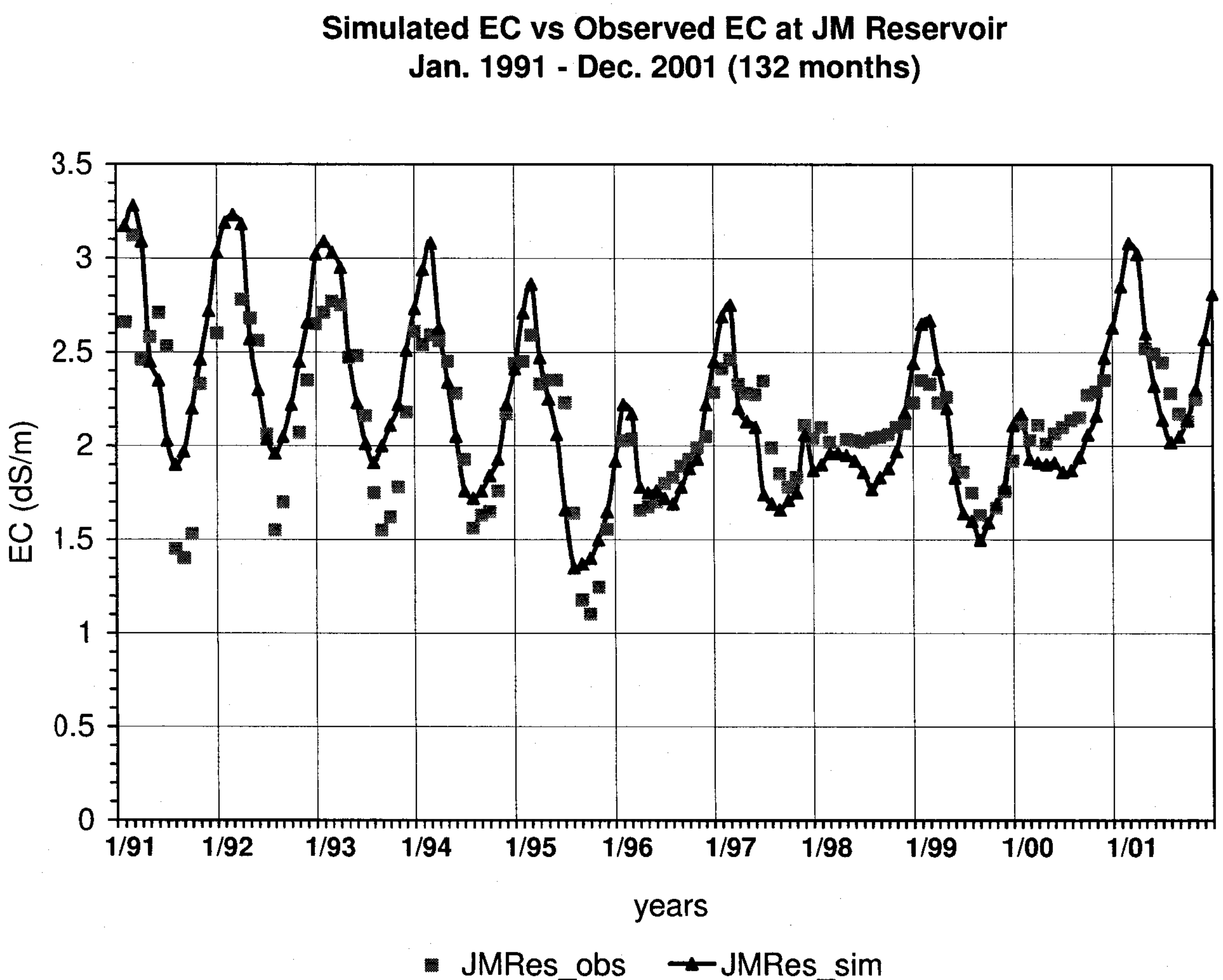


Figure 5.22: Simulated EC vs. Observed EC in the Arkansas River Downstream of John Martin Reservoir (1991-2001, 132 Months)

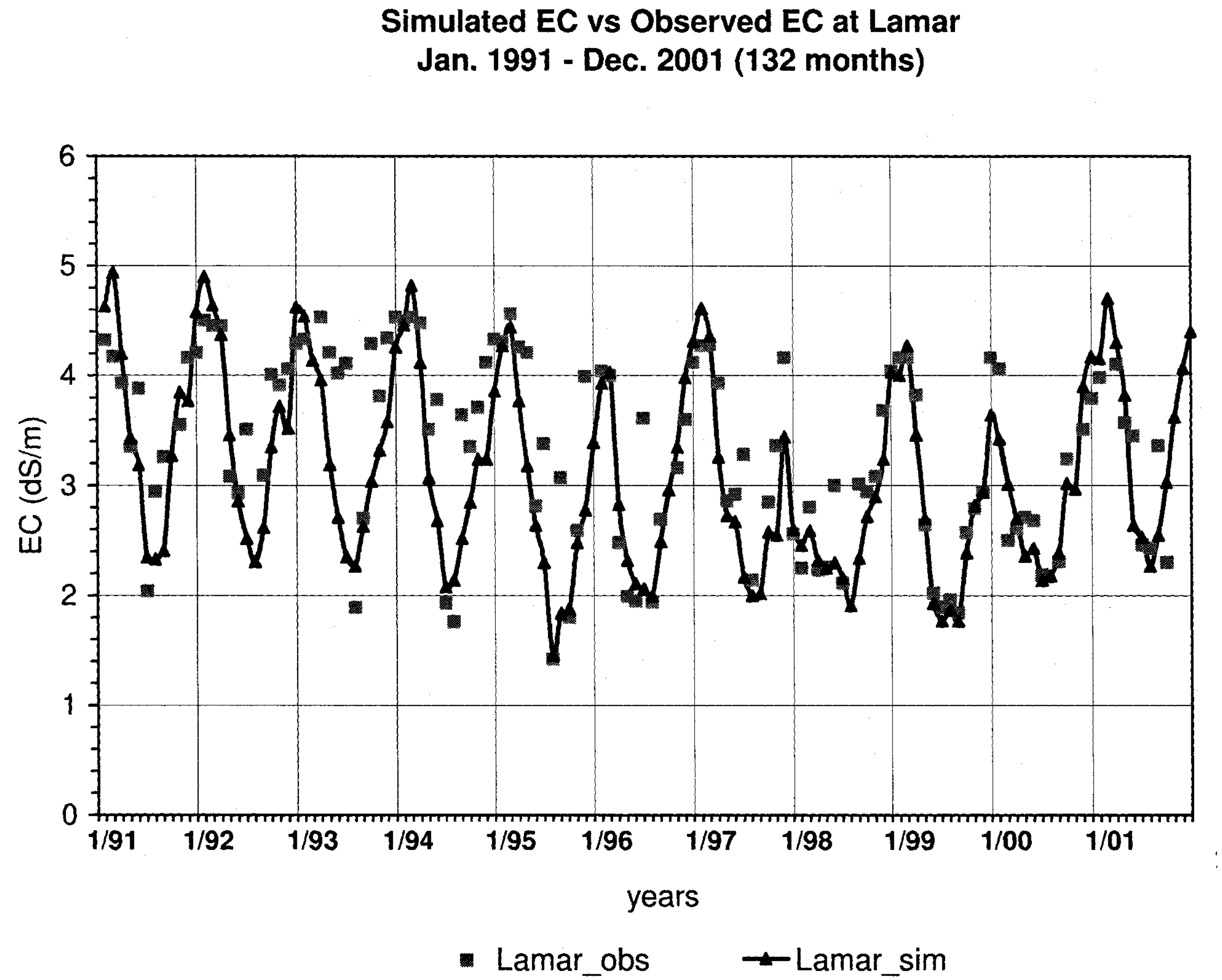


Figure 5.23: Simulated EC vs. Observed EC in the Arkansas River at Lamar (1991-2001, 132 Months)

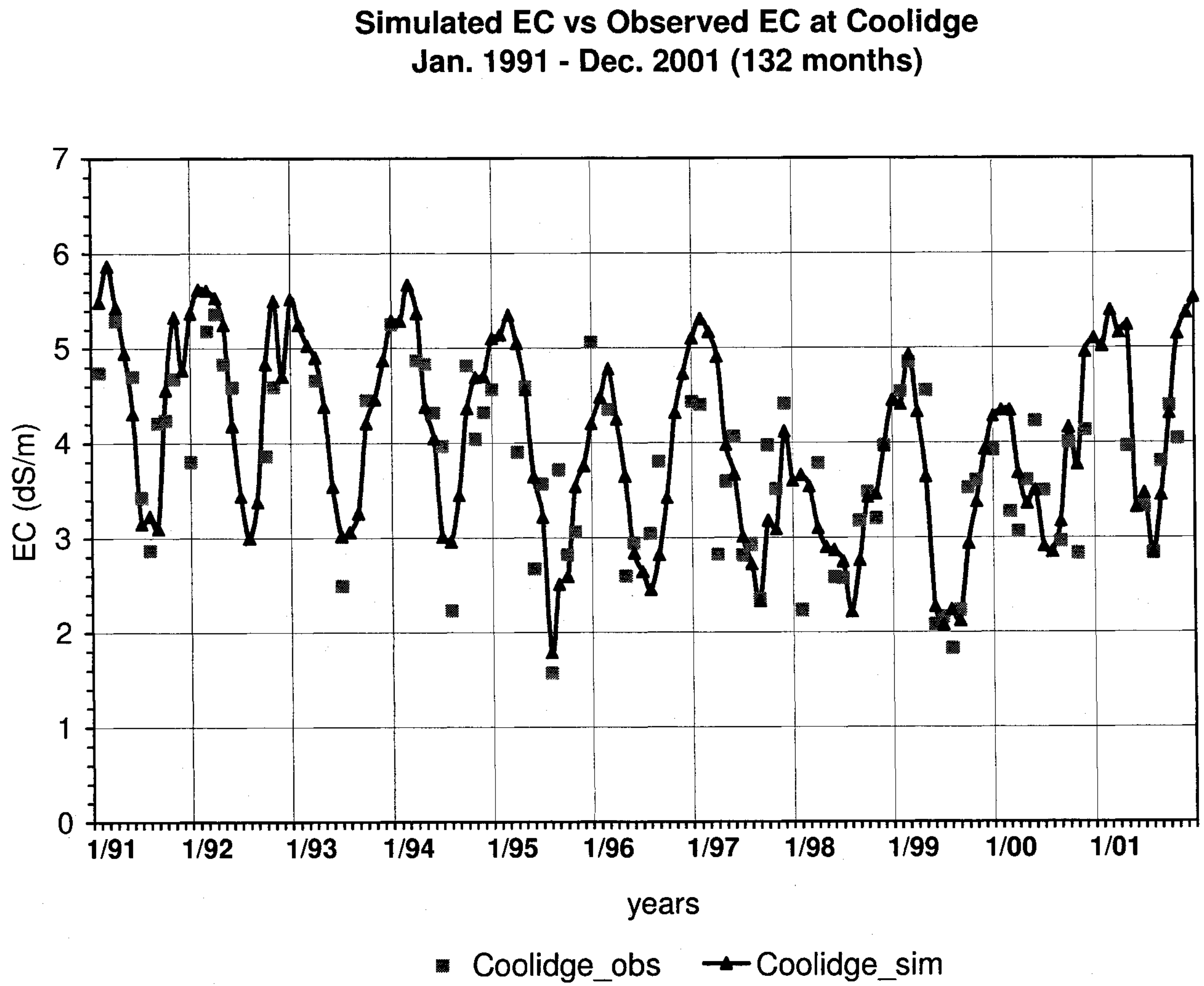


Figure 5.24: Simulated EC vs. Observed EC in the Arkansas River at Coolidge (1991-2001, 132 Months)

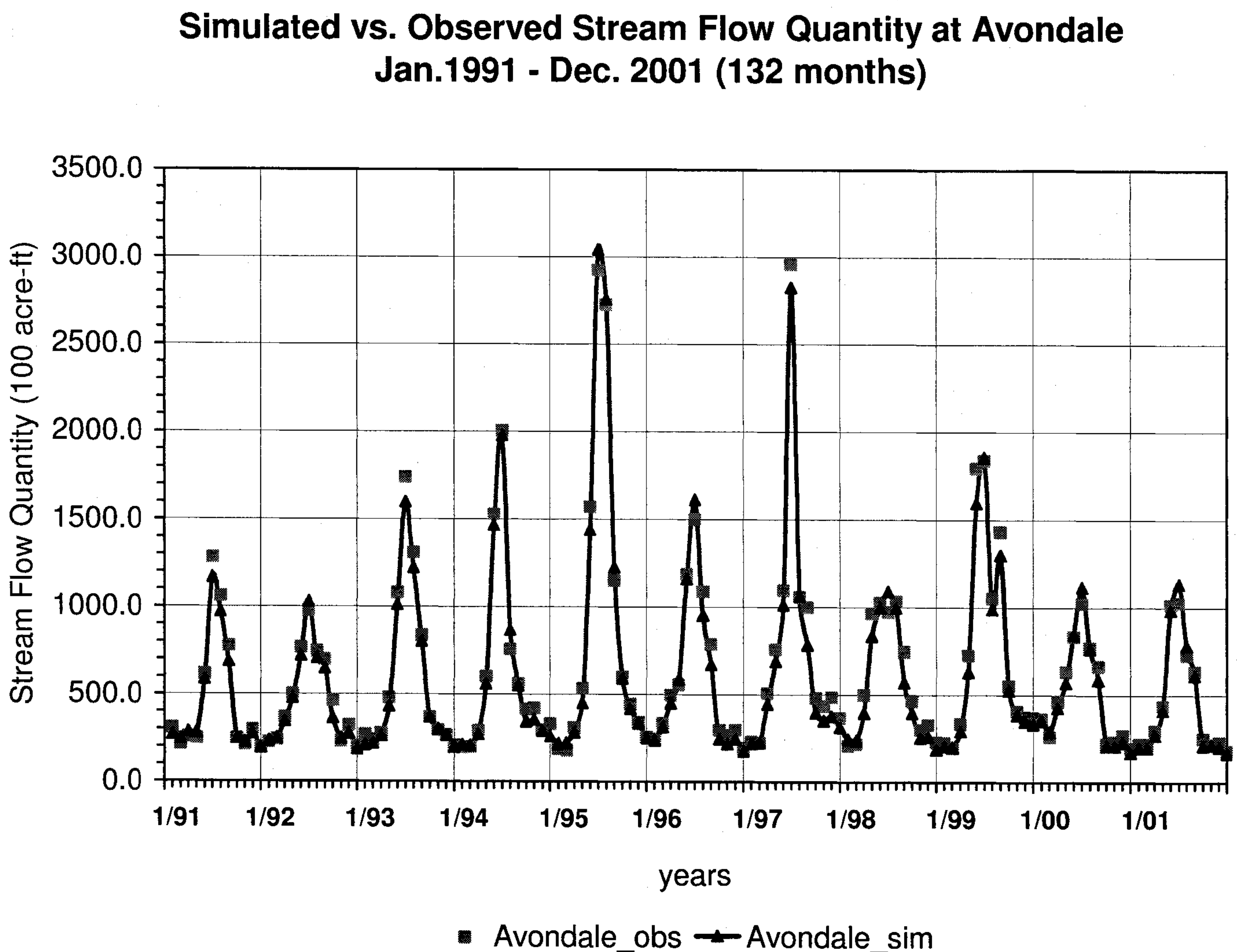


Figure 5.25: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Avondale (1991-2001, 132 Months)

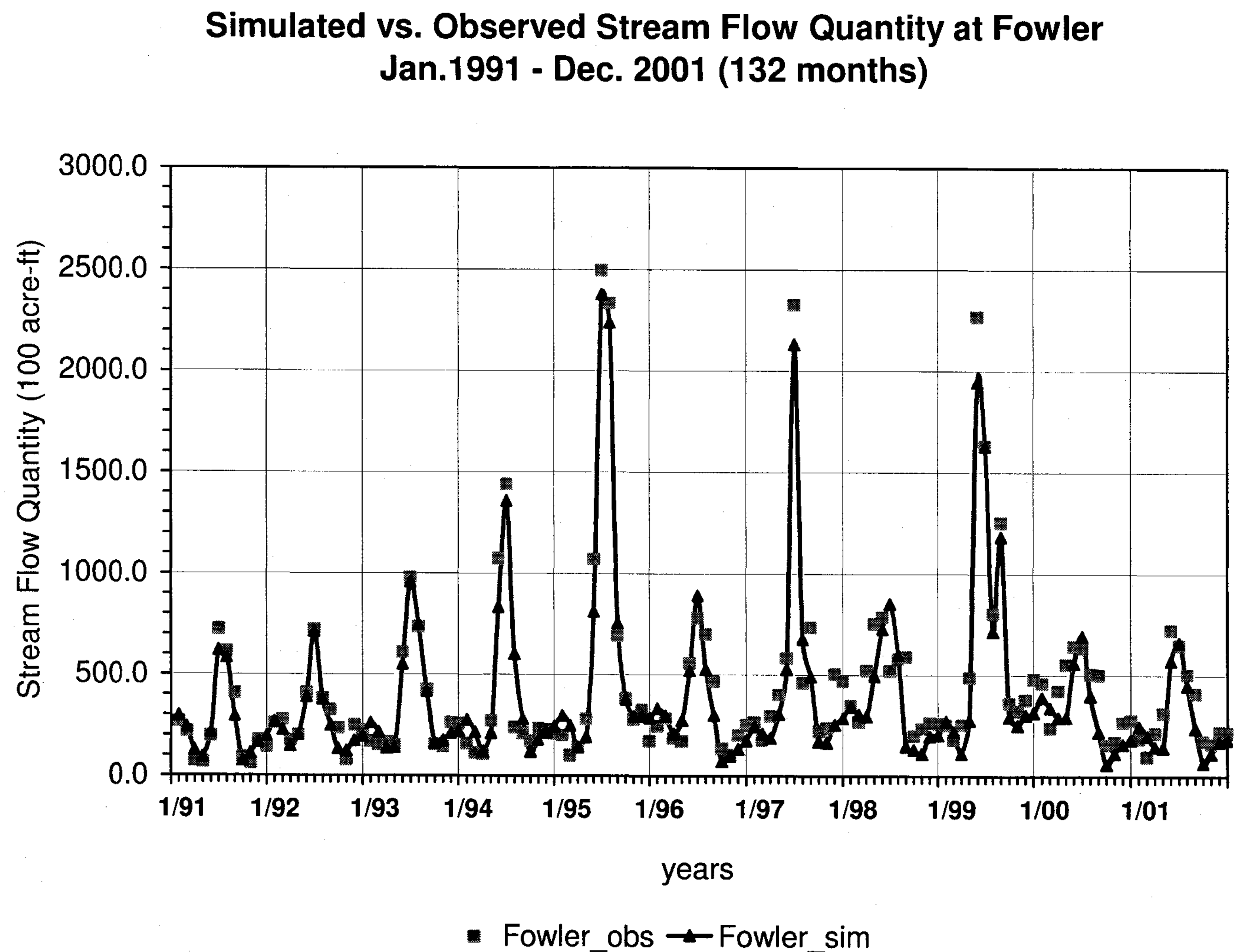


Figure 5.26: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Fowler (1991-2001, 132 Months)

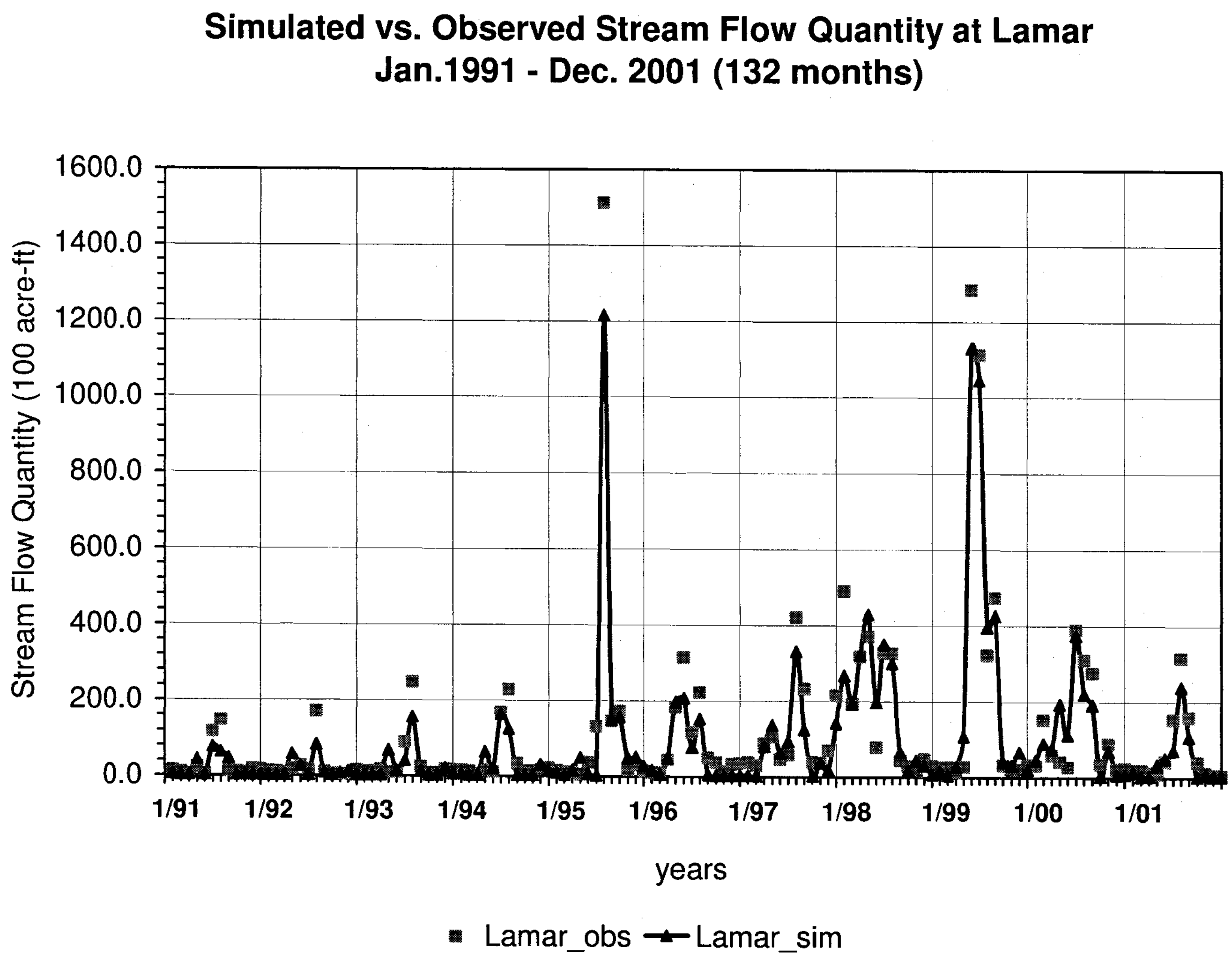


Figure 5.27: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Lamar (1991-2001, 132 Months)

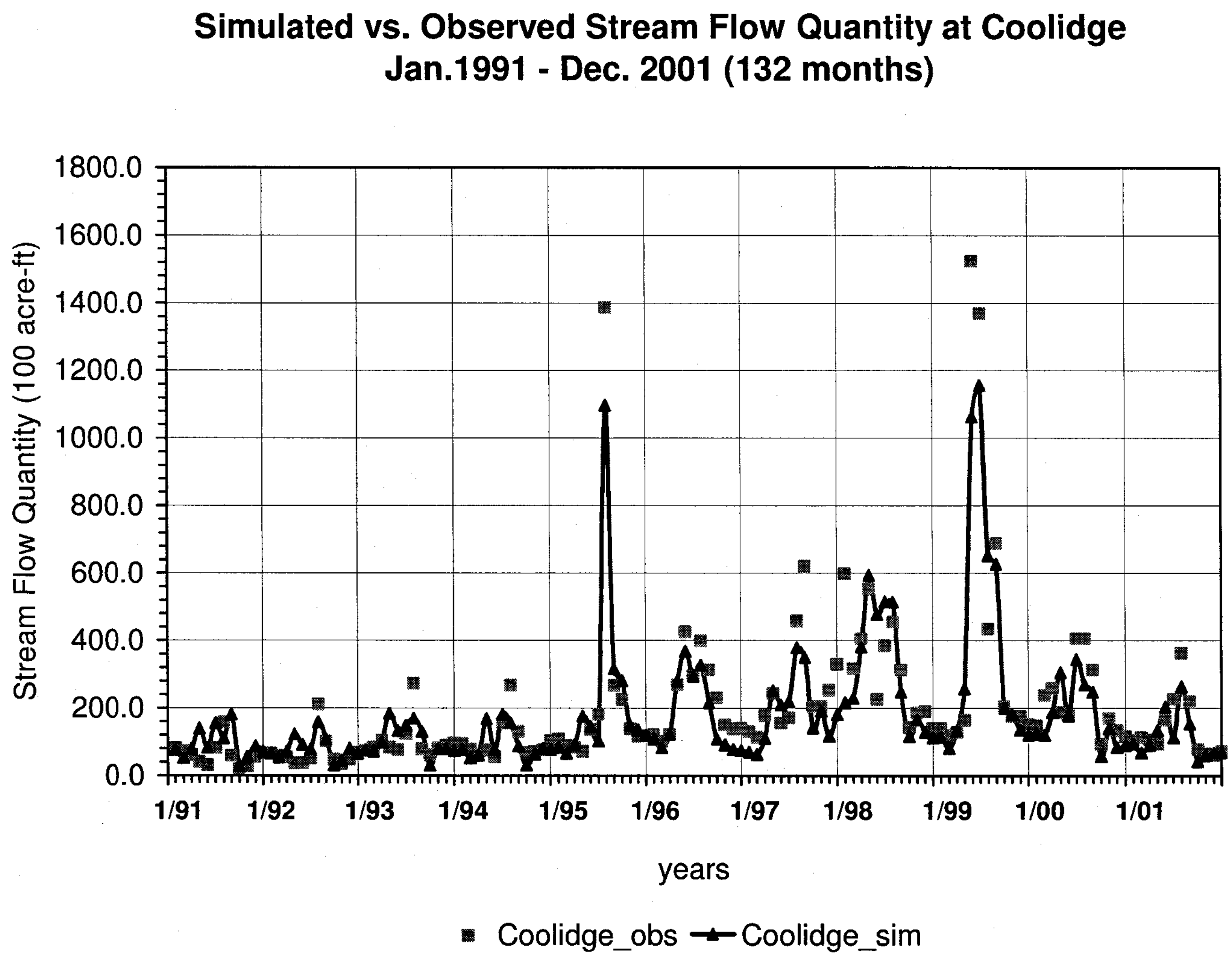


Figure 5.28: Simulated vs. Observed Stream Flow Quantity in the Arkansas River at Coolidge (1991-2001, 132 Months)

As shown in figures 5.20 to 5.28, the simulated EC and the simulated stream flow quantity match observed data well at different locations. The correlation coefficient *r*, as shown in equation (5.6), is used to evaluate the simulation. The results are shown in table 5.9. *r* values obtained for the calibration period (January 1986 to December 1990) are also listed in the table for comparison with those obtained during the validation period. As shown in the table, *r* values for the validation period are as good as those for the calibration period at different river locations. The detail simulation data and the observed data as well as the differences between them in the entire simulation period (January 1986 to December 2001) can be checked in Appendix A and B for further evaluation of how the model is working.

Table 5.9: Correlation Coefficients, *r*, for Simulation in Validation Period

(a) *r* for Water Quality Simulation

	Avondale	Fowler	JM Res	Lamar	Coolidge
***r* for validation**	***0.917***	***0.865***	***0.809***	***0.831***	***0.779***
r for calibration	*0.868*	--	*0.866*	*0.838*	*0.756*

(b) *r* for Water Quantity Simulation

	Avondale	Fowler	Lamar	Coolidge
***r* for validation**	***0.994***	***0.967***	***0.975***	***0.929***
r for calibration	*0.993*	*0.950*	*0.987*	*0.974*

In this study simulated EC represents monthly average EC. Ideally, if the observed EC are also monthly average EC, they can be compared under the same standard. However, as mentioned before, the observed EC data is sampled at the frequency of approximately once a month. The data actually are not on an "average" base. Therefore, the *r* value for water quality simulation could be improved if there were more observed EC data.

In the calibration and validation process, there are some major water budget components such as tributary inflow, phreatophyte consumption, groundwater baseflow and river channel loss ratio used in these periods. These components are shown in table 5.10.

Table 5.10: Major Water Budget Components Used for Calibration and Validation

(a) Tributary Inflow, Phreatophyte Consumption and Groundwater Baseflow

units: acre-ft

	Calibration, 1986-1990	Validation, 1991-1996
Tributary Inflow	*848,013*	*1,924,300*
Phreatophyte Consumption	*410,078*	*729,067*
Groundwater Baseflow	*1,357,032*	*2,985,470*

(b) Monthly River Channel Loss Ratio

Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
.474	***.450***	***.360***	***.230***	***.121***	***.080***	***.080***	***.121***	***.230***	***.360***	***.450***	***.474***

5.4 Program Application and Limitation

One objective of this study is to develop a river quantity and quality simulation program that provides a user friendly environment for inputting data, executing the simulation and viewing the results.

To operate the simulation program, the user must first open a project. Then, all required input data and parameters need to be entered into the program via the graphical user interface (GUI). The methodology for preparing input data and parameters for the Arkansas River from 1986 to 1990 has been described in section 5.2.

After a project is opened, the irrigation network layout (figure 5.7) is displayed. By clicking on the reaches or nodes, corresponding property sheets will pop up to assist the user in the input process. Figure 5.29 is an example that shows a property sheet of the input data for a canal service area. The input data are organized into different categories: precipitation, groundwater (GW) pumping, crop consumptive use, soil treatment, surface water (SW) return flows, GW return flows and stream depletion. The user clicks on each category and keys in the appropriate data and parameters. Figure 5.30 is an example that shows a property sheet for data input for a canal. As shown in the figure, the input for a canal includes the canal transit loss ratio and return flow response function due to canal transit loss in the simulation years. Figure 5.31 shows a property sheet for data input for a river reach. The input for a river reach is organized into four categories: tributary inflows, phreatophyte consumption, river channel loss and base flows. The user clicks on each category and enters the appropriate data and parameters for the simulation period.

After all input data and parameters are set up, the program can be executed. Figure 5.32 shows a layout that displays the simulation results for the Arkansas River. As a circle node or a square node is clicked, a corresponding graph will pop up. Figure 5.33 shows simulated river quantity and quality at a river reach (reach 8). In the figure, the monthly volume of the Arkansas River and the chemical composition from 1986 to 1990 is shown. Figure 5.34 shows simulated soil water composition in the Rocky Ford Canal service area. The simulation output can also be displayed as river quantity and quality for the entire river in a specific month. Figure 5.35 is an example that illustrates volume and chemical composition of the Arkansas River versus river mileage in June, 1987.

With this salinity simulation program, users can set up and modify data and parameters conveniently. The program also provides for graphical display of simulation results. With the graphs, users can view the spatial and temporal variation of river quantity and quality in the entire river basin.

As shown in the previous sections, the salinity simulation is successfully applied on the Arkansas River Basin in Colorado from 1986 to 1990 and from 1991 to 2001. It should be noted that the simulation for these two periods are based upon historical data such as precipitation, tributary inflow (both quantity and quality) and river diversion. If this program is going to be applied to future years for forecasting purposes, the input data must be estimated with appropriate programs separately. This program does not include any stochastic features for the prediction of hydrologic components. Neither does this program include water rights simulations to predict river diversions.

It also should be noted that the irrigation network shown in figure 5.7 is based on geographical characteristics of the Arkansas River Basin from Pueblo to the Kansas state line. The figure reflects the relative locations of different canal service areas, reservoirs and river reaches. If the program is applied to other watersheds, the GUI would need to be modified based on the geographical characteristics of the targeted watershed.

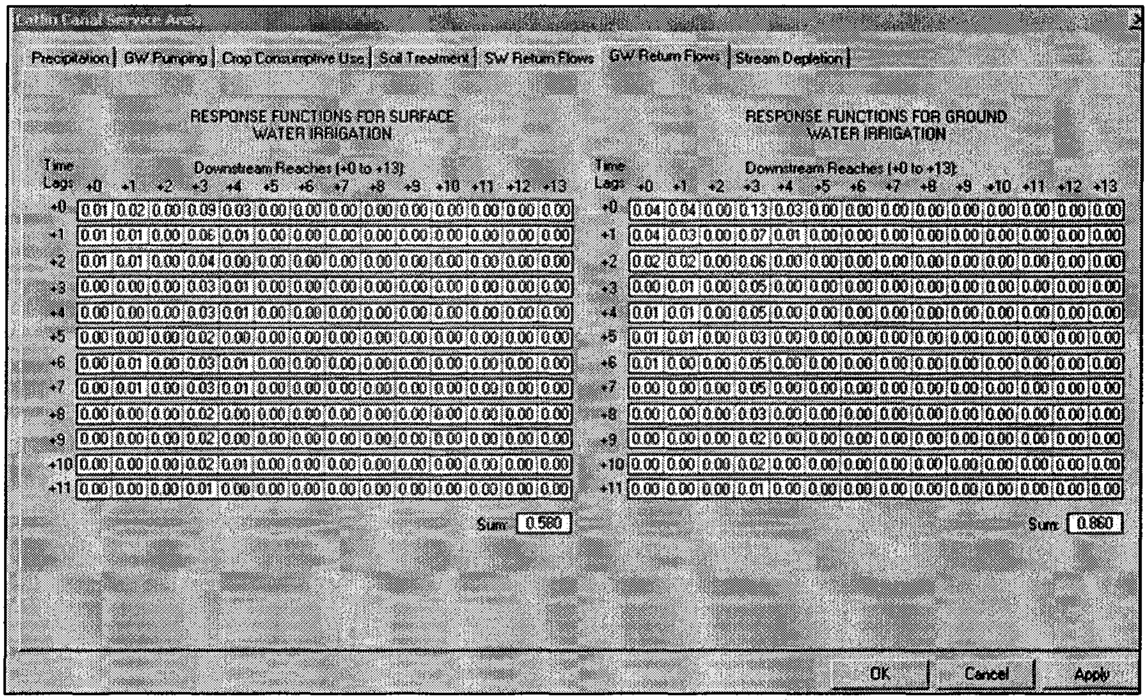


Figure 5.29 Property Sheets for Input for Canal Service Areas

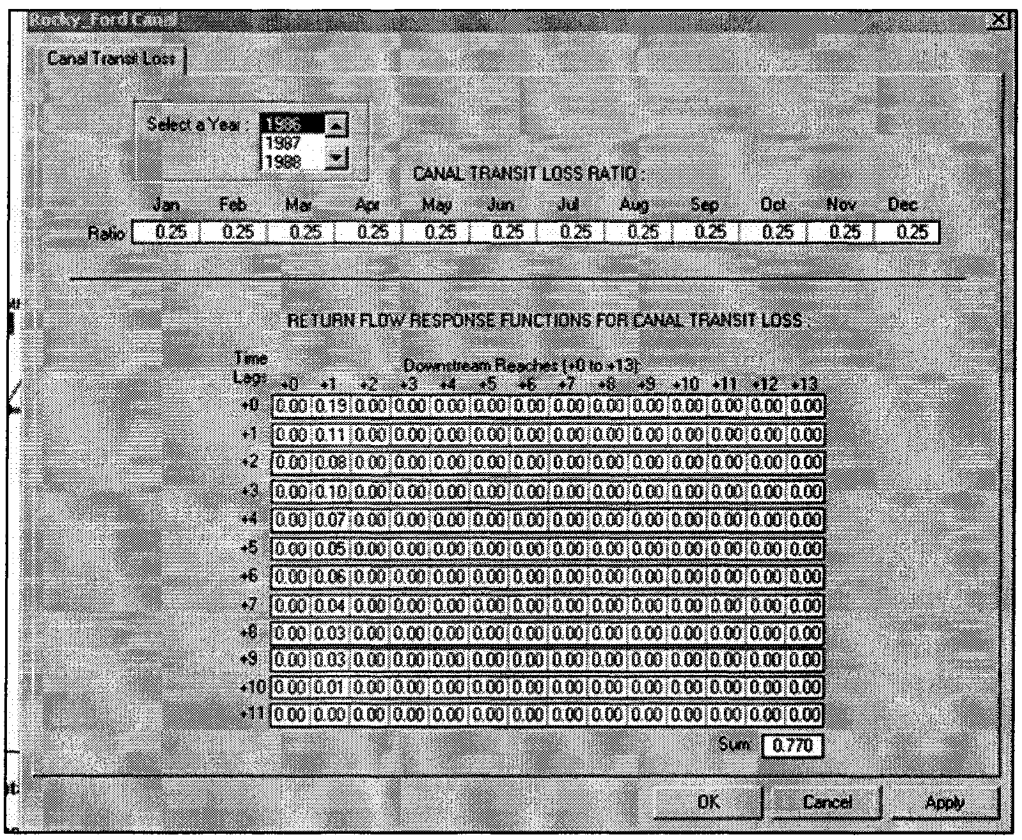


Figure 5.30 Property Sheets for Input for Canals

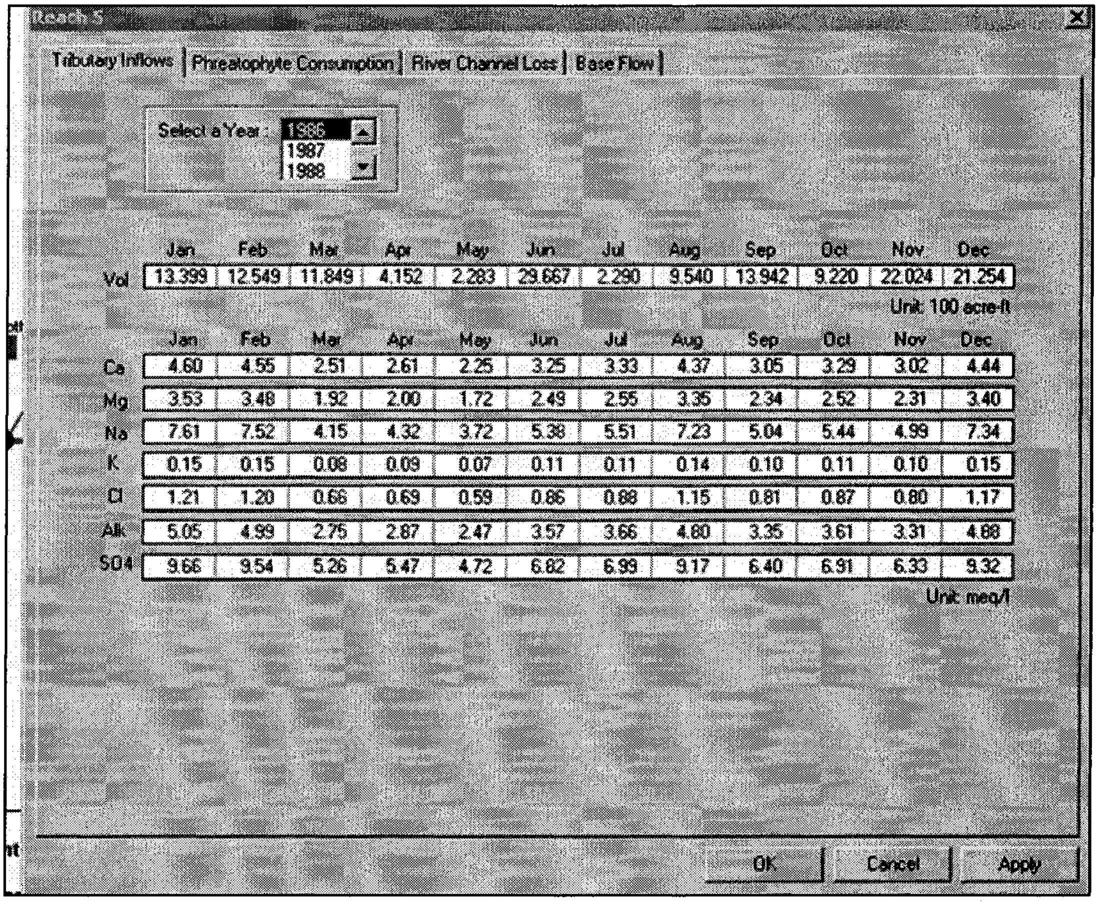


Figure 5.31 Property Sheets for Input for River Reaches

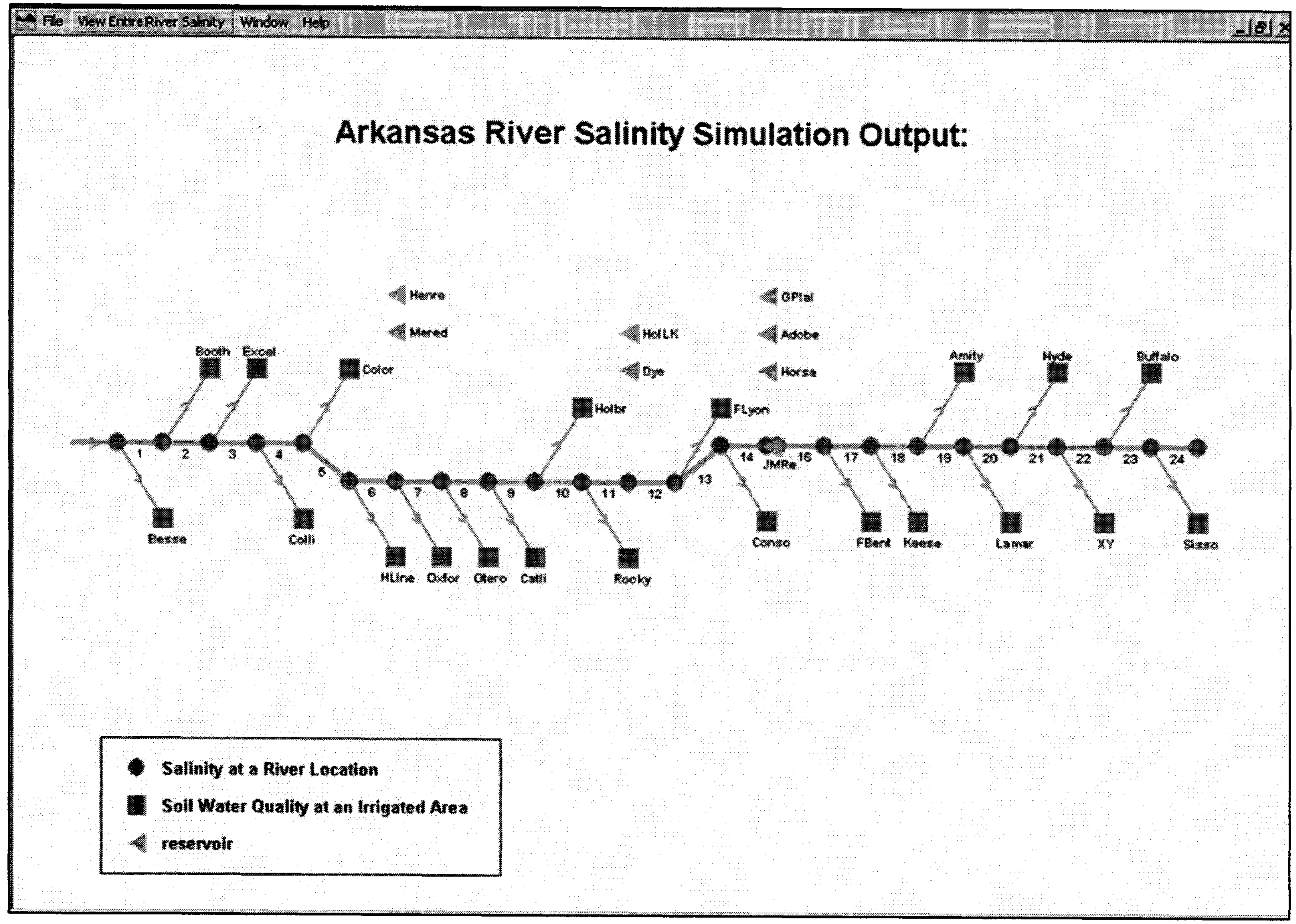


Figure 5.32: Simulation Output Display from the Arkansas River Network

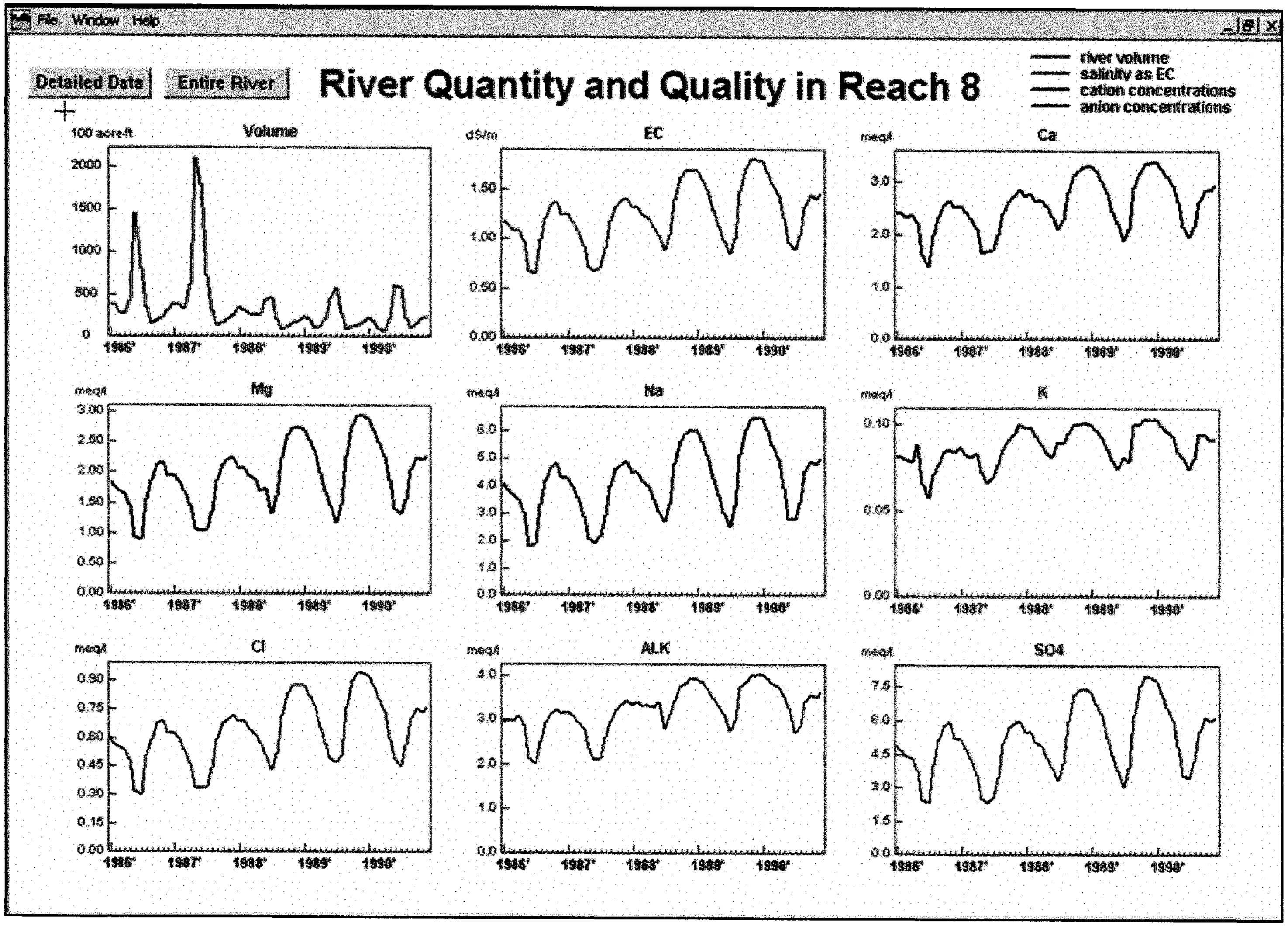


Figure 5.33: Simulated River Quantity and Quality in a Reach

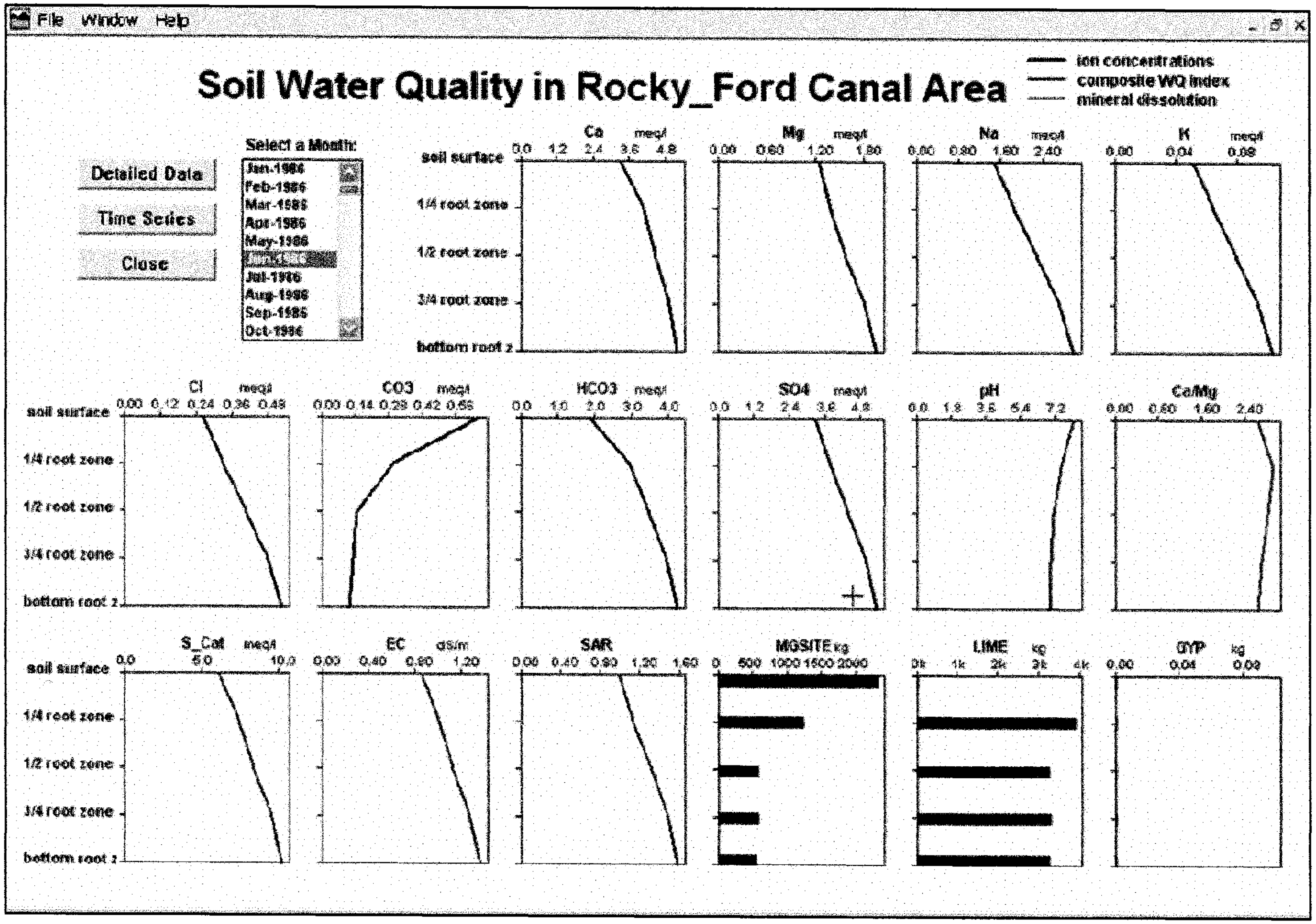


Figure 5.34: Simulated Soil Water Composition in a Canal Service Area

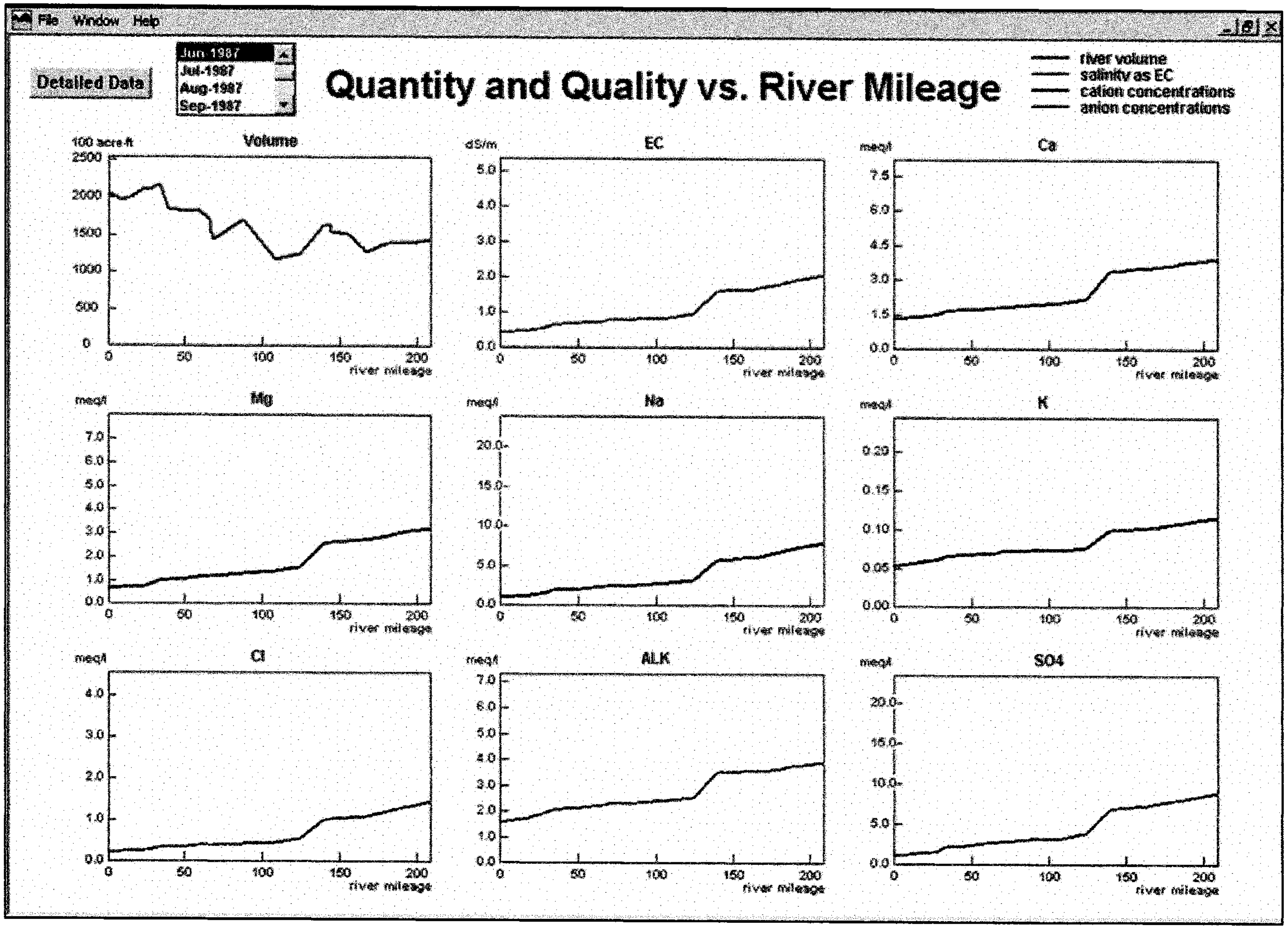


Figure 5.35: Simulated River Quantity and Quality versus River Mileage

CHAPTER 6

DETAILED RESULTS AND SENSITIVITY ANALYSIS

In chapter 5, the salinity simulation model is calibrated for the Arkansas River Basin in Colorado for the period from 1986 to 1990. The calibrated model is also validated for the period from 1991 to 2001. In this chapter, detailed results of the river salinity simulation from 1997 to 2001 are shown (these period was selected so that enough detail can be shown in the output graphs). Then, the salinity model is analyzed for its sensitivity to selected parameters.

6.1 Hydro-salinity Simulation from 1997 to 2001

In chapter 5, the simulated river quantity and quality are shown at five locations along the Arkansas River. In this section, the simulated river quantity and quality as well as soil water salinity are shown from upstream to downstream. The Arkansas River is divided into 24 reaches based on the locations of canal intakes. Figure 6.1 shows the distribution of the river reaches.

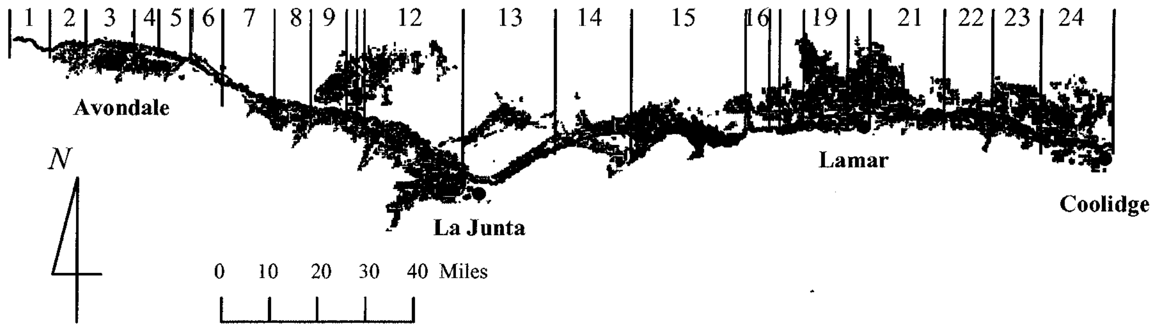


Figure 6.1: Distribution of the 24 Reaches in the Arkansas River

6.1.1 River Quantity Results from 1997 to 2001

As described in section 3.3, the simulation of river quantity is based on applying the mass balance equation to each river reach (equation 3.50). Figures 6.2 to 6.6 show the simulated monthly river quantity (100 acre-ft) versus river mileage in the Arkansas River from January, 1997 to December, 2001. Each figure contains simulation data from January to December so that the seasonal fluctuation of the river quantity may be observed. The notes "R1", "R3", .., "R24", shown in the figures indicate river reach 1, reach 3,..., reach 24. It should be noted that John Martin Reservoir is located in river reach 15 (at river miles 124 to 140.4). The reservoir volume is simulated and is shown in figure 6.7.

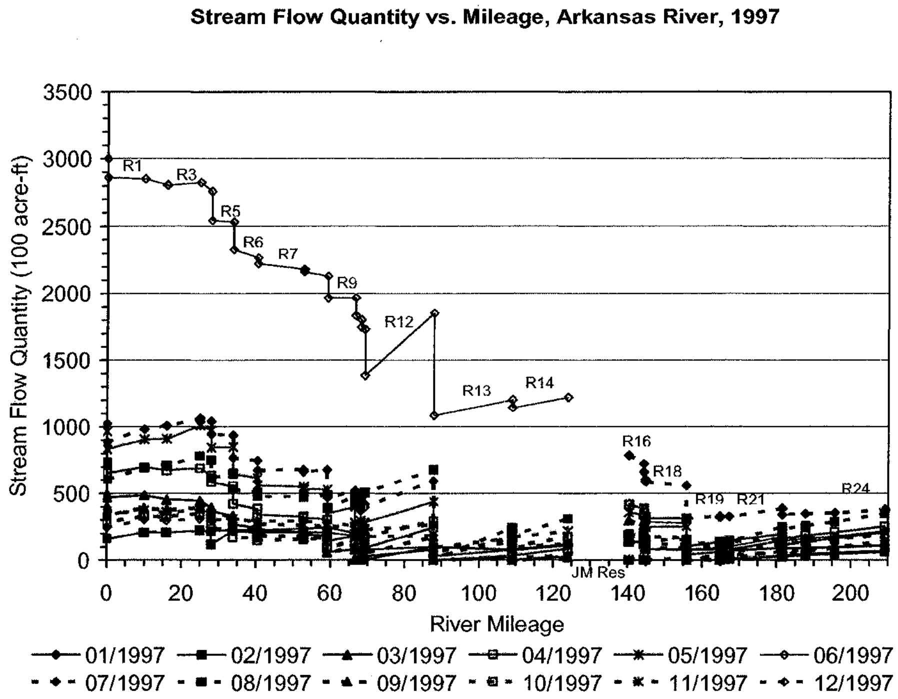


Figure 6.2: Simulated Stream Flow Quantity vs. River Mileage in the Arkansas River in 1997

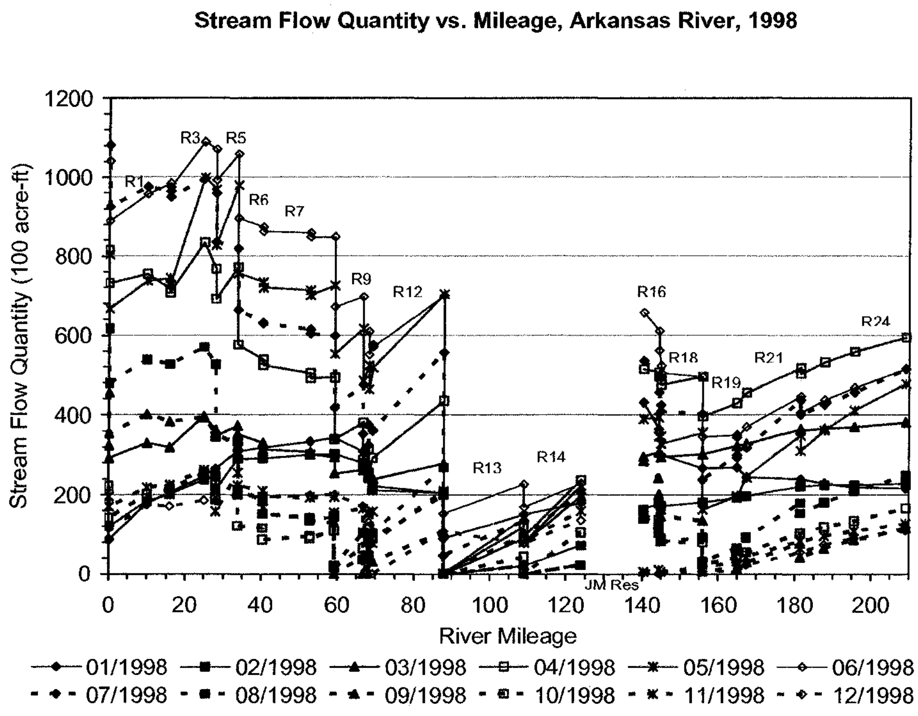


Figure 6.3: Simulated Stream Flow Quantity vs. River Mileage in the Arkansas River in 1998

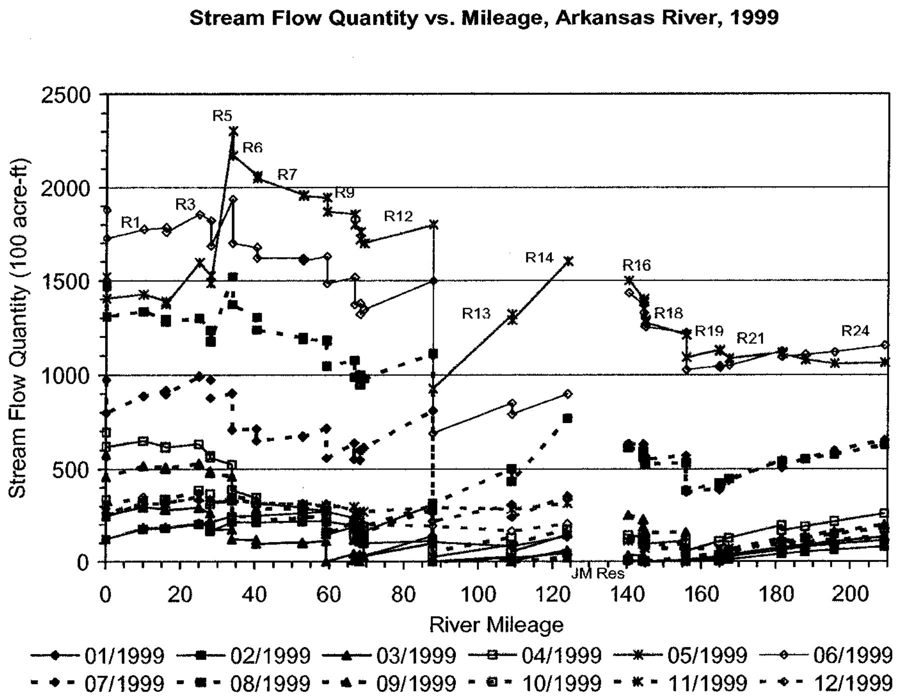


Figure 6.4: Simulated Stream Flow Quantity vs. River Mileage in the Arkansas River in 1999

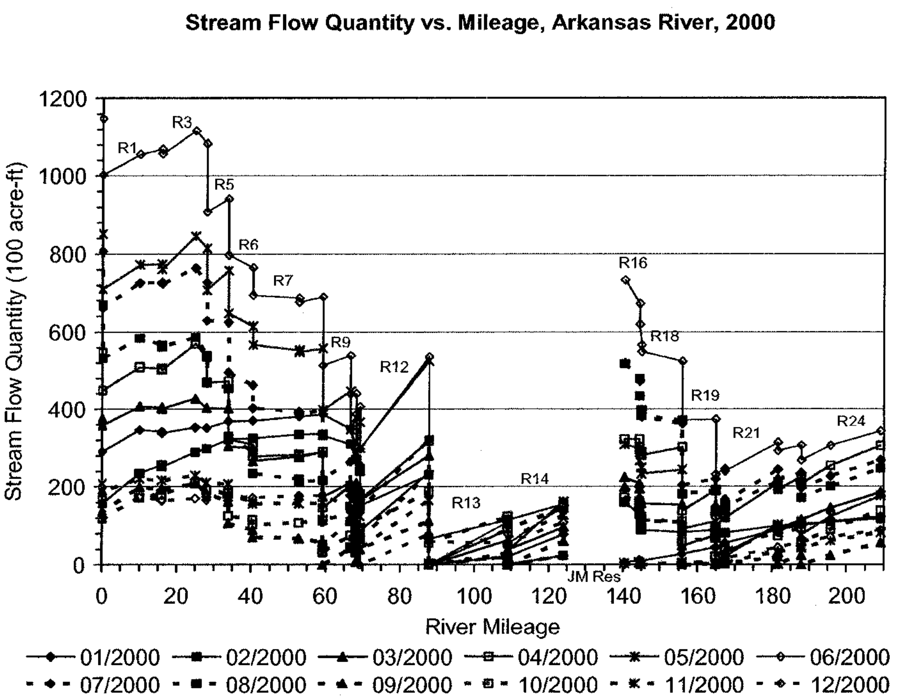


Figure 6.5: Simulated Stream Flow Quantity vs. River Mileage in the Arkansas River in 2000

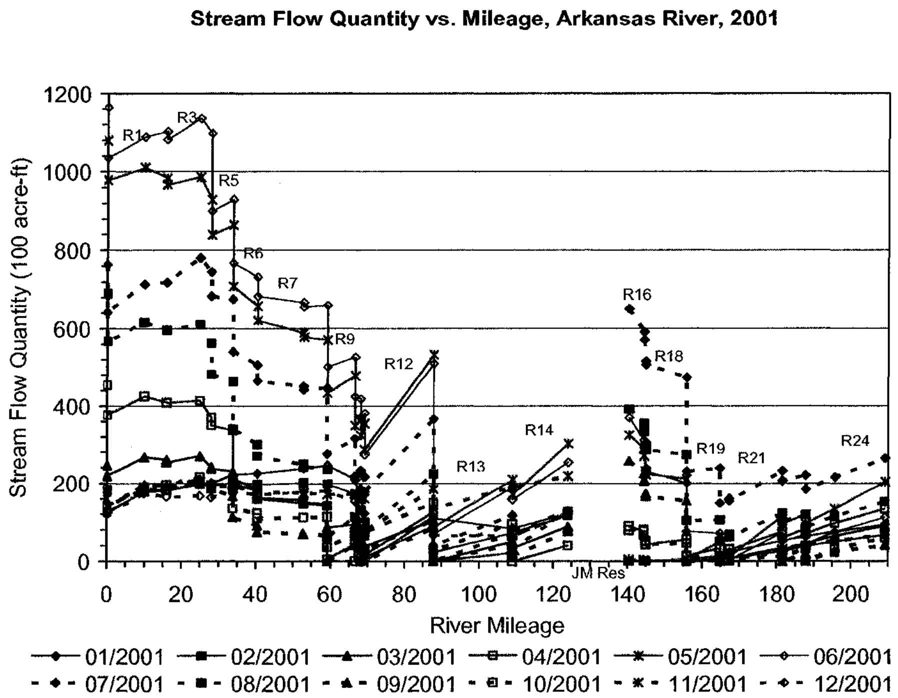


Figure 6.6: Simulated Stream Flow Quantity vs. River Mileage in the Arkansas River in 2001

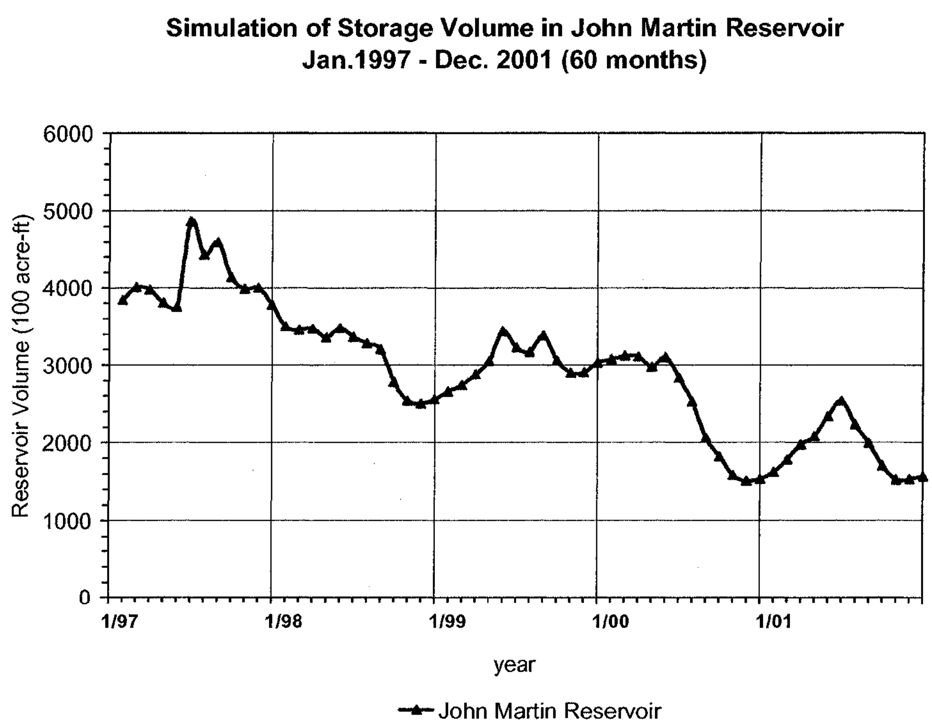


Figure 6.7: Simulation of Storage Volume in John Martin Reservoir, 1997 to 2001 (60 months)

River quantity fluctuates seasonally during the simulation period. As shown in figures 6.2 to 6.6, river quantity is higher during the irrigation season and lower in the off-season. In reach 1 (denoted as R1 in the figures), river quantity in May, June and July can be five times greater than river quantity in December, January and February. This is a result of the releases from Pueblo Reservoir (located upstream of reach 1) into the river to meet the demand for irrigation water in the Arkansas River Basin during the growing season. As the river water travels from upstream reaches to downstream reaches, it is diverted by canals, and the river quantity decreases. The Fort Lyon Canal, the canal which diverts water at the head of reach 13 (at river mile 87.8 in the figures), is the canal which owns the most water shares in the Arkansas River. River quantity usually decreases significantly downstream of the Fort Lyon Canal diversion. The river flows into John Martin Reservoir at the end of reach 14. The monthly volume of John Martin Reservoir from January 1997 to December 2001 is shown in figure 6.7. John Martin Reservoir releases to the Arkansas River at the head of reach 16. The amount of the release is based on the demands of users along reach 16 to reach 24 as well as demands from the State of Kansas.

As shown in figures 6.2 to 6.6, river quantity changes within a reach. For example, in June 1997 (figure 6.2), after the Fort Lyon Storage Canal diversion at the head of reach 12, river quantity increases from 139,437 acre-ft to 189,798 acre-ft at the end of reach 12. As described in section 3.3, river quantity is affected by several factors including irrigation tail water, groundwater return flow, drainage flow, tributary inflow, river channel loss, river bank phreatophyte consumption, and river depletion due to pumping. The net effect of these factors will determine whether the river quantity within a reach increases or decreases. Generally, the increase of river quantity in reach 12 and 21 during the irrigation season can be attributed to return flows from nearby major irrigation areas. Reach 12 is located near the Rocky Ford Canal, Catlin Canal and Highline Canal irrigation areas. Reach 21 is located near the irrigation area of the Fort Lyon Canal. The tail water, drainage, as well as groundwater flow created from these areas substantially influences the neighboring river reaches. In addition to reaches 12 and 21, reaches 13, 14, 22, 23, and 24 also receive significant amounts of irrigation return flows. The increase of river quantity in reaches 3 and 5 can be attributed to tributary inflows. Two major tributaries, the St. Charles River and the Huerfano River, flow into reach 3 and reach 5 respectively. The decrease in river quantity in reaches 4, 6 and 16 can be attributed to channel losses.

6.1.2 River Quality Results from 1997 to 2001

As described in section 3.3, simulation of river quality is based on calculating the mixture of different return flows in each reach. Concentrations of Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, SO_4^{2-} and alkalinity in the river water are calculated. River EC (dS/m) can then be calculated based on the concentration of these major ions. Figures 6.8 to 6.12 show simulated river EC (dS/m) versus river mileage in the Arkansas River from 1997 to 2001. Each figure shows the simulation results from January to December so that the seasonal fluctuation of the river EC can be observed. The notes "R1", "R3", .. ,"R24", shown in the figures indicate river reach 1, reach 3,..., reach 24. The note "JM Res" shown in the figure indicates John Martin Reservoir which is located in river reach 15 between miles 124 and 140.4. Figure 6.13 shows simulated reservoir EC in John Martin Reservoir.

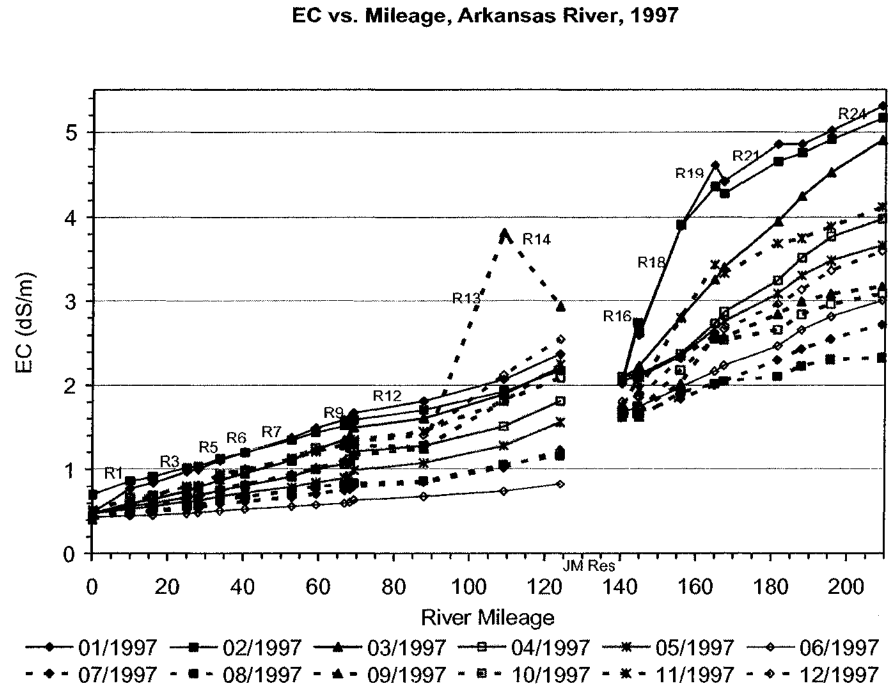


Figure 6.8: Simulated EC vs. River Mileage in the Arkansas River in 1997

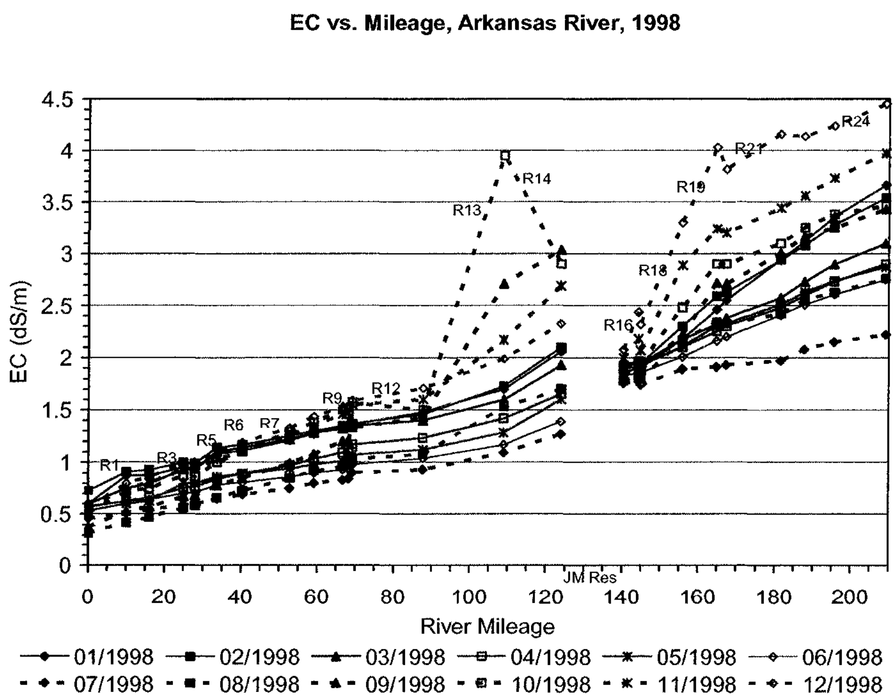


Figure 6.9: Simulated EC vs. River Mileage in the Arkansas River in 1998

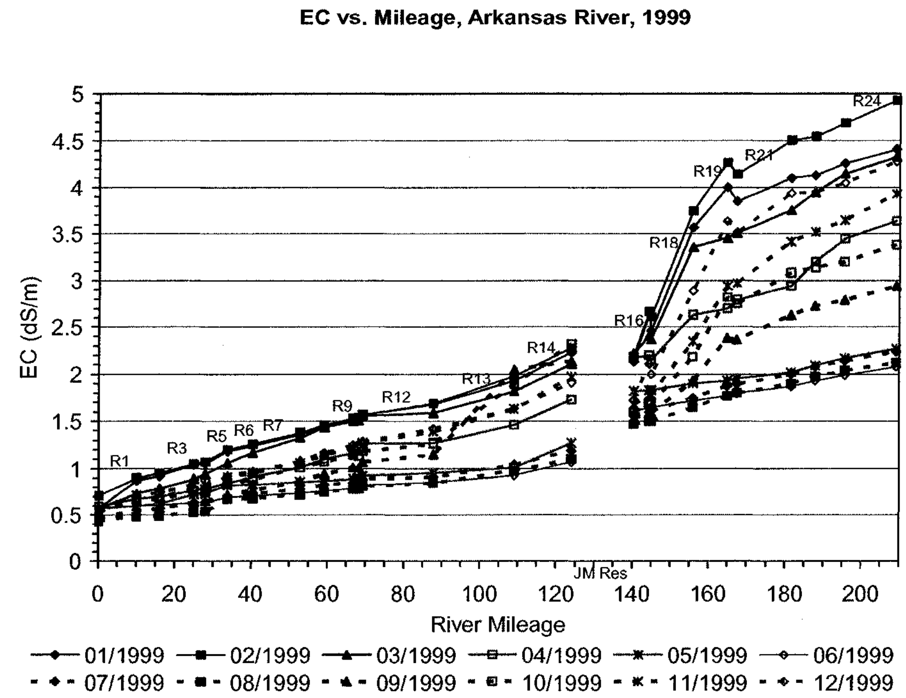


Figure 6.10: Simulated EC vs. River Mileage in the Arkansas River in 1999

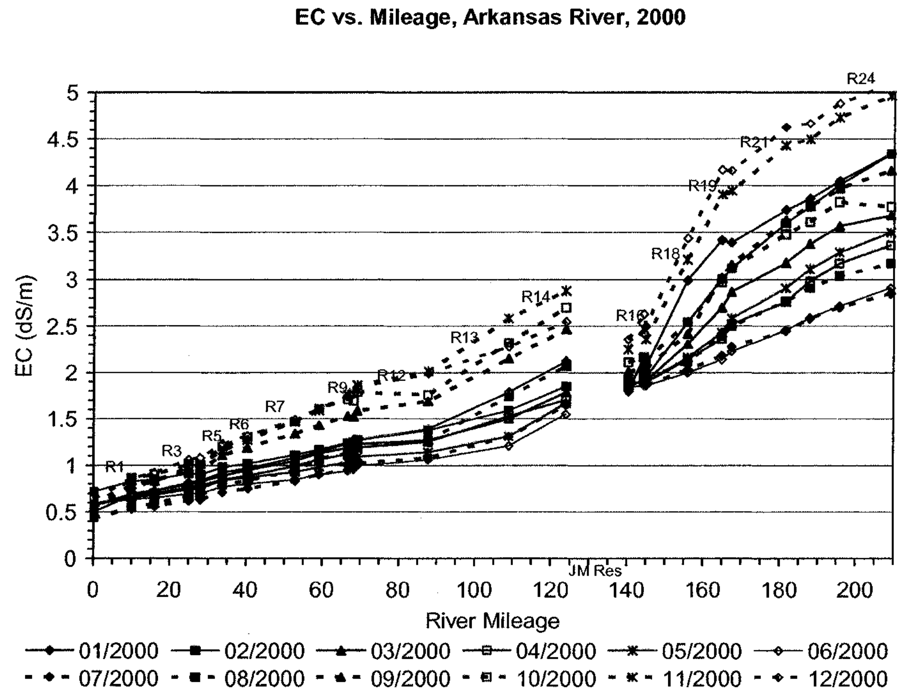


Figure 6.11: Simulated EC vs. River Mileage in the Arkansas River in 2000

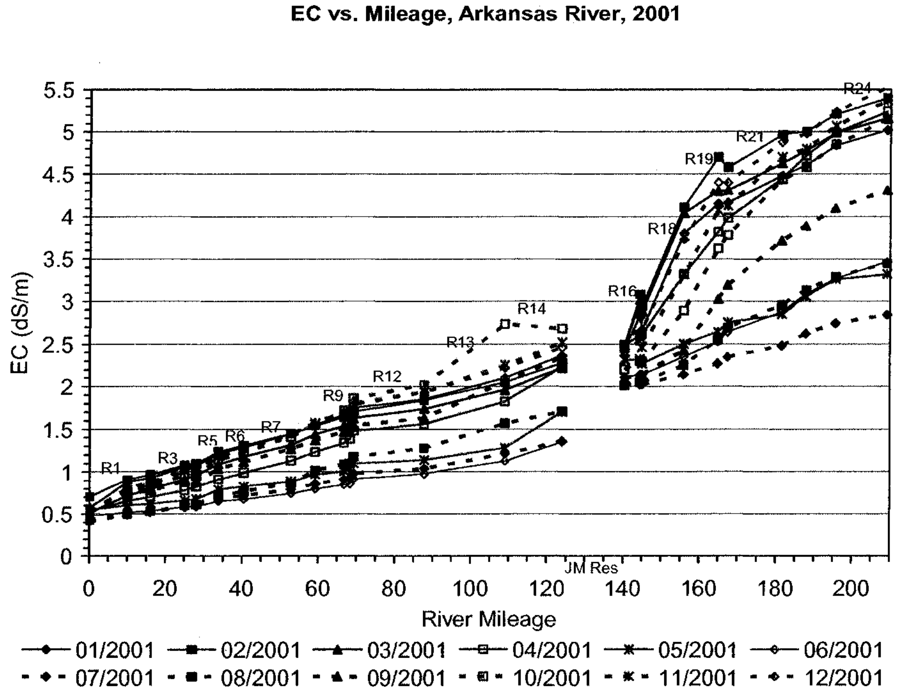


Figure 6.12: Simulated EC vs. River Mileage in the Arkansas River in 2001

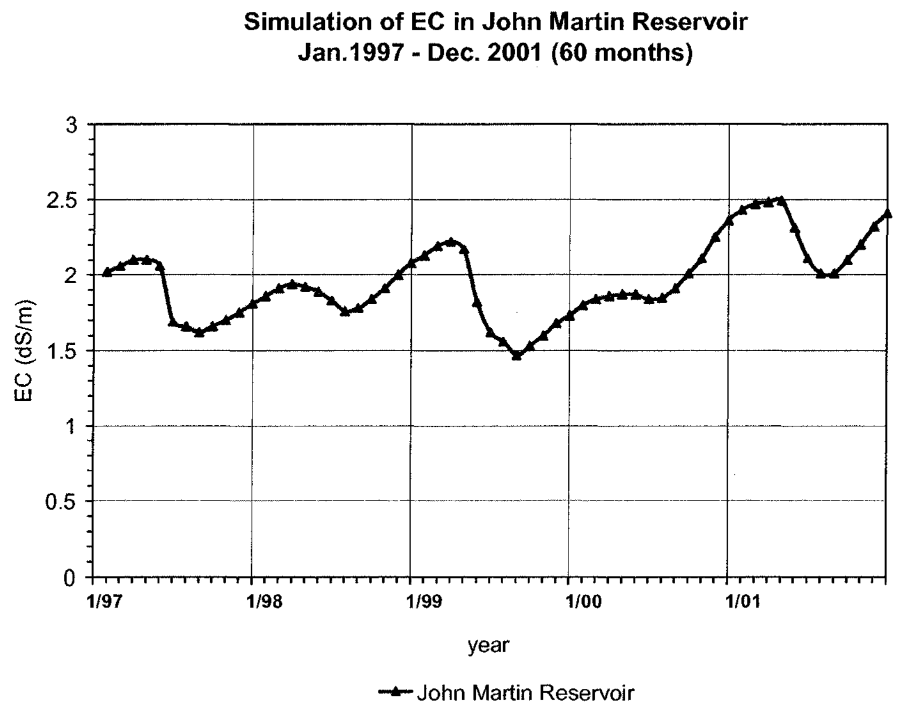


Figure 6.13: Simulation of EC in John Martin Reservoir, 1997 to 2001 (60 months)

As shown in figures 6.8 to 6.12, river EC shows seasonal fluctuation. The river EC is higher during the off-season and is lower during the irrigation season. In each year, the highest river EC month is usually December, January or February and the lowest EC month is usually June, July or August.

Basically, river water in each reach contains two major portions: groundwater and surface water. The groundwater portion includes groundwater return flow, base flow, and subsurface drainage flow. The surface water portion includes tributary inflow, irrigation tail water, and inflow from the adjacent upstream reach. The salinity of groundwater is usually much higher than that of surface water because groundwater will carry salts from the alluvium. Therefore, if the river water contains a high percentage of groundwater, river EC level will be high. During the off-season, the river is primarily fed by groundwater base flow. There is only a limited amount of surface water in the river. Therefore, this results in much higher river EC during the off-season than during the irrigation season.

In addition to the seasonal fluctuation, river EC also shows significant changes from upstream reaches to downstream reaches. During the simulation period, the average river EC was 0.53 dS/m at the head of reach 1 and 3.50 dS/m at the end of reach 24; over a 210-mile stretch of river, the average river EC increases almost sevenfold. This is a result of groundwater flow bringing salts into the river and salts accumulate as the river goes from upstream reaches to downstream reaches.

Figure 6.13 shows simulated reservoir EC in John Martin Reservoir. Salinity of the reservoir water ranges from 1.42 to 2.40 dS/m. Since the reservoir will store inflows from a number of months, its salinity fluctuation is less evident than that of the river.

6.1.3 Soil Water Salinity Results from 1997 to 2001

As described in Section 3.2. The salinity model can simulate soil water composition at five soil depths (soil surface, ¼, ½, ¾ and bottom of the root zone soil depths) for each canal service area. The soil water composition includes concentrations of Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}, SO_4^{2-}, HCO_3^{-} and CO_3^{2-}. In this section, the simulation results in five canal service areas are presented. They are the service areas of the following canals: Bessemer, Colorado, Rocky Ford, Fort Bent, and Amity. Five canals were selected for simulation, rather than all twenty-one canals in the basin, because these five canals are representative of the conditions in the whole basin. The selected canals are all major canals in the Arkansas River and their service areas cover upstream, mid-stream, and downstream portions of the whole study area. Soil water EC (dS/m) at five depths are presented for these canal service areas in figures 6.14 to 6.18.

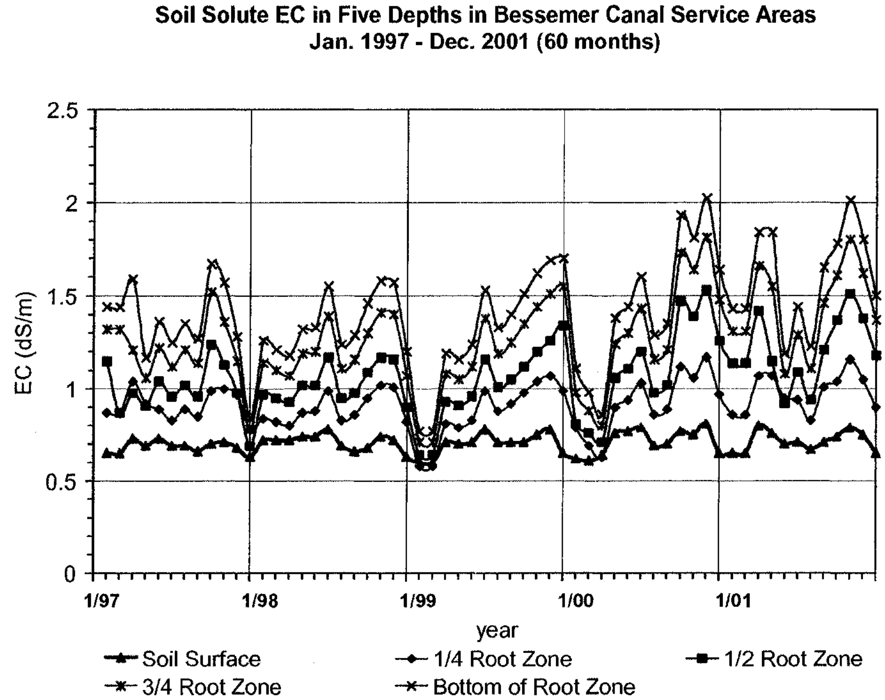


Figure 6.14: EC of Soil Water at Five Soil Depths in the Bessemer Canal Service Area, 1997 to 2001 (60 months)

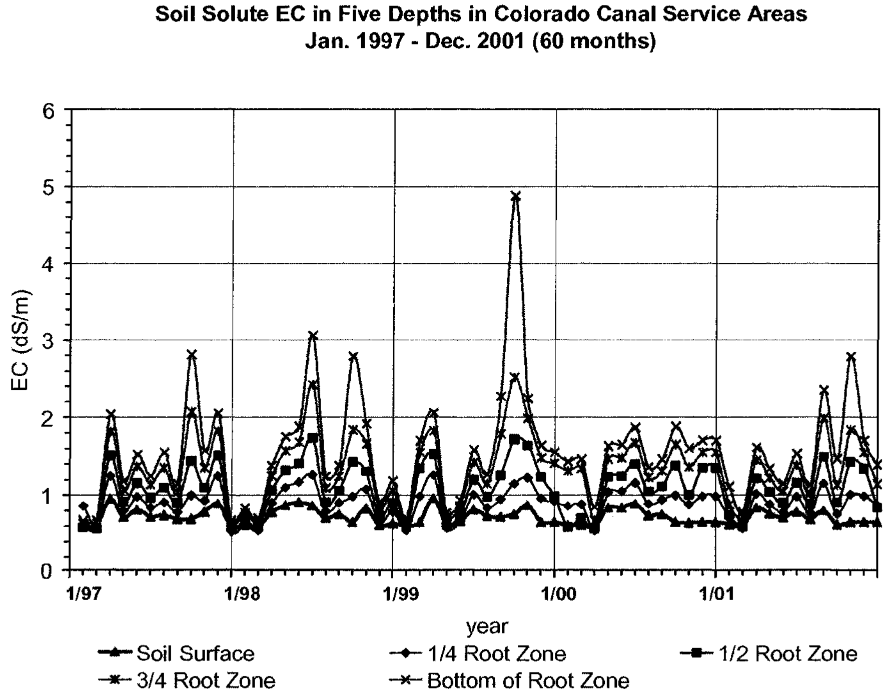


Figure 6.15: EC of Soil Water at Five Soil Depths in the Colorado Canal Service Area, 1997 to 2001 (60 months)

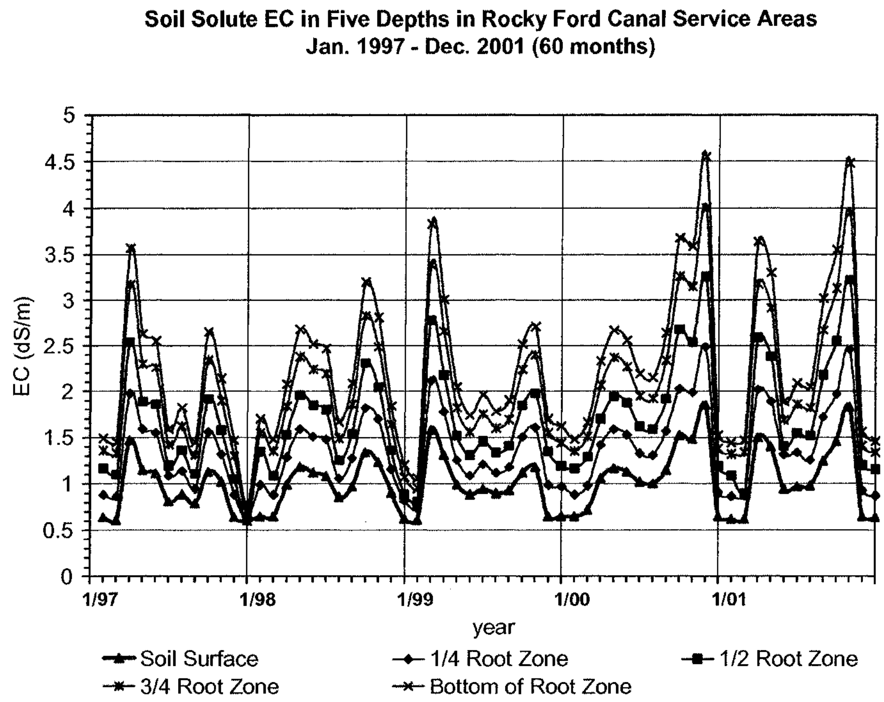


Figure 6.16: EC of Soil Water at Five Soil Depths in the Rocky Ford Canal Service Area, 1997 to 2001 (60 months)

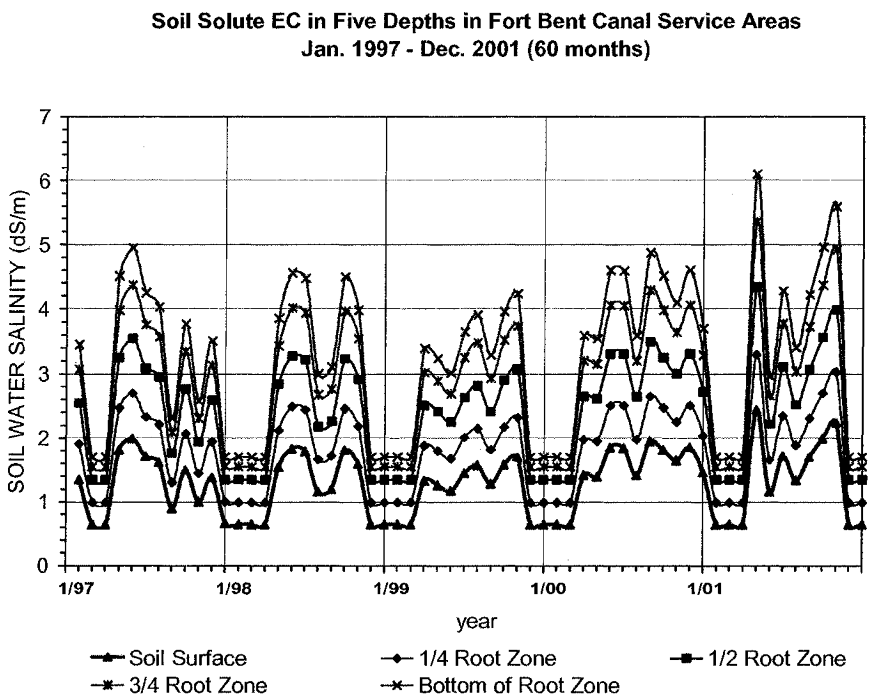


Figure 6.17: EC of Soil Water at Five Soil Depths in the Fort Bent Canal Service Area, 1997 to 2001 (60 months)

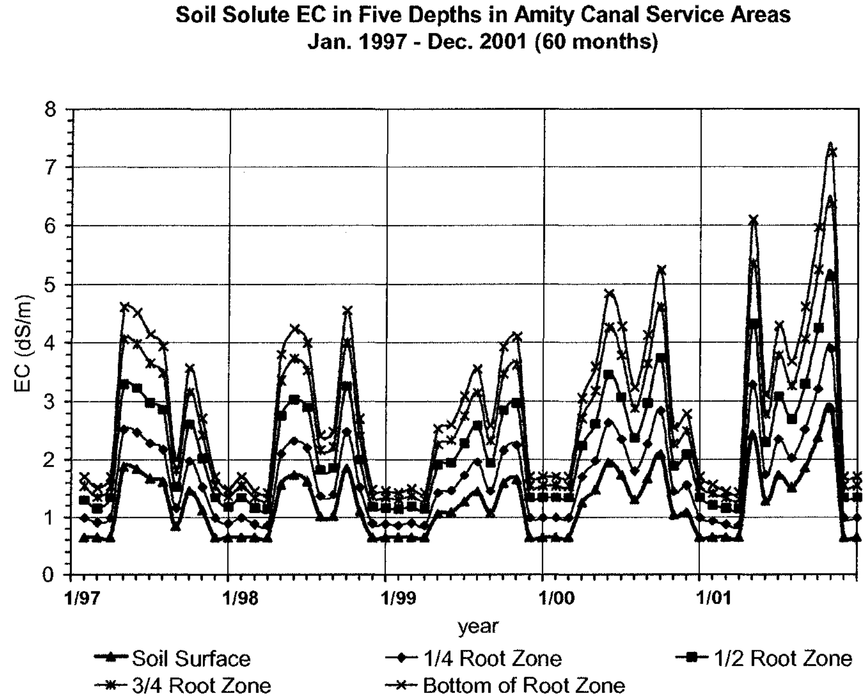


Figure 6.18: EC of Soil Water at Five Soil Depths in the Amity Canal Service Area, 1997 to 2001 (60 months)

Figures 6.14 to 6.18 show simulated soil water EC (dS/m) in the five canal service areas. As shown in the figures, soil water EC fluctuates from month to month. Downstream areas have higher EC than upstream areas. Deeper root zone layers have higher EC than the upper root zone. The major factors that affect soil water salinity include the amount of irrigation water, the amount of crop consumptive use, the quality of the river water, and the amount of rainfall and minerals in the soil. During the irrigation season, water is applied to the fields and infiltrates to become soil water. Crop consumptive use causes an increase in soil water salinity because plants use nearly pure water (Miles, 1977) leaving salts in the soil. The ratio of crop consumptive use to the amount of irrigation water determines the occurrence of chemical reactions in soil water hence determining soil water EC.

On average, downstream areas, such as service areas of the Fort Bent Canal and the Amity Canal (figure 6.17 and figure 6.18), have higher soil water EC than upstream areas such as the Bessemer Canal and the Colorado Canal (figure 6.14 and figure 6.15). This is because the river water is more saline in downstream reaches than upstream reaches as discussed in section 6.1.2. As shown in figures 6.16 to 6.18, soil water EC is usually higher in lower root zones than in upper root zones. This is because plant roots continue to uptake water as soil water filtrates down making soil water salinity higher in the deeper root zone.

Excess rainfall on the irrigated fields will dilute soil water and result in low soil water salinity. Using Fort Bent Canal (figure 6.17) as an example, there is excess rainfall in the amount of 19.1cm, 20.0cm and 15.5cm in August, 1997, July 1998 and May, 2001 respectively (average monthly rainfall is only 3.7 cm). It is evident that soil water is diluted by the low salinity rainfall and soil EC decrease significantly in these months.

Minerals in soil also influence soil EC. During the irrigation season, minerals can accumulate if the leaching fraction in soil is low. As irrigation seasons end, these minerals remains in the soil and will be dissolved by rain or snow in winter. This is why soil EC is high in some winter months as shown in figures 6.16 to 6.18.

6.1.4 Discussion

From January 1997 to December 2001, a total of 7,488,286 acre-ft of irrigation water was applied to the Arkansas River Basin. The irrigation water is derived from three sources: surface water, groundwater and precipitation. Their amounts and percentages are shown in table 6.1(a). After irrigation, these waters turn into canal transit loss, in-field deep percolation, crop / non-crop consumptive use, tail water and surface runoff. Their amounts and percentages are shown in table 6.1(b).

Table 6.1 Summary of Distribution of Irrigation Water in the Arkansas River Basin (1997 to 2001)

(a) Source of Irrigation Water

	Volume (acre-ft)	Percentage
Surface Water	*4,954,503*	*66.2%*
Groundwater	*524,797*	*7.0%*
Precipitation	*2,008,986*	*26.8%*
Total	***7,488,286***	*100%*

(b) Distribution of the Water after Irrigation

	Volume (acre-ft)	Percentage	Note
Canal Transit Loss	*1,591,444*	*21.3%*	Becomes Groundwater
In-field Deep Percolation	*1,664,637*	*22.2%*	Becomes Groundwater
Crop Consumptive Use	*2,514,074*	*33.6%*	Consumed
Tail Water and Runoff	*1,718,121*	*22.9%*	Becomes Surface Water
Total	***7,488,286***	*100%*	

As shown in table 6.1(a), 66.2% of the irrigation water comes from surface water, 7.0% is from groundwater, and 26.8% is from precipitation. The use of surface and groundwater is usually limited to the irrigation season. In contrast, precipitation occurs throughout the year. Of the total precipitation amount of 2,008,986 acre-ft, 27.0% of it occurred during the off-season, between October and March. Since there is limited farming activity during this time, precipitation cannot be utilized by crops.

As shown in table 6.1(b), 21.3% of the irrigation water turns into canal transit loss and 22.2% of it turns into in-field deep percolation. These two portions of water, percolate into the saturated aquifer. It is estimated that the salinity level of the water from canal transit loss is much lower than that of in-field deep percolation. Figures 6.14 to 6.18 show that as irrigation water infiltrates from the soil surface to the bottom of the root zone, its salinity level can increase to two or three times its original level. In other words, in-field deep percolation water has substantial influence on the deterioration of groundwater quality.

As shown in table 6.1(b), 33.6% of the irrigation water is consumptively used by crops and other plants. 22.9% of the irrigation water becomes tail water and runoff. Tail water and runoff returns to the river system via surface ditch systems and directly influence river quantity and quality.

The quantity and quality of water in the Arkansas River is influenced by groundwater and surface water that returns to the river. These return flows are dominated by canal transit losses, in-field deep percolation, and tail water created in the irrigation process. If irrigation practice or physical properties of the basin change, a change will occur in how these components affect river quantity and quality. This topic will be discussed in the next section on sensitivity analysis.

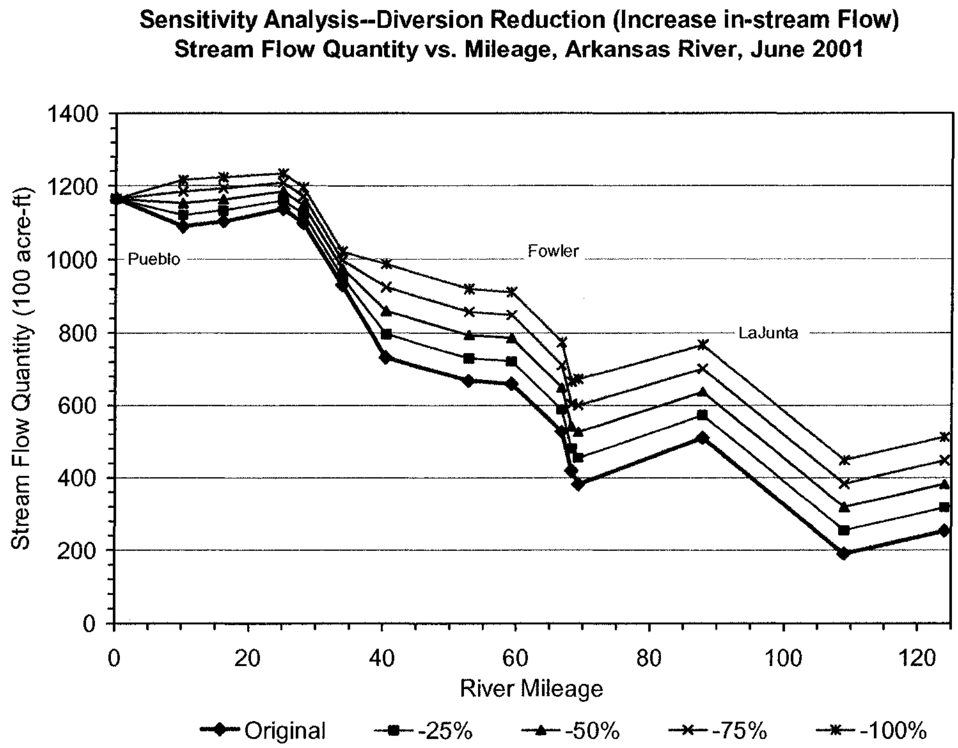
6.2 Sensitivity Analysis of Selected Parameters

It is important to address how parameters influence the simulation. The procedure is usually referred to as sensitivity analysis. In this section, two types of parameters are chosen for the sensitivity analyses: river diversion and tail water ratio. To analyze the sensitivity of these parameters the model is executed with these parameters at different settings. Then, the outcomes are compared with the outcomes obtained from the original parameter setting. River reaches which are located downstream of John Martin Reservoir are excluded for the sensitivity analysis. This is because different parameter settings will result in different amounts of inflow to John Martin Reservoir. Due to the change in reservoir volume, releases from the reservoir to downstream users will also change making the comparison between different outcomes very difficult.

Results of sensitivity analyses in June 2001 for the two selected parameters are shown in the following two sections. June 2001 was selected for analyses because June is a major month during the irrigation season and 2001 is the most recent year in the simulation period.

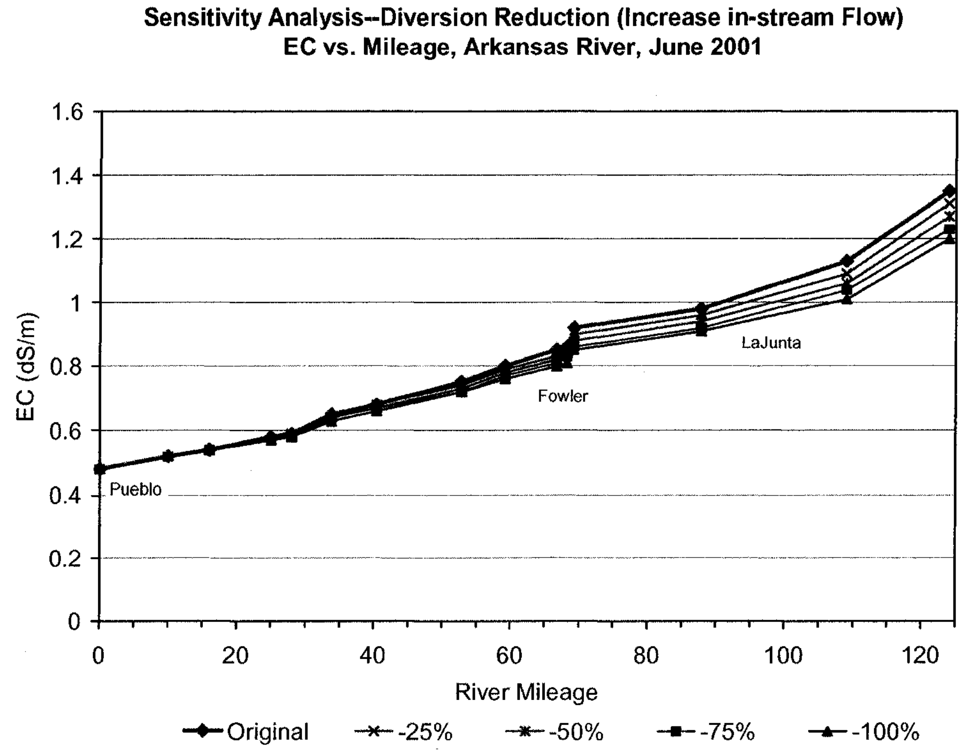
6.2.1 Sensitivity of River Diversion

The sensitivity analysis for river diversion is performed on three canals, Bessemer, Highline and Rocky Ford. The locations of the canals and their service areas are illustrated schematically in figures 5.2 and 5.7. Two scenarios are applied for the sensitivity analyses: (a) let the river diversion from these three canals be reduced to 25%, 50%, 75% and 100% of the original amount. Assuming that the amounts by which the diversions are reduced are released to the river; (b) let the river diversion from these three canals be reduced to 25% and 50% of the original amount. Assuming that the amounts by which the diversions are reduced (e.g. sold to cities) and diverted directly from Pueblo Reservoir. It is noted that in scenario (b), the reduction of river diversion is set to 25% and 50% instead of 25% to 100% as scenario (a). This is due to the regulations of the State of Colorado that when agricultural water is transferred for other uses, only the portion of the crop consumptive use can be transferred. The return flow must remain in the river. Therefore, the maximum reduction of river diversion in scenario (b) is set not to be above 50%. The simulation results of scenario (a) are shown in figures 6.19 and 6.20. The simulation results of scenario (b) are shown in figures 6.21 and 6.22.



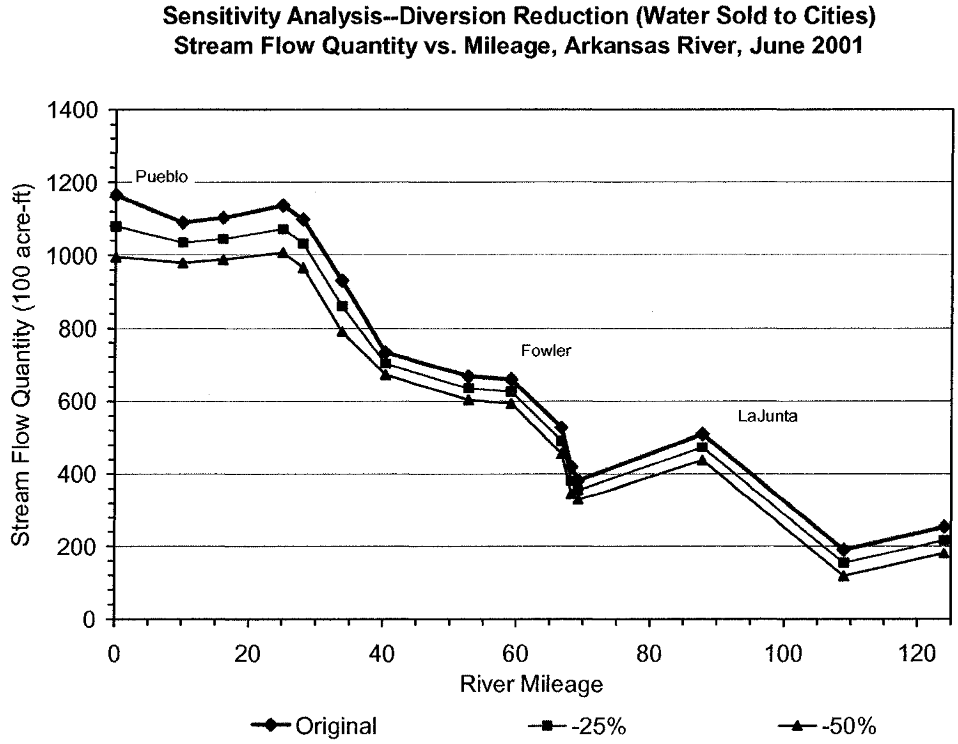
	0 mile	25 miles	59 miles	88 miles	124 miles
Original	***1164.70***	***1136.09***	***660.07***	***509.40***	***253.19***
-25%	***1164.70***	***1160.63***	***722.86***	***573.51***	***317.82***
-50%	***1164.70***	***1185.16***	***785.65***	***637.62***	***382.46***
-75%	***1164.70***	***1209.70***	***848.44***	***701.73***	***447.09***
-100%	***1164.70***	***1234.23***	***911.22***	***765.84***	***511.72***

Figure 6.19: Stream Flow Quantity vs. River Mileage under River Diversion Reduction (Increase in-stream Flow), June 2001



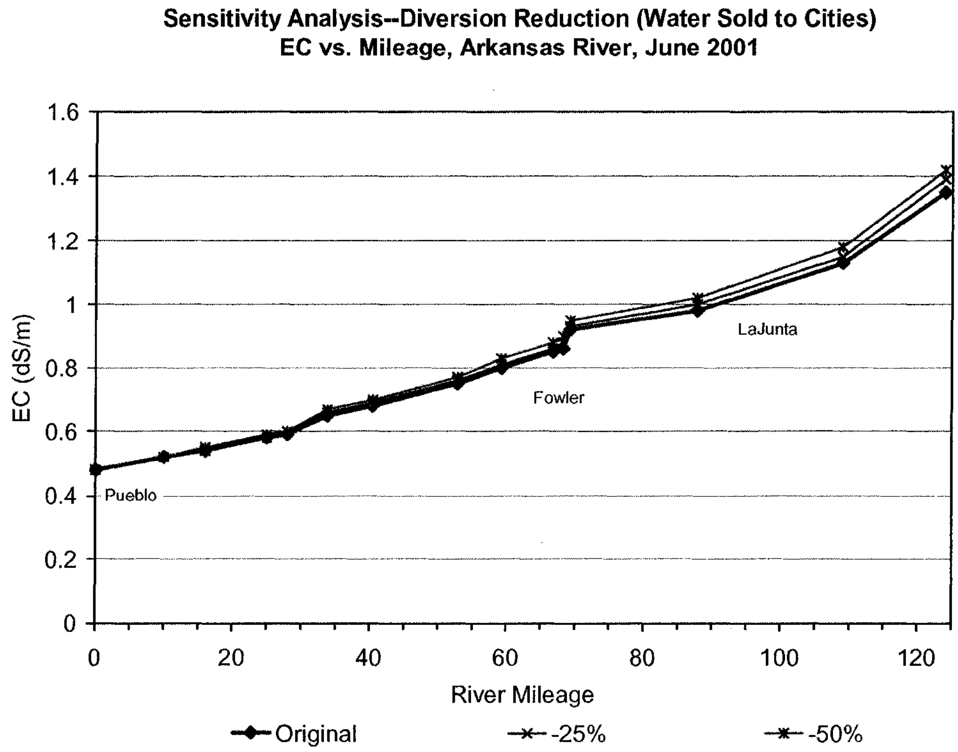
	0 mile	25 miles	59 miles	88 miles	124 miles
Original	***0.48***	***0.58***	***0.80***	***0.98***	***1.35***
-25%	***0.48***	***0.57***	***0.79***	***0.96***	***1.31***
-50%	***0.48***	***0.57***	***0.78***	***0.94***	***1.27***
-75%	***0.48***	***0.57***	***0.77***	***0.92***	***1.23***
-100%	***0.48***	***0.57***	***0.76***	***0.91***	***1.20***

Figure 6.20: EC vs. River Mileage under River Diversion Reduction (Increase in-stream Flow), June 2001



	0 mile	25 miles	59 miles	88 miles	124 miles
Original	***1164.70***	***1136.09***	***660.07***	***509.40***	***253.19***
-25%	***1079.64***	***1071.70***	***627.00***	***474.02***	***216.84***
-50%	***994.59***	***1007.32***	***593.93***	***438.63***	***180.48***

Figure 6.21: Stream Flow Quantity vs. River Mileage under River Diversion Reduction (e.g. Water Sold to Cities), June 2001



	0 mile	25 miles	59 miles	88 miles	124 miles
Original	***0.48***	***0.58***	***0.80***	***0.98***	***1.35***
-25%	***0.48***	***0.58***	***0.81***	***1.00***	***1.39***
-50%	***0.48***	***0.59***	***0.83***	***1.02***	***1.42***

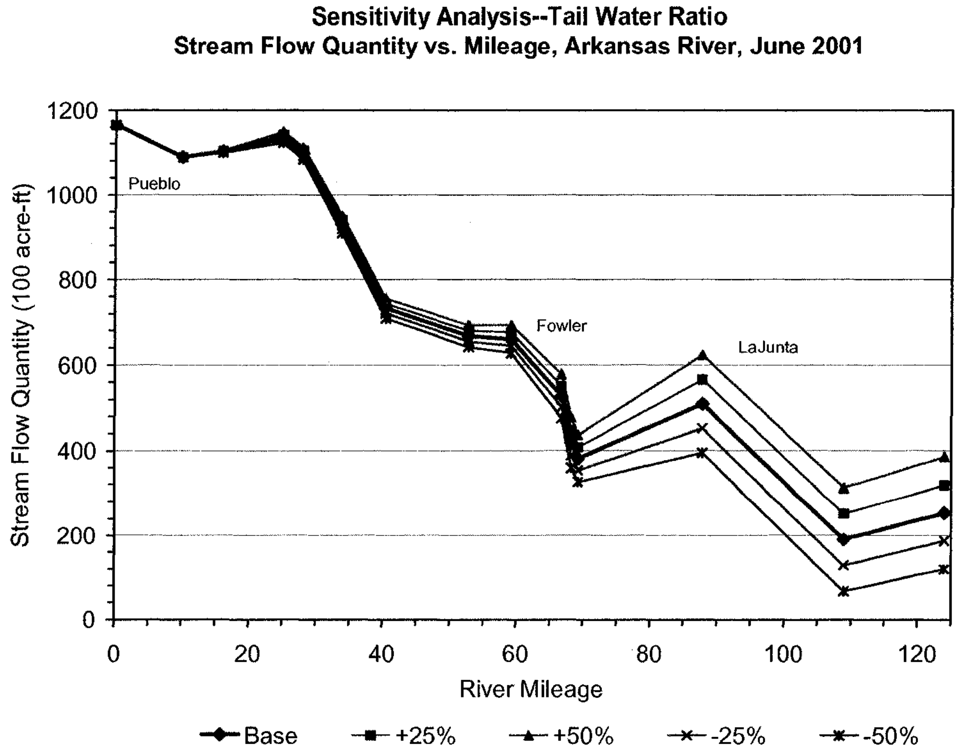
Figure 6.22: EC vs. River Mileage under River Diversion Reduction (e.g. Water Sold to Cities), June 2001

As shown in figure 6.19, reduction of river diversions results in increased river quantity if the amounts of the reduction are released to the river. The more the reduction is, the more the river quantity is increased. This is because crop consumptive use is reduced as canals divert less water for irrigation. As shown in figure 6.20, reduction of river diversions result in decreased river salinity in downstream river reaches. This is because the amount of irrigation return flow is reduced as agricultural water use is curtailed resulting in less salt loading to the river.

Figures 6.21 and 6.22 show the simulation results for the scenario where river diversions are reduced and the amount by which the diversions are reduced is diverted from Pueblo Reservoir (upstream reservoir). As shown in figures 6.21 and 6.22, the more the diversion is reduced, the smaller the flow in river and the higher the river salinity. Furthermore, as water is diverted from the upstream reservoir, it reduces the amount of low-salinity water which can be use for diluting downstream river reaches. Therefore, river salinity will increase as shown in figure 6.22.

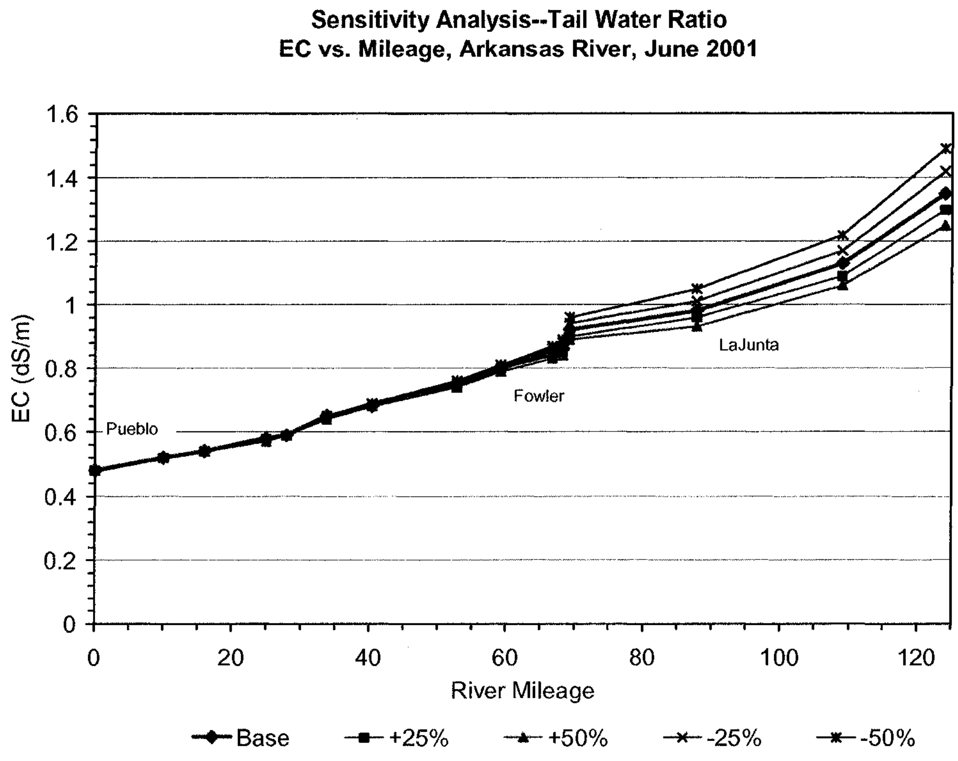
6.2.2 Sensitivity of Tail Water Ratio

As mentioned in equation 3.35, tail water ratio is a parameter indicating the portion of surface irrigation water which returns as surface return flow. The original setting for tail water ratio is 0.32. In this section, this parameter is changed to four settings for sensitivity analysis. The four settings are: a 25% increase, a 50% increase, a 25% decrease and a 50% decrease of the original value. The results are shown in figures 6.23 and 6.24.



	0 mile	25 miles	59 miles	88 miles	124 miles
Original	***1164.70***	***1136.09***	***660.07***	***509.40***	***253.19***
+25%	***1164.70***	***1142.663***	***676.55***	***566.38***	***319.45***
+50%	***1164.70***	***1149.22***	***692.95***	***623.18***	***385.51***
-25%	***1164.70***	***1129.53***	***643.55***	***452.36***	***186.87***
-50%	***1164.70***	***1122.96***	***627.02***	***395.33***	***120.55***

Figure 6.23: Stream Flow Quantity vs. River Mileage for Different Tail Water Ratio Settings, June 2001



	0 mile	25 miles	59 miles	88 miles	124 miles
Original	***0.48***	***0.58***	***0.80***	***0.98***	***1.35***
+25%	***0.48***	***0.58***	***0.80***	***0.96***	***1.30***
+50%	***0.48***	***0.57***	***0.79***	***0.93***	***1.25***
-25%	***0.48***	***0.58***	***0.81***	***1.01***	***1.42***
-50%	***0.48***	***0.58***	***0.81***	***1.05***	***1.49***

Figure 6.24: Stream Flow Quantity vs. River Mileage for Different Tail Water Ratio Settings, June 2001

As shown in figures 6.23 and 6.24, an increase in tail water ratio results in greater river quantity but decreases river EC. On the other hand, a decrease in tail water ratio results in decreased river quantity and increased river EC. This is because tail water ratio will influence the amount of surface return flow. Increased surface return flows will result in increased river quantity. Since surface return flows are generally low in salinity, increased surface return flows dilute the river and lower the river EC.

6.2.3 Discussion

Over-irrigation will cause excess salts to accumulate in the root zone and cause an increase in river salinity (Ghassemi et al., 1995). The sensitivity analyses for river diversions in section 6.2.1 indicate that reducing river diversions for agricultural irrigation will help alleviate the river salinity problem if the amounts by which the diversions are reduced are released to the river and not diverted downstream by other canals. This is because when less irrigation water is applied, less salt is loaded to the aquifer and returned to the river. However, if the amounts by which the diversions are reduced are removed from the river system, it will decrease river quantity and increase river salinity in downstream river reaches. This is because there is a smaller amount of low-salinity water available for diluting downstream river reaches.

Irrigation return flows in the Arkansas River Basin contain significant amounts of tail water (Cain, 1985). The sensitivity analysis for the tail water ratio in section 6.2.2 indicates that decreased tail water will result in decreased river quantity and increased river salinity and will negatively impact downstream water users.

CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS

7.1 Summary

In this study, a model was developed for simulating river quantity and quality in irrigated river basins. The model provides users with a comprehensive tool to analyze agricultural salinity problems. The development of the model consists of three steps. The first step is to simulate soil water composition. The second step is to simulate the spatial and temporal distribution of groundwater return flows. The third step is to integrate all hydrological components into the computation of river quantity and quality. A computer program with a graphical user interface was designed to facilitate the computation and to display the results.

The simulation of soil water composition is based on simulating various chemical reactions including precipitation and dissolution of calcite ($CaCO_3$), magnesite ($MgCO_3$) and gypsum ($CaSO_4$), association and dissociation of major species of ion pairs and dissolution of carbon dioxide (CO_2) in the soil. Soil water composition is simulated at four soil depths: the soil surface, ¼, ½, ¾ and the bottom of root zone soil. By simulating soil water composition, quantity and quality of the deep percolating water can be obtained.

Irrigation results in deep percolation which becomes a source of groundwater return flow. The spatial and temporal distribution of groundwater return flows to the river system is expressed as response functions multiplied by the volume of the stress applied to the basin. The response functions are derived based on two-dimensional porous media flow equations. To solve the flow equations numerically, the aquifer is discretized into rectangular cells. A probabilistic approach is developed to simulate the travel time of groundwater flows under irrigated fields that have subsurface drainage devices.

All related hydrological components are integrated into the computation of river quantity and quality. These components include: groundwater return flow, irrigation tail water, tributary inflow, river diversions, phreatophyte consumption, river channel losses and river depletion due to pumping. The river system is divided into reaches and the river quantity and quality is simulated from upstream reaches to downstream reaches sequentially.

The Arkansas River Basin in Colorado was selected as the study area for implementing the hydro-salinity simulation. The simulation period is from January 1986 to December 2001, a total of 192 months. The first 60 months (1986 to 1990) are used to calibrate the model. Another 132 months (1991 to 2001) are used to validate the calibrated model. A GIS program, ArcView 3.2, was applied to generate a groundwater table surface and develop response functions for groundwater return flows in each canal service area. By comparing the simulated river EC (dS/m) and river quantity (acre-ft) with observed data at five different river locations, the model was shown to be capable of simulating both river quantity and quality in the Arkansas River from 1986 to 2001. The simulation results show that the irrigation return flow (including surface and groundwater return flow) cause significant increases of river quantity in reaches 12, 13, 14, 21, 22, 23 and 24. River EC is primarily determined by the percentage of groundwater return flow in river. Both river quantity and river EC show significant seasonal fluctuation. The simulation results also indicate that soil water composition in irrigated areas is primarily affected by: (a) the ratio of crop consumptive use to the amount of irrigation water and (b) the amount of soluble minerals in the soil. The simulation from 1997 to 2001 indicates that of the estimated 7,488,286 acre-ft of irrigation water applied to the region, 21.3%, 22.2% and 22.9 % turns into canal transit loss, in-field deep percolation and tail water respectively. The former two (a sum of 43.5%) become groundwater return flow while tail water becomes surface return flow to the river.

Parameter sensitivity analyses for the Arkansas River Basin indicates that decreasing agricultural water use can alleviate river salinization if the amount by which the agricultural water is decreased is released to the river. The sensitivity analyses also indicates that reducing irrigation tail water will cause an increase in river salinity.

7.2 Major Contributions

A steady state soil water composition simulation program WATSUIT (Rhoades, 1990) was modified to be able to handle dynamic state simulation. The modification was accomplished by dividing the simulation period into multiple time steps and executing the model sequentially from the first time step to the last time step. The amounts of minerals including calcite, magnesite and gypsum that precipitate in a month are considered as the initial mineral condition in the soil for the soil water composition simulation in the next month.

A groundwater model was designed specifically for generating response functions due to deep percolating irrigation water. This model contains essential elements of simulating groundwater flow paths and groundwater travel time with which the spatial and temporal distribution of irrigation return flow can be obtained. With this model, response functions can be generated based on the characteristics of the aquifer.

The integration of the soil water simulation model, the groundwater return flow model and the river quantity and quality simulation with a user-friendly graphical interface has several advantages. It facilitates data input and parameter editing. It reduces model execution time and enhances execution reliability, and it provides for convenient review of simulation results by comparing river quantity and quality as well as soil water salinity in different river reaches.

7.3 Conclusions

According to the results of the simulation for the Arkansas River Basin from 1986 to 2001, some river reaches receive significantly more irrigation return flow than others. For the upstream segment of the river from Pueblo to John Martin Reservoir, river reaches 12, 13 and 14 (from river mile 69.2 to 124.0) receive significant amounts of groundwater and tail water generated from the service areas of the Colorado, Highline, Catlin, Rocky Ford, Fort Lyon, and Consolidated canals. For the downstream river segment from John Martin Reservoir to the Kansas state line, river reaches 21, 22, 23 and 24 (from river mileage 167.4 to 209.2) receive significant amounts of groundwater and tail water from the Ft Lyon Canal, Amity Canal, Lamar Canal and XY Graham Canal service areas.

In the Arkansas River, irrigation tail water is a major factor in the difference between river salinity during the irrigation season and the rest of the year. During the non-irrigation season, the river is primarily fed by saline groundwater. During the irrigation season, about 32% of the irrigation water will turn into tail water. Since the tail water contains less salt compared to the groundwater, it lowers river salinity as it returns to the river system. Therefore, tail water plays an important role in maintaining river quality.

The simulation of soil water composition in the Arkansas River has shown the significance of soil water salinity changes. Depending on the amount of crop consumptive use, different leaching fractions can result, the salinity of the irrigation water can increase several folds in the process of infiltrating from the soil surface to the bottom of the root zone. As a result, the soil water brings salt into the aquifer and this has a major influence on the quality of groundwater and the quality of the river.

7.4 Recommendations

Recommended future research work is listed below:

1. Additional research on river channel losses in the Arkansas River. The research can be focused on (a) surveys of the river bed conductance, or (b) the relationship between the flow volume in the channel and the corresponding leakage. In this study, the river channel leakage is expressed as river reach volume multiplied by a leakage percentage. With additional information concerning river bed conductance or the relationship between flow volume and leakage, the settings of the leakage percentage for each month in each river reach could be refined. As a result, the simulation of river quantity might be improved.
2. Update of the drainage portion settings in the aquifer grid. "Drainage portion" is a parameter created in this study to account for the irrigated fields that have

subsurface drainage devices. The parameter is set uniformly at 0.25 for the entire Arkansas River basin. It is recommended that an additional survey for the distribution of subsurface drainage systems in the basin be conducted. Then, the settings of the drainage portion can be updated to improve the simulation.

3. Additional experiments to determine the dissolution of un-weathered minerals in soil should be conducted. As irrigation water contacts the soil, it dissolves the un-weathered minerals in the soil. The amount of dissolution is dependent on the geo-chemical characteristics of the soil as well as the quality of the soil water. Field experiments can help determine the amount of the dissolution of the un-weathered minerals in the soil.
4. Evaluating long term effects of reducing agricultural water use. Due to the increasing demand for water in urban areas, there has been a tendency to transfer agricultural water to municipal use in the Arkansas River Basin. As less agricultural water is available, less deep percolation is recharged to the aquifer. In the long term, this will result in a decrease of groundwater base flow to the river system. It is recommended that the base flow settings used in this study be re-evaluated if this trend continues. The model should be executed over multiple years with reduced base flows to evaluate the long term effect of reducing agricultural water use on river quantity and quality.

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APPENDICES

Appendix A: Data of Observed and Simulated Monthly Stream Flow Quantity in the Arkansas River from January 1986 to December 2001

units:100 acre-ft

yr/mon	Avondale			Fowler			Lamar			Coolidge		
	Obs	sim	error	obs	sim	error	obs	sim	error	obs	sim	error
1986/01	363	374.8	11.76	500.9	368.3	-133	23.09	2.76	-20.3	100.3	74.11	-26.2
1986/02	334	372.5	38.5	307.9	361.4	53.54	13.95	2.76	-11.2	86.57	57.63	-28.9
1986/03	374.8	421.1	46.28	155.7	288.8	133.1	129.2	101.7	-27.5	155.5	86.28	-69.2
1986/04	487	527.6	40.68	121.6	283.7	162.1	280.2	251.7	-28.5	302.5	244.9	-57.6
1986/05	703.8	758.9	55.04	224.8	432.1	207.3	122.1	208.7	86.6	210.5	214.5	4.02
1986/06	2398	2281	-117	1748	1454	-294	119.2	130.9	11.71	203.7	258.2	54.5
1986/07	1637	1449	-188	1144	793.7	-351	236.2	234.4	-1.8	315.7	278.6	-37
1986/08	686.3	702	15.7	243.4	338.2	94.84	111.4	74.49	-36.9	359.6	258.4	-101
1986/09	388.1	380.5	-7.61	195	126.5	-68.5	25.26	2.76	-22.5	177.6	70.85	-107
1986/10	394.7	371.4	-23.2	280	186.4	-93.6	28.48	7.64	-20.8	156	114.5	-41.5
1986/11	363.6	349.9	-13.7	300.1	209.3	-90.8	31.08	2.76	-28.3	148.2	91.05	-57.2
1986/12	441.3	387.1	-54.2	475.3	296.6	-179	22.1	2.76	-19.3	119.5	81.54	-37.9
1987/01	316.8	352.6	35.71	399.1	370.7	-28.3	19.36	8.09	-11.3	109.2	83.53	-25.7
1987/02	391.8	406.1	14.31	431.3	373.5	-57.8	18.59	37.05	18.46	102.2	104.1	1.88
1987/03	529.8	515.6	-14.1	400.8	335.8	-65	127.9	96.09	-31.8	164.7	157.7	-7.07
1987/04	1121	1033	-87.9	781.9	615.1	-167	648	537	-111	726.6	480	-247
1987/05	2113	1979	-134	2268	2114	-154	1318	1223	-94.5	1317	1116	-201
1987/06	1953	2080	126.8	1481	1799	317.9	1242	1290	47.63	1477	1372	-105
1987/07	795.7	963.4	167.7	252.3	691.6	439.3	226.2	336.3	110.1	428.5	551.8	123.3
1987/08	623.7	676.7	53.03	195.6	312.7	117.1	168.6	269.2	100.6	294	404.2	110.3
1987/09	371.8	344.2	-27.6	140.3	111.3	-28.9	39.83	80.05	40.22	161.6	169.6	8.04
1987/10	285	294	9.05	127.7	146	18.26	70.65	78.69	8.04	175.9	136.8	-39.1
1987/11	263.8	269.4	5.59	229.2	192.2	-37	18.84	25.42	6.58	152.4	131	-21.4
1987/12	239.6	240.1	0.51	252.7	230.4	-22.3	26.72	33.14	6.42	133.5	124.4	-9.1
1988/01	385.1	336	-49.1	353.7	324.2	-29.5	21.84	20.83	-1.01	117	113.4	-3.53
1988/02	310.2	306.9	-3.23	301.2	304.2	2.99	21.42	2.76	-18.7	118.8	80.76	-38.1
1988/03	310.7	368.3	57.57	144.3	272.1	127.8	28.13	37.61	9.48	119.8	103.8	-16
1988/04	362.1	416.6	54.51	226.5	268.5	41.92	162.4	197.6	35.21	249.5	249.6	0.14
1988/05	529.1	572.9	43.87	167.8	286.4	118.6	57.28	52.45	-4.83	155.4	156	0.52
1988/06	944.6	917.5	-27.1	356.4	446	89.57	158.5	157.6	-0.88	223.9	198.4	-25.5
1988/07	851.6	869.9	18.28	430.1	463.6	33.43	324.8	299	-25.8	386.5	257.8	-129
1988/08	688.9	629.9	-59	266.3	191.5	-74.9	280.9	240.3	-40.6	385.1	259.4	-126
1988/09	260.6	262.6	1.98	78.4	75.39	-3.01	33.01	8.31	-24.7	157.9	120.4	-37.5
1988/10	221.2	240	18.8	59.65	100.2	40.59	7.81	9.81	2	122.8	97.15	-25.6
1988/11	194.6	206.2	11.61	127.1	139.7	12.53	13.92	4.53	-9.39	109.5	76.86	-32.7
1988/12	165.2	161.6	-3.64	193.3	159.5	-33.8	18.05	2.76	-15.3	122.4	79.1	-43.3
1989/01	177.2	174.9	-2.29	225.2	220.3	-4.92	17.18	2.76	-14.4	103.9	75.9	-28
1989/02	271.2	234.3	-37	243.5	195	-48.5	13.69	2.76	-10.9	89.05	55.52	-33.5
1989/03	449.1	374.7	-74.4	195.3	101.3	-94	12.62	21.19	8.57	99.5	81.34	-18.2
1989/04	499.5	485.6	-13.9	111.3	121.7	10.38	21.1	35.18	14.08	49.84	121.8	71.95
1989/05	442.1	523.9	81.8	205.4	218.5	13.15	118	79.44	-38.6	190.7	158.7	-32
1989/06	726.7	805	78.23	391.9	466.4	74.56	25.59	2.76	-22.8	89.05	56.21	-32.8
1989/07	1112	1079	-32.9	580.8	568.2	-12.7	184.8	94.8	-90	173.4	75.99	-97.4

1989/08	743.1	685.5	-57.6	390	281.3	-109	128.9	64.24	-64.7	210.8	142.3	-68.6
1989/09	219.7	241	21.32	47.29	75.35	28.06	12.49	2.76	-9.73	58.44	34.28	-24.2
1989/10	203.5	226.5	23.07	67.02	101	33.98	55.24	30.69	-24.6	98.9	61.82	-37.1
1989/11	196.9	203.5	6.61	108.8	112.4	3.61	13.57	2.76	-10.8	73.85	45.26	-28.6
1989/12	152.8	153.1	0.39	148.2	151.1	2.89	15.85	2.76	-13.1	78.32	45.95	-32.4
1990/01	169.5	160.6	-8.95	107.6	210.7	103.1	12.14	5.23	-6.91	89.25	52.64	-36.6
1990/02	230.7	207	-23.7	172.8	175.3	2.53	11.72	2.76	-8.96	72.15	54.54	-17.6
1990/03	290.9	263.4	-27.6	196.1	99.76	-96.3	11.75	2.76	-8.99	83.73	65	-18.7
1990/04	304.2	297.5	-6.74	78.17	82.06	3.89	14.22	39.83	25.61	59.14	131.5	72.32
1990/05	634.4	568.6	-65.8	338.1	234.8	-103	11.16	2.76	-8.4	47.85	91.72	43.87
1990/06	1254	1167	-86.5	757	615	-142	164	100.7	-63.3	100.4	155.5	55.07
1990/07	931	940.1	9.18	710	599.3	-111	88.19	27.92	-60.3	136.4	90.11	-46.3
1990/08	507.7	529.2	21.5	296.2	228.6	-67.6	14.42	2.76	-11.7	38.25	43.64	5.39
1990/09	313.9	301.9	-12	99.22	97.4	-1.82	3.04	2.76	-0.28	30.92	30.92	0
1990/10	392.9	343	-49.9	247.4	152.4	-95.1	2.76	2.76	0	42.08	49.07	6.99
1990/11	345.3	314.5	-30.8	192.5	211.7	19.23	17.79	24.96	7.17	64	82.53	18.53
1990/12	177.5	217.9	40.4	81.98	222.5	140.5	14.6	15.55	0.95	71.99	74.64	2.65
1991/01	305.3	270	-35.3	266.4	296.8	30.35	13.77	5.05	-8.72	82.07	76.3	-5.77
1991/02	213.6	245.5	31.85	218	240.3	22.34	9.51	2.76	-6.75	71.26	52.63	-18.6
1991/03	255.1	286.3	31.22	73.63	126.8	53.21	3.64	2.76	-0.88	60.53	79.54	19.01
1991/04	247	280.1	33.1	66.46	92.62	26.16	22.71	43.42	20.71	41.27	139.4	98.08
1991/05	614	586.9	-27.1	195.4	214.5	19.18	4.34	2.76	-1.58	30.95	82.39	51.44
1991/06	1281	1168	-113	721.1	619.1	-102	114.4	76.12	-38.2	82.09	155.4	73.34
1991/07	1060	972.8	-87	612.5	583.9	-28.7	145.1	62.64	-82.5	157.9	110.9	-47
1991/08	774.7	687.7	-87	404.8	294.7	-110	11.18	46.78	35.6	58.01	181.5	123.5
1991/09	246.5	248.9	2.43	89.9	74.7	-15.2	10.11	2.76	-7.35	19.21	30.92	11.71
1991/10	209.1	226.3	17.18	57.99	109.6	51.58	7.25	2.76	-4.49	27.07	55.76	28.69
1991/11	294.8	273.4	-21.3	173.3	165.9	-7.42	17.65	2.76	-14.9	54.93	86.9	31.97
1991/12	194.9	194	-0.96	139.7	194.1	54.34	15.83	2.76	-13.1	66.16	71.79	5.63
1992/01	224.2	234	9.75	257.8	265	7.27	11.84	2.76	-9.08	64.89	65.68	0.79
1992/02	240.4	249.5	9.14	274.1	225.6	-48.6	11.42	2.76	-8.66	58.3	53.52	-4.78
1992/03	366.1	346.8	-19.3	171.6	145.3	-26.3	8.06	2.76	-5.3	54.52	71.44	16.92
1992/04	500.6	478.3	-22.3	199.9	200.8	0.91	18.12	57.65	39.53	36.53	123.1	86.57
1992/05	766.3	722	-44.3	407.6	389.2	-18.5	26.18	27.81	1.63	37.96	90.76	52.8
1992/06	976.9	1031	53.73	719.5	712.1	-7.32	15.52	2.76	-12.8	50.57	78.85	28.28
1992/07	743.5	707.1	-36.4	380.7	377.2	-3.53	169.5	82.66	-86.8	210.6	158.4	-52.1
1992/08	695.3	648	-47.3	323.1	248.8	-74.3	9.41	10.28	0.87	101.6	108.1	6.49
1992/09	463.2	363.7	-99.5	232.8	132.3	-101	5.27	2.76	-2.51	46.93	30.92	-16
1992/10	230.3	249	18.72	76.54	122.6	46.02	3.57	9.06	5.49	34.5	45.4	10.9
1992/11	319.4	274.9	-44.6	246.9	172	-74.9	3.47	12.03	8.56	47.71	80.22	32.51
1992/12	200.6	185.3	-15.2	221	193.8	-27.2	14.6	2.76	-11.8	67.69	65.38	-2.31
1993/01	269.4	210.9	-58.5	168.5	260.7	92.21	10.85	2.76	-8.09	72.56	76.42	3.86
1993/02	240.8	220	-20.8	151	216.1	65.17	11.64	2.76	-8.88	82.96	72.31	-10.7
1993/03	275.6	262.5	-13.1	164.5	136.7	-27.8	16.21	2.76	-13.5	103.8	98.37	-5.38
1993/04	478	431.4	-46.7	148.1	139.1	-9.01	9.21	67.93	58.72	83.08	184.3	101.2
1993/05	1080	1012	-67.2	607.8	550.1	-57.6	9.41	14.15	4.74	76.57	132.3	55.7
1993/06	1737	1600	-136	977.5	957.6	-20	88.71	40.23	-48.5	124.1	147.9	23.82
1993/07	1309	1223	-85.3	733.7	749.3	15.57	246.8	155.6	-91.3	272.9	169.7	-103
1993/08	835.9	803.2	-32.6	423.1	416.4	-6.69	24.1	12.04	-12.1	78.46	130.4	51.9
1993/09	367.2	369	1.82	154.7	158.1	3.44	7.77	2.76	-5.01	60.37	30.92	-29.5

1993/10	292.8	302.1	*9.26*	142.3	173.5	*31.24*	7.71	2.76	*-4.95*	80.96	79.52	*-1.44*
1993/11	264.5	263.9	*-0.69*	259.7	209.4	*-50.4*	18.12	20.79	*2.67*	88.63	79.9	*-8.73*
1993/12	204.1	197.4	*-6.64*	257.6	218.8	*-38.8*	15.77	5.74	*-10*	96.01	72.67	*-23.3*
1994/01	194.3	210.5	*16.11*	158.4	275.6	*117.1*	15	6.3	*-8.7*	94.05	77.46	*-16.6*
1994/02	197.4	207	*9.61*	112.8	216	*103.1*	12.08	2.76	*-9.32*	78.38	51.18	*-27.2*
1994/03	289.9	271	*-18.9*	104.2	120.1	*15.94*	5.67	2.76	*-2.91*	58.76	60.67	*1.91*
1994/04	600.6	559.5	*-41.1*	270.6	211.5	*-59.1*	14.95	63.98	*49.03*	75.3	168.4	*93.08*
1994/05	1527	1469	*-58.3*	1073	833	*-240*	10.67	20.9	*10.23*	55.53	76.02	*20.49*
1994/06	2005	1980	*-24.9*	1438	1358	*-79.3*	167.5	163.1	*-4.39*	155.3	181	*25.69*
1994/07	753.8	867.9	*114.1*	238	600.3	*362.3*	227.5	124.7	*-103*	267.7	156.2	*-112*
1994/08	559.9	550.3	*-9.61*	209.1	280.1	*70.97*	32.83	2.76	*-30.1*	129.8	87.12	*-42.7*
1994/09	413.6	344.5	*-69.1*	170.9	119.1	*-51.8*	12.38	2.76	*-9.62*	67.6	30.92	*-36.7*
1994/10	419.2	354.5	*-64.6*	231.7	178.6	*-53.1*	13.27	2.76	*-10.5*	70.5	63.12	*-7.38*
1994/11	296.6	289.8	*-6.8*	226.9	213.2	*-13.7*	15.7	30.72	*15.02*	76.43	79.37	*2.94*
1994/12	329.6	260.4	*-69.2*	205.7	243.6	*37.91*	22.51	14.94	*-7.57*	102.6	77.57	*-25*
1995/01	185.5	218.4	*32.91*	198.2	296.4	*98.24*	14.85	9.27	*-5.58*	108.5	83.86	*-24.7*
1995/02	177.8	222.2	*44.33*	100.1	247.8	*147.7*	10.75	2.76	*-7.99*	88.78	66	*-22.8*
1995/03	307.5	280.9	*-26.6*	137.9	141.9	*4.03*	10.42	15.98	*5.56*	82.53	92.6	*10.07*
1995/04	532.3	451.2	*-81.1*	279.2	192.4	*-86.8*	3.51	48.94	*45.43*	71.59	175.9	*104.3*
1995/05	1569	1439	*-130*	1070	810.4	*-260*	36.18	6.45	*-29.7*	135.2	144.9	*9.69*
1995/06	2923	3041	*117.7*	2495	2378	*-117*	131	2.76	*-128*	181.2	102.5	*-78.7*
1995/07	2725	2757	*32.84*	2332	2238	*-93.6*	1510	1216	*-295*	1386	1097	*-289*
1995/08	1150	1228	*77.79*	690.6	753.2	*62.6*	145.6	148.9	*3.37*	267	315.6	*48.59*
1995/09	597.4	592.9	*-4.5*	383	378	*-5.02*	171.2	156.8	*-14.4*	224.7	281	*56.3*
1995/10	444.1	417.8	*-26.3*	276.1	292.6	*16.49*	15.88	48.34	*32.46*	137.7	148	*10.36*
1995/11	338.1	334.6	*-3.52*	323.9	294.6	*-29.3*	30.38	51.37	*20.99*	115.7	135.4	*19.76*
1995/12	253.5	250.6	*-2.92*	170.4	285.7	*115.3*	14.41	26.83	*12.42*	117.5	119	*1.5*
1996/01	243.5	236.9	*-6.56*	244.1	330.9	*86.82*	9.11	15.91	*6.8*	120.4	108	*-12.4*
1996/02	333	313.1	*-20*	282.8	295.1	*12.32*	9.13	3.94	*-5.19*	101.5	82.41	*-19.1*
1996/03	493.4	448.7	*-44.7*	185.2	207.3	*22.1*	43.63	48.41	*4.78*	121.4	122.1	*0.7*
1996/04	555	587.6	*32.52*	169.4	272.9	*103.5*	182.5	196.6	*14.12*	268.5	281.4	*12.93*
1996/05	1186	1162	*-23.9*	553.9	520.7	*-33.2*	313	206.7	*-106*	426	368.2	*-57.8*
1996/06	1500	1615	*115*	778.9	889.9	*111*	116.7	76.54	*-40.1*	296	294.3	*-1.69*
1996/07	1088	953.3	*-134*	697.9	526.4	*-172*	220.2	152	*-68.2*	398.7	326.9	*-71.8*
1996/08	785.9	671.4	*-115*	464.9	299.5	*-165*	50.15	4.89	*-45.3*	313.2	214.9	*-98.3*
1996/09	296.6	246.6	*-50.1*	134	70.79	*-63.2*	36.73	2.76	*-34*	230.5	107.7	*-123*
1996/10	265.9	217.7	*-48.2*	100.9	96.89	*-4.01*	13.25	2.76	*-10.5*	149.7	90.63	*-59.1*
1996/11	296.3	245.1	*-51.1*	199.3	132.8	*-66.5*	32.32	2.76	*-29.6*	138.2	77.28	*-60.9*
1996/12	192	173.9	*-18.1*	249.2	172.7	*-76.5*	34.86	2.76	*-32.1*	139.3	73.75	*-65.5*
1997/01	228	220	*-8.02*	263.6	242.3	*-21.3*	37.6	2.76	*-34.8*	129.6	69.25	*-60.4*
1997/02	222.5	225.9	*3.43*	175.8	207.5	*31.69*	29.55	2.76	*-26.8*	113	62.05	*-50.9*
1997/03	506.2	445.3	*-60.9*	294.7	186.5	*-108*	87.45	80.84	*-6.61*	177.3	110.8	*-66.5*
1997/04	755.1	689.3	*-65.9*	400.6	303.2	*-97.4*	105.4	135.9	*30.52*	242.7	250.7	*7.94*
1997/05	1094	1012	*-82.7*	579.8	526.5	*-53.3*	45.87	62.37	*16.5*	155.3	209.1	*53.77*
1997/06	2957	2826	*-131*	2326	2131	*-195*	58.54	94.1	*35.56*	170.6	218.4	*47.71*
1997/07	1056	1065	*9.13*	460.1	676.1	*216*	419.3	330.8	*-88.4*	457.9	378.6	*-79.3*
1997/08	997.6	781.7	*-216*	734.1	490.7	*-243*	232.4	123.6	*-109*	620.6	349.9	*-271*
1997/09	478.6	395	*-83.6*	220.4	170.2	*-50.2*	38.55	2.76	*-35.8*	204.2	140.9	*-63.3*
1997/10	432.8	351	*-81.9*	236.6	165.6	*-71*	36.45	37.03	*0.58*	204.3	191.8	*-12.5*
1997/11	483	376.1	*-107*	502.8	252.7	*-250*	69.56	18.45	*-51.1*	252.2	116.7	*-135*

1997/12	365.5	312.3	*-53.3*	465.7	284.9	*-181*	214.9	141.2	*-73.8*	328.5	180.7	*-148*
1998/01	208.1	246.7	*38.62*	347.1	342.5	*-4.6*	489.6	268.1	*-222*	597.4	216	*-381*
1998/02	216.5	238.1	*21.54*	265.8	302.6	*36.78*	192.8	192.9	*0.12*	316.6	229	*-87.6*
1998/03	497.7	397	*-101*	520.6	294.2	*-226*	317.2	323.3	*6.1*	404.3	381.4	*-22.9*
1998/04	962.6	834.1	*-128*	751.7	493.6	*-258*	370.9	428.9	*58.07*	552.9	594.3	*41.36*
1998/05	1027	998.9	*-28.3*	782.4	725.9	*-56.5*	77.36	197.2	*119.8*	225.7	477.5	*251.8*
1998/06	972	1090	*118.1*	518.1	849.2	*331.1*	327.7	350.1	*22.39*	384.1	515.2	*131.1*
1998/07	1033	997.1	*-35.9*	577.3	598.8	*21.54*	325.7	299.7	*-26*	454.3	514.7	*60.37*
1998/08	743.7	570.6	*-173*	586.5	145.8	*-441*	43.51	65.17	*21.66*	312.5	248	*-64.5*
1998/09	461.8	394.1	*-67.7*	197.6	130.6	*-67.1*	28.26	15.38	*-12.9*	142.4	116	*-26.4*
1998/10	299.1	253.7	*-45.4*	232.6	110	*-123*	17.99	42.86	*24.87*	185.1	166.1	*-19*
1998/11	327.3	261.2	*-66.1*	261	195.7	*-65.3*	48.44	31.3	*-17.1*	188.5	127.8	*-60.8*
1998/12	228.8	187	*-41.9*	257.8	201.1	*-56.7*	29.98	7.97	*-22*	138.5	111.1	*-27.3*
1999/01	224.2	206.1	*-18.1*	249.9	271	*21.14*	25.98	16.19	*-9.79*	138.6	114.2	*-24.5*
1999/02	193.9	200.5	*6.62*	177.4	216.6	*39.19*	27.03	5.07	*-22*	117.9	80.78	*-37.1*
1999/03	333.1	292.1	*-41*	250.2	111.3	*-139*	13.86	27.51	*13.65*	128.2	133.7	*5.52*
1999/04	724.1	631.2	*-92.9*	484.4	273.8	*-211*	27.72	107	*79.29*	162.8	256.9	*94.16*
1999/05	1793	1596	*-197*	2266	1945	*-321*	1284	1130	*-155*	1523	1063	*-460*
1999/06	1837	1857	*19.45*	1630	1631	*0.55*	1112	1046	*-66.5*	1369	1156	*-213*
1999/07	1052	991.2	*-60.3*	800.4	714.2	*-86.2*	322	394.4	*72.41*	433.3	652.6	*219.3*
1999/08	1430	1299	*-131*	1250	1181	*-68.5*	473	426.6	*-46.5*	687.9	627	*-60.9*
1999/09	550.4	528	*-22.4*	358.1	290.4	*-67.7*	31.81	40.1	*8.29*	204.1	199	*-5.08*
1999/10	405	383.9	*-21.1*	320.1	250.6	*-69.6*	20.48	31.39	*10.91*	174.3	180.2	*5.883*
1999/11	370.9	348.8	*-22.2*	376.7	300.8	*-75.8*	19.83	66.51	*46.68*	174.9	134.5	*-40.5*
1999/12	369.3	333.6	*-35.7*	478.4	313.6	*-165*	32.24	21.03	*-11.2*	151.1	119.6	*-31.6*
2000/01	363	352.8	*-10.2*	457.6	386.7	*-70.9*	31	44.91	*13.91*	147	123.7	*-23.3*
2000/02	260.6	289.3	*28.7*	236.2	336.7	*100.5*	151.1	89	*-62.1*	236.8	119.3	*-117*
2000/03	460.6	427.6	*-32.9*	419.5	289	*-131*	58.31	75	*16.69*	258.2	186.4	*-71.9*
2000/04	631.9	570.2	*-61.8*	551	290	*-261*	41.06	192.4	*151.3*	186.2	305.6	*119.3*
2000/05	831.9	845.8	*13.93*	639.7	557.3	*-82.4*	27.13	111.4	*84.29*	177.3	177.5	*0.178*
2000/06	1024	1117	*92.96*	631.3	690.3	*58.91*	389.2	374.5	*-14.7*	405.2	344.1	*-61.1*
2000/07	768.8	764.1	*-4.66*	504	396.9	*-107*	310.6	219.8	*-90.8*	405.2	268.9	*-136*
2000/08	662.9	586.6	*-76.3*	499.2	217.1	*-282*	274.3	191	*-83.3*	313	247.6	*-65.4*
2000/09	221.4	208.9	*-12.5*	152.3	57.76	*-94.6*	34.69	2.76	*-31.9*	93.42	56.57	*-36.9*
2000/10	230.3	209.6	*-20.7*	167.8	112.4	*-55.4*	88.66	69.14	*-19.5*	168.4	139.2	*-29.2*
2000/11	268.4	229.7	*-38.7*	264.2	158.4	*-106*	21.9	6.92	*-15*	133.3	82.99	*-50.3*
2000/12	194	170.9	*-23.1*	274.3	178.8	*-95.5*	23.68	8.25	*-15.4*	116	89.3	*-26.7*
2001/01	219	197.3	*-21.7*	185.7	246.2	*60.5*	18.92	13.34	*-5.58*	103.5	92.96	*-10.6*
2001/02	218.4	197.6	*-20.8*	96.99	197.8	*100.8*	18.86	2.76	*-16.1*	113.1	67.43	*-45.6*
2001/03	294	270.5	*-23.5*	212.4	145.5	*-67*	11.84	2.76	*-9.08*	105.3	90.26	*-15.1*
2001/04	436.2	413.3	*-22.9*	310.6	141.7	*-169*	9.997	33.7	*23.7*	93.42	132.7	*39.27*
2001/05	1013	985.8	*-26.9*	719.4	570.6	*-149*	41.89	50.45	*8.559*	163.6	204	*40.39*
2001/06	1025	1136	*110.8*	645	660.1	*15.05*	152.9	71.83	*-81.1*	226.1	112.2	*-114*
2001/07	727.1	781	*53.82*	502.2	446.5	*-55.7*	313.6	238.7	*-74.9*	361.8	263.7	*-98.1*
2001/08	633.7	611.2	*-22.5*	408.2	236.2	*-172*	158.3	106.4	*-51.9*	219.6	152.4	*-67.2*
2001/09	250.5	210.8	*-39.7*	172	66.01	*-106*	40.52	2.76	*-37.8*	77.36	41.44	*-35.9*
2001/10	213.6	216.6	*2.97*	158.9	112.9	*-46*	11.9	8.96	*-2.94*	64.26	60.1	*-4.16*
2001/11	227.9	203.2	*-24.7*	219	170.5	*-48.5*	6.307	2.76	*-3.55*	59.5	67.24	*7.736*
2001/12	179.1	168.3	*-10.8*	211.8	179.8	*-32*	4.838	2.76	*-2.08*	70.81	67.89	*-2.92*

Appendix B: Data of Observed and Simulated Monthly River EC in the Arkansas River from January 1986 to December 2001

units: dS/m

yr/mon	Avondale			Fowler			JM Reservoir			Lamar			Coolidge		
	obs	sim	*error*	obs	sim	*error*	obs	sim	*error*	obs	sim	*error*	obs	Sim	*error*
1986/01	0.76	0.68	-0.08		1.16		3.31	3.21	-0.10		4.91			5.59	
1986/02	0.80	0.75	-0.05		1.12		3.28	3.11	-0.17		4.70		4.89	5.69	0.80
1986/03	0.85	0.79	-0.05		1.07		2.18	2.52	0.34		3.29			5.23	
1986/04	0.82	0.80	-0.02		1.06		2.20	2.38	0.18		2.81			4.20	
1986/05	0.72	0.73	0.02		0.96		2.24	2.23	-0.01		2.66		4.00	3.80	-0.20
1986/06	0.55	0.51	-0.04		0.67		2.15	1.93	-0.22		2.25			2.88	
1986/07	0.31	0.41	0.10		0.64		2.14	1.76	-0.38		2.02		2.89	2.62	-0.27
1986/08	0.51	0.54	0.03		0.97		2.10	1.75	-0.35		2.13			2.41	
1986/09	0.69	0.69	0.00		1.20		2.17	1.90	-0.27		2.82			3.58	
1986/10	0.76	0.76	0.01		1.32		2.21	1.92	-0.29	3.33	2.97	-0.36	3.63	3.61	-0.02
1986/11	1.19	0.78	-0.41		1.36		2.46	2.30	-0.16	4.40	3.63	-0.77		4.36	
1986/12	0.77	0.72	-0.05		1.23		2.45	2.45	0.00	4.41	4.16	-0.25		4.79	
1987/01	0.88	0.78	-0.10		1.22		2.43	2.61	0.18	4.50	4.19	-0.31		4.88	
1987/02	0.78	0.75	-0.03		1.14		2.40	2.43	0.03	4.55	3.23	-1.32		4.30	
1987/03	0.75	0.72	-0.03		1.05			2.13			2.91		4.64	4.07	-0.57
1987/04	0.59	0.71	0.12		0.92		2.14	1.97	-0.17	2.26	2.22	-0.04	2.88	3.04	0.16
1987/05	0.57	0.59	0.02		0.71		2.01	1.59	-0.42	2.19	1.70	-0.49	2.38	2.09	-0.29
1987/06	0.44	0.46	0.02		0.67		1.56	1.40	-0.16	1.65	1.53	-0.12		1.84	
1987/07	0.45	0.50	0.05		0.71		1.39	1.42	0.03	1.54	1.76	0.22	2.24	2.25	0.01
1987/08	0.53	0.60	0.07		0.89		1.40	1.41	0.01	1.68	1.74	0.06		2.23	
1987/09	0.71	0.72	0.01		1.19		1.50	1.50	0.00	1.61	2.11	0.50		2.75	
1987/10	0.81	0.83	0.02		1.34		1.55	1.58	0.03		2.48		4.22	3.54	-0.68
1987/11	0.81	0.89	0.08		1.37		1.65	1.75	0.10	3.40	2.89	-0.51	3.95	3.69	-0.26
1987/12	1.14	0.94	-0.20		1.39		2.23	2.00	-0.23	4.22	3.23	-0.99	4.41	4.00	-0.41
1988/01	0.81	0.82	0.01		1.29		2.26	2.25	-0.01	4.21	3.75	-0.46		4.26	
1988/02	0.89	0.91	0.02		1.29		2.26	2.50	0.24	4.40	4.25	-0.15		4.70	
1988/03	0.91	0.88	-0.03		1.21		2.00	2.05	0.05	3.30	3.36	0.06	4.47	4.63	0.16
1988/04	0.84	0.88	0.04		1.18		1.93	1.91	-0.02	2.09	2.42	0.33	2.53	3.66	1.13
1988/05	0.73	0.83	0.10		1.09		2.01	1.93	-0.08	3.61	2.48	-1.13	3.40	3.37	-0.03
1988/06	0.74	0.83	0.09		1.03		2.04	1.93	-0.11		2.26		2.63	2.97	0.34
1988/07	0.58	0.65	0.07		0.88		2.09	1.92	-0.17	2.17	2.16	-0.01	1.98	2.82	0.84
1988/08	0.60	0.71	0.11		1.04		2.12	1.99	-0.13	2.24	2.36	0.12	2.67	3.01	0.34
1988/09	0.79	0.86	0.07		1.37		2.16	2.10	-0.06	2.60	2.75	0.15	4.10	3.28	-0.82
1988/10	0.86	0.98	0.12		1.63		2.29	2.27	-0.02	3.80	3.41	-0.39	4.48	4.43	-0.05
1988/11	1.10	1.09	-0.01		1.68		2.44	2.63	0.19	3.80	3.93	0.13		4.90	
1988/12	1.16	1.14	-0.02		1.68		2.64	2.81	0.17	4.40	4.24	-0.16	2.54	5.16	2.62
1989/01	1.13	1.13	0.00		1.66		2.75	3.09	0.34	4.35	4.67	0.32	4.42	5.32	0.90
1989/02	0.93	1.02	0.10		1.55		2.80	3.21	0.41	4.40	4.84	0.44		5.58	
1989/03	0.82	0.90	0.08		1.47		2.69	2.83	0.14	4.02	3.99	-0.03	4.61	5.29	0.68
1989/04	0.73	0.85	0.12		1.26		2.69	2.67	-0.02	3.20	3.65	0.45	4.59	5.09	0.50
1989/05	0.75	0.77	0.02		1.11		2.71	2.50	-0.21		2.84		4.58	3.77	-0.81
1989/06	0.60	0.72	0.12		0.98		2.69	2.43	-0.26	3.09	2.85	-0.24	4.11	3.68	-0.43
1989/07	0.54	0.61	0.07		0.84		2.60	2.27	-0.33	2.99	2.65	-0.34	3.56	3.59	0.03
1989/08	0.57	0.64	0.07		0.99		2.16	2.27	0.11	3.39	2.68	-0.71		3.25	

1989/09	0.93	0.92	-0.01		1.45		2.08	2.48	0.40	3.70	3.22	-0.48	4.13	4.30	0.17
1989/10	0.83	1.01	0.18		1.70		2.18	2.67	0.49	2.41	3.77	1.36		5.40	
1989/11	0.82	1.10	0.28		1.80		2.49	3.15	0.66	3.76	4.63	0.87		5.99	
1989/12	1.08	1.15	0.07		1.77		2.75	3.23	0.48	4.26	4.74	0.48		6.01	
1990/01	1.15	1.19	0.04		1.73		2.89	3.34	0.45		4.48			5.73	
1990/02	1.00	1.07	0.08		1.62		2.89	3.28	0.39	4.58	4.16	-0.42		5.33	
1990/03	0.80	0.99	0.19		1.50		2.89	3.34	0.45	4.47	4.42	-0.05	5.40	5.45	0.05
1990/04	0.96	0.95	-0.01	1.12	1.39	0.27	2.86	2.85	-0.01	4.36	3.84	-0.52		5.05	
1990/05	0.93	0.86	-0.07	1.05	1.17	0.12	2.67	2.50	-0.17	3.84	3.07	-0.77	3.97	4.16	0.19
1990/06	0.65	0.75	0.10	0.72	0.95	0.23	2.43	2.28	-0.15	3.71	2.55	-1.16	2.72	3.37	0.65
1990/07	0.65	0.59	-0.06	0.93	0.87	-0.06	1.53	1.94	0.41	2.60	2.33	-0.27	3.56	3.19	-0.37
1990/08	0.61	0.69	0.08	0.72	1.02	0.30	1.49	2.03	0.54	3.12	2.81	-0.31	4.67	4.03	-0.64
1990/09		0.82		0.86	1.28	0.42		2.15		3.62	3.06	-0.56	4.35	4.32	-0.03
1990/10	0.87	0.86	-0.01	1.08	1.41	0.33	1.88	2.34	0.46	3.43	3.65	0.22	4.50	5.34	0.84
1990/11		0.90		1.20	1.38	0.18		2.61		3.87	3.54	-0.33	4.91	4.93	0.02
1990/12	1.12	1.00	-0.12	1.51	1.41	-0.10	2.59	2.79	0.20	4.26	4.01	-0.25	5.26	5.35	0.09
1991/01		0.92		1.17	1.35	0.18	2.66	3.17	0.51	4.32	4.63	0.31	4.74	5.49	0.75
1991/02		0.95			1.34		3.12	3.28	0.16	4.17	4.94	0.77		5.87	
1991/03	0.80	0.91	0.11	1.02	1.30	0.28	2.46	3.09	0.63	3.93	4.20	0.27	5.29	5.43	0.14
1991/04	0.85	0.93	0.08	1.21	1.30	0.09	2.58	2.45	-0.13	3.35	3.43	0.08		4.95	
1991/05	0.70	0.79	0.09	0.90	1.11	0.21	2.71	2.35	-0.36	3.88	3.19	-0.69	4.70	4.31	-0.39
1991/06	0.60	0.65	0.05	0.68	0.88	0.20	2.53	2.03	-0.50	2.04	2.35	0.31	3.43	3.15	-0.28
1991/07	0.55	0.62	0.07	0.59	0.87	0.28	1.45	1.90	0.45	2.94	2.33	-0.61	2.87	3.23	0.37
1991/08	0.61	0.67	0.06	0.64	0.99	0.35	1.40	1.97	0.57	3.26	2.41	-0.85	4.21	3.10	-1.11
1991/09	0.62	0.86	0.24	1.21	1.39	0.18	1.53	2.20	0.67		3.27		4.24	4.56	0.32
1991/10	1.05	1.03	-0.02	1.32	1.62	0.30	2.33	2.46	0.13	3.55	3.85	0.30	4.68	5.33	0.66
1991/11		0.96			1.56			2.72		4.16	3.77	-0.39		4.77	
1991/12	1.23	1.07	-0.16	1.50	1.58	0.08	2.60	3.03	0.43	4.21	4.58	0.37	3.80	5.37	1.57
1992/01		1.00			1.50			3.19		4.50	4.90	0.40		5.62	
1992/02		0.95		1.41	1.43	0.02		3.23		4.45	4.64	0.19	5.18	5.61	0.43
1992/03	0.81	0.86	0.05	0.96	1.32	0.36	2.78	3.18	0.40	4.45	4.37	-0.08	5.36	5.53	0.17
1992/04	0.77	0.78	0.02	0.91	1.15	0.24	2.68	2.57	-0.11	3.08	3.46	0.38	4.83	5.25	0.42
1992/05	0.72	0.74	0.02	0.84	0.99	0.15	2.56	2.30	-0.26	2.93	2.86	-0.07	4.59	4.18	-0.41
1992/06	0.59	0.62	0.03	0.65	0.83	0.18	2.06	2.04	-0.02	3.51	2.52	-0.99		3.44	
1992/07	0.52	0.62	0.10	0.63	0.88	0.25	1.55	1.96	0.41		2.31			3.00	
1992/08	0.62	0.68	0.06	0.78	0.99	0.21	1.70	2.05	0.35	3.09	2.62	-0.47		3.38	
1992/09		0.79			1.22			2.22		4.01	3.35	-0.66	3.86	4.83	0.97
1992/10	0.85	0.92	0.07	1.14	1.47	0.33	2.07	2.45	0.38	3.91	3.72	-0.19	4.59	5.50	0.91
1992/11		0.88			1.43		2.35	2.66	0.31	4.06	3.52	-0.54		4.70	
1992/12		1.00			1.49		2.65	3.02	0.37	4.29	4.62	0.33		5.52	
1993/01	0.97	0.99	0.02	1.36	1.46	0.10	2.71	3.09	0.38	4.33	4.54	0.21		5.25	
1993/02		0.97			1.43		2.77	3.03	0.26		4.14			5.03	
1993/03	0.84	0.90	0.06	1.04	1.40	0.36	2.75	2.95	0.20	4.53	3.96	-0.57	4.66	4.90	0.24
1993/04		0.94			1.28		2.47	2.48	0.01	4.21	3.19	-1.02		4.38	
1993/05		0.87			1.13		2.48	2.23	-0.25	4.02	2.71	-1.31		3.54	
1993/06		0.66			0.85		2.16	2.01	-0.15	4.11	2.35	-1.76	2.49	3.02	0.53
1993/07		0.56			0.77		1.75	1.91	0.16	1.89	2.27	0.38		3.06	
1993/08		0.53			0.79		1.55	2.00	0.45	2.70	2.63	-0.07		3.26	
1993/09		0.71			1.10		1.62	2.11	0.49	4.29	3.04	-1.25	4.45	4.21	-0.24
1993/10	0.81	0.72	-0.09		1.21		1.78	2.22	0.44	3.81	3.32	-0.49		4.46	

1993/11	1.00	0.88	-0.12		1.30		2.18	2.51	0.33	4.34	3.58	-0.76		4.87	
1993/12	1.15	0.97	-0.18		1.36		2.61	2.73	0.12	4.53	4.26	-0.27	5.25	5.29	0.04
1994/01	1.06	0.97	-0.09		1.37		2.54	2.94	0.40	4.50	4.46	-0.04		5.29	
1994/02	0.94	0.99	0.05		1.38		2.59	3.08	0.49	4.53	4.82	0.29		5.67	
1994/03	0.83	0.92	0.10		1.33		2.56	2.63	0.07	4.48	4.12	-0.36	4.87	5.37	0.50
1994/04	0.65	0.63	-0.02		1.04		2.45	2.34	-0.11	3.51	3.07	-0.44	4.83	4.38	-0.45
1994/05	0.60	0.61	0.01	0.65	0.80	0.15	2.28	2.05	-0.23	3.78	2.68	-1.10	4.32	4.05	-0.27
1994/06	0.48	0.54	0.06	0.55	0.68	0.13	1.93	1.76	-0.17	1.93	2.08	0.15	3.96	3.01	-0.95
1994/07	0.44	0.48	0.04	0.66	0.71	0.05	1.56	1.72	0.16	1.76	2.14	0.38	2.23	2.96	0.73
1994/08	0.61	0.65	0.04	0.82	0.91	0.09	1.63	1.76	0.13	3.64	2.52	-1.12		3.45	
1994/09	0.72	0.72	0.00	0.99	1.16	0.17	1.65	1.84	0.19	3.35	2.85	-0.50	4.81	4.36	-0.45
1994/10	0.86	0.79	-0.07	1.07	1.27	0.20	1.76	1.93	0.17	3.71	3.24	-0.47	4.04	4.69	0.65
1994/11		0.85		1.06	1.36	0.30	2.17	2.22	0.05	4.12	3.24	-0.88	4.32	4.70	0.38
1994/12	1.07	0.86	-0.21	1.27	1.31	0.04	2.44	2.41	-0.03	4.33	3.86	-0.47	4.56	5.10	0.54
1995/01	1.12	0.97	-0.15		1.35		2.45	2.71	0.26	4.31	4.27	-0.04		5.14	
1995/02	1.10	1.01	-0.09	1.37	1.35	-0.01	2.59	2.86	0.27	4.56	4.44	-0.12		5.35	
1995/03	0.87	0.94	0.07	1.26	1.34	0.08	2.33	2.47	0.14	4.26	3.77	-0.49	3.90	5.05	1.15
1995/04		0.82			1.19		2.35	2.25	-0.10	4.21	3.18	-1.03	4.59	4.56	-0.03
1995/05	0.63	0.63	0.01		0.83		2.35	2.06	-0.29	2.81	2.64	-0.17	2.67	3.64	0.97
1995/06	0.55	0.43	-0.12		0.55		2.23	1.66	-0.57	3.38	2.30	-1.08	3.57	3.22	-0.35
1995/07	0.38	0.33	-0.05		0.49		1.64	1.35	-0.29	1.42	1.46	0.04	1.57	1.79	0.22
1995/08	0.40	0.48	0.08		0.62		1.18	1.37	0.19	3.07	1.84	-1.23	3.72	2.51	-1.21
1995/09		0.50		0.82	0.75	-0.07	1.10	1.40	0.30	1.80	1.88	0.08	2.82	2.59	-0.23
1995/10	0.69	0.67	-0.02		0.95		1.25	1.50	0.25	2.59	2.48	-0.11	3.06	3.54	0.48
1995/11	0.86	0.79	-0.07		1.06		1.56	1.65	0.10	3.99	2.78	-1.21		3.76	
1995/12		0.85			1.14			1.92			3.39		5.06	4.20	-0.86
1996/01	1.02	0.92	-0.10		1.20		2.03	2.22	0.19	4.04	3.93	-0.11		4.48	
1996/02	0.89	0.86	-0.03	1.18	1.19	0.01	2.04	2.17	0.13	4.00	4.03	0.03	4.35	4.78	0.43
1996/03	0.71	0.76	0.05	0.97	1.13	0.16	1.66	1.78	0.13	2.48	2.83	0.35		4.25	
1996/04	0.67	0.74	0.07	0.95	1.02	0.07	1.68	1.75	0.08	1.99	2.32	0.33	2.59	3.65	1.06
1996/05	0.63	0.65	0.02		0.85		1.70	1.76	0.06	1.95	2.11	0.16	2.94	2.84	-0.10
1996/06	0.56	0.61	0.05		0.77		1.80	1.72	-0.08	3.61	2.06	-1.55		2.64	
1996/07	0.79	0.64	-0.15	0.89	0.83	-0.06	1.83	1.69	-0.14	1.94	2.00	0.06	3.04	2.45	-0.59
1996/08	0.51	0.59	0.08	0.68	0.88	0.20	1.89	1.78	-0.11	2.69	2.49	-0.20	3.80	2.82	-0.98
1996/09	0.88	0.83	-0.05	1.28	1.28	0.00	1.93	1.88	-0.05		2.96			3.42	
1996/10	0.95	0.97	0.02		1.47		1.99	1.93	-0.06	3.16	3.35	0.19		4.32	
1996/11		0.96			1.57		2.05	2.22	0.17	3.60	3.98	0.38		4.73	
1996/12	1.09	1.05	-0.04	1.43	1.56	0.13	2.29	2.45	0.17	4.12	4.31	0.19	4.43	5.10	0.67
1997/01	0.86	0.97	0.11		1.49		2.41	2.69	0.28	4.27	4.61	0.34	4.40	5.31	0.91
1997/02	1.02	1.02	0.00	1.14	1.44	0.30	2.46	2.75	0.29	4.28	4.36	0.08		5.17	
1997/03	0.67	0.74	0.07	1.04	1.24	0.20	2.33	2.20	-0.13	3.93	3.26	-0.67	2.82	4.91	2.09
1997/04	0.62	0.67	0.05	0.72	1.00	0.28	2.28	2.13	-0.15	2.86	2.73	-0.13	3.59	3.98	0.39
1997/05	0.57	0.62	0.05		0.85		2.27	2.10	-0.17	2.92	2.67	-0.25	4.06	3.67	-0.39
1997/06	0.45	0.48	0.03	0.51	0.58	0.07	2.35	1.74	-0.61	3.28	2.17	-1.11	2.81	3.01	0.20
1997/07		0.54		0.57	0.71	0.14	1.99	1.69	-0.30	2.14	2.00	-0.14	2.92	2.72	-0.20
1997/08		0.56		1.13	0.80	-0.33	1.85	1.66	-0.19		2.02		2.35	2.33	-0.02
1997/09	0.70	0.67	-0.03	1.00	1.02	0.03	1.78	1.71	-0.07	2.85	2.58	-0.27	3.97	3.18	-0.79
1997/10	0.76	0.79	0.03	1.14	1.25	0.11	1.83	1.75	-0.08	3.36	2.55	-0.81	3.51	3.09	-0.42
1997/11	0.80	0.80	0.00	0.99	1.21	0.23	2.11	2.06	-0.05	4.16	3.44	-0.72	4.41	4.12	-0.29
1997/12	0.78	0.80	0.02	1.05	1.21	0.16	2.04	1.87	-0.17	2.56	2.60	0.04		3.60	

1998/01	1.11	0.95	-0.16	1.40	1.27	-0.13	2.10	1.90	-0.20	2.25	2.46	0.21	2.23	3.66	1.43
1998/02	1.12	1.00	-0.12	1.39	1.30	-0.09	2.02	1.96	-0.06	2.80	2.59	-0.21		3.54	
1998/03	0.83	0.93	0.10	1.18	1.28	0.11		1.96		2.23	2.32	0.09	3.78	3.10	-0.68
1998/04	0.74	0.75	0.01	0.88	1.03	0.15	2.04	1.95	-0.09	2.26	2.25	-0.01		2.90	
1998/05	0.62	0.80	0.18	0.81	0.98	0.17	2.03	1.92	-0.11	3.00	2.30	-0.70	2.58	2.87	0.29
1998/06	0.63	0.70	0.07	0.80	0.90	0.10	2.02	1.86	-0.16	2.11	2.16	0.05	2.57	2.75	0.18
1998/07	0.51	0.59	0.08	0.65	0.80	0.15	2.04	1.77	-0.27		1.91			2.22	
1998/08	0.62	0.56	-0.06	0.89	0.92	0.03	2.05	1.83	-0.22	3.01	2.34	-0.67	3.18	2.76	-0.42
1998/09	0.63	0.67	0.04	0.97	1.08	0.11	2.06	1.88	-0.18	2.94	2.72	-0.22	3.48	3.43	-0.05
1998/10	0.88	0.87	-0.01	1.19	1.38	0.19	2.10	1.97	-0.13	3.08	2.90	-0.18	3.20	3.46	0.26
1998/11	1.03	0.93	-0.10	1.39	1.35	-0.04	2.12	2.18	0.06	3.68	3.24	-0.44	3.97	3.97	0.00
1998/12	1.08	0.99	-0.09	1.38	1.44	0.06	2.23	2.44	0.21	4.04	4.03	-0.01		4.45	
1999/01	1.05	1.04	-0.01	1.36	1.44	0.09	2.35	2.65	0.30	4.16	4.00	-0.16	4.54	4.41	-0.13
1999/02	1.04	1.05	0.01	1.46	1.46	0.00	2.33	2.67	0.34	4.16	4.27	0.11	4.85	4.93	0.08
1999/03	0.86	0.89	0.03	1.36	1.44	0.08	2.23	2.41	0.18	3.82	3.46	-0.36		4.33	
1999/04	0.87	0.76	-0.11	1.06	1.10	0.04	2.26	2.20	-0.06	2.64	2.70	0.06	4.55	3.64	-0.91
1999/05	0.68	0.72	0.04	0.86	0.87	0.01	1.93	1.83	-0.10	2.02	1.93	-0.09	2.08	2.27	0.19
1999/06	0.60	0.63	0.03	0.70	0.76	0.06	1.86	1.64	-0.22	1.90	1.77	-0.13	2.16	2.08	-0.08
1999/07	0.61	0.63	0.02	0.82	0.83	0.01	1.75	1.60	-0.15	1.96	1.87	-0.09	1.83	2.24	0.41
1999/08	0.59	0.53	-0.06	0.78	0.75	-0.03	1.63	1.50	-0.13	1.84	1.77	-0.07	2.23	2.12	-0.11
1999/09	0.75	0.64	-0.11	0.94	0.94	0.00		1.59		2.57	2.39	-0.18	3.52	2.94	-0.58
1999/10	0.88	0.75	-0.13	1.17	1.07	-0.10	1.67	1.69	0.02	2.79	2.82	0.03	3.60	3.38	-0.22
1999/11	0.91	0.83	-0.08	1.13	1.14	0.01	1.76	1.78	0.03	2.94	2.94	0.00		3.93	
1999/12	0.95	0.79	-0.16	1.25	1.17	-0.08	1.92	2.11	0.19	4.16	3.64	-0.52	3.92	4.28	0.36
2000/01	0.87	0.81	-0.06	1.18	1.15	-0.03	2.12	2.17	0.05	4.06	3.42	-0.64		4.34	
2000/02	1.06	0.91	-0.15	1.43	1.17	-0.26	2.03	1.93	-0.10	2.50	3.01	0.51	3.27	4.34	1.07
2000/03	0.86	0.79	-0.07	1.21	1.13	-0.08	2.11	1.91	-0.20	2.61	2.70	0.09	3.06	3.68	0.62
2000/04	0.77	0.76	-0.01	1.04	1.06	0.02	2.01	1.90	-0.11	2.71	2.36	-0.35	3.60	3.36	-0.24
2000/05	0.72	0.74	0.02	0.95	0.98	0.03	2.07	1.91	-0.16	2.68	2.43	-0.25	4.23	3.50	-0.73
2000/06	0.64	0.69	0.05	0.69	0.90	0.21	2.10	1.86	-0.24	2.19	2.14	-0.05	3.50	2.91	-0.59
2000/07	0.54	0.62	0.08	0.79	0.90	0.11	2.14	1.87	-0.27		2.18			2.85	
2000/08	0.57	0.67	0.10	0.74	1.07	0.33	2.15	1.94	-0.21	2.31	2.39	0.08	2.96	3.17	0.21
2000/09	0.99	0.94	-0.05	1.30	1.43	0.13	2.27	2.06	-0.21	3.24	3.02	-0.22	4.00	4.16	0.16
2000/10	0.97	1.01	0.05	1.27	1.61	0.34	2.29	2.16	-0.13		2.97		2.83	3.77	0.94
2000/11	1.01	0.98	-0.03	1.09	1.59	0.50	2.35	2.47	0.12	3.51	3.90	0.39	4.13	4.96	0.83
2000/12	1.06	1.06	0.00	1.35	1.61	0.26		2.63		3.79	4.17	0.38		5.10	
2001/01	1.06	1.05	-0.01	1.39	1.55	0.16		2.85		3.98	4.15	0.17		5.02	
2001/02	1.04	1.08	0.04	1.53	1.54	0.01		3.08			4.70			5.40	
2001/03	0.93	0.91	-0.02	1.18	1.44	0.26		3.02		4.10	4.30	0.20		5.16	
2001/04	0.76	0.78	0.02	0.99	1.23	0.24	2.52	2.60	0.08	3.57	3.82	0.25	3.96	5.24	1.28
2001/05	0.67	0.66	-0.01	0.80	0.97	0.18	2.49	2.32	-0.17	3.45	2.64	-0.81		3.32	
2001/06	0.55	0.58	0.03	0.70	0.80	0.10	2.45	2.14	-0.31	2.46	2.53	0.07	3.33	3.47	0.15
2001/07	0.58	0.59	0.01	0.85	0.86	0.01	2.28	2.02	-0.26	2.43	2.27	-0.16	2.84	2.84	0.00
2001/08	0.62	0.60	-0.02	0.76	1.02	0.26	2.17	2.05	-0.12	3.36	2.55	-0.81	3.80	3.45	-0.35
2001/09	0.93	0.89	-0.04	1.18	1.37	0.19	2.13	2.14	0.01	2.30	3.03	0.73	4.39	4.31	-0.08
2001/10	1.10	0.96	-0.14	1.49	1.56	0.07	2.25	2.30	0.05		3.62		4.04	5.15	1.11
2001/11	1.01	1.01	0.00		1.57			2.57			4.07			5.37	
2001/12	0.93	1.03	0.10		1.58			2.81			4.40			5.53	

Appendix C: Groundwater Table Elevation Data in 1968 and 1999

(1) Spring 1968 vs. April 5, 1999

UTM_x	UTM_y	Land Elevation (m)	GW Level in April 5, 1999 (m)	GW Level in Spring, 1968 (m)	*Difference (m)*
603165.6	**4212327.9**	1309.33	1303.10	1305.16	*-2.06*
608133.3	**4218329.2**	1275.15	1270.78	1272.83	*-2.05*
609663.2	**4217031.1**	1273.84	1270.02	1269.44	*0.58*
606918.5	**4216961.8**	1282.95	1274.88	1277.77	*-2.89*
608285.7	**4217406.1**	1275.23	1271.97	1271.61	*0.36*
609249.7	**4215011.3**	1283.45	1277.12	1275.01	*2.11*
611288.6	**4213300.4**	1276.36	1274.22	1271.97	*2.25*
613996.0	**4214009.1**	1263.12	1261.32	1260.96	*0.36*
617703.1	**4211315.0**	1259.68	1253.81	1255.06	*-1.25*
620961.3	**4208802.4**	1250.51	1247.07	1246.22	*0.85*
622852.2	**4207698.6**	1247.20	1244.28	1243.45	*0.83*
631340.0	**4206846.5**	1231.09	1229.56	1227.65	*1.91*
632044.2	**4208149.2**	1229.46	1228.50	1225.12	*3.38*
635431.2	**4210605.2**	1228.21	1224.33	1221.29	*3.04*
641079.1	**4213027.8**	1213.41	1210.31	1210.33	*-0.02*
614760.9	**4212856.1**	1260.67	1259.63	1259.23	*0.40*
		Average (m) =	**1256.31**	**1255.82**	***0.49***
		Std Dev(m)=	**24.26**	**25.23**	

(2) Fall 1968 vs. October 1, 1999

UTM_x	UTM_y	Land Elevation (m)	GW Level in Oct 1, 1999 (m)	GW Level in Fall, 1968 (m)	*Difference (m)*
603165.6	**4212327.9**	1309.33	1303.12	1305.62	*-2.50*
604214.6	**4218759.6**	1282.77	1279.85	1280.46	*-0.61*
608133.3	**4218329.2**	1275.15	1270.53	1272.05	*-1.52*
606918.5	**4216961.8**	1282.95	1274.98	1277.20	*-2.22*
609249.7	**4215011.3**	1283.45	1277.15	1276.71	*0.44*
611288.6	**4213300.4**	1276.36	1273.94	1272.18	*1.76*
613996.0	**4214009.1**	1263.12	1261.62	1260.96	*0.66*
619305.1	**4210903.7**	1254.18	1252.33	1250.67	*1.66*
620961.3	**4208802.4**	1250.51	1247.25	1246.43	*0.82*
622852.2	**4207698.6**	1247.2	1243.93	1243.65	*0.28*
626175.4	**4205690.2**	1241.19	1236.57	1236.43	*0.14*
631340.0	**4206846.5**	1231.09	1229.46	1227.33	*2.13*
632044.2	**4208149.2**	1229.46	1229.03	1224.97	*4.06*
635431.2	**4210605.2**	1228.21	1225.19	1221.03	*4.16*
639394.1	**4211832.2**	1215.54	1213.07	1212.33	*0.74*
614760.9	**4212856.1**	1260.67	1259.71	1258.60	*1.11*
		Average (m) =	**1254.86**	**1254.16**	***0.69***
		Std Dev(m)=	**24.48**	**25.83**	

Appendix D: Summary of Crop Reduction Threshold, $EC_{e,\ threshold}$, and Reduction Slope, b, for Selected Crops

Crop Type	$EC_{e,\ threshold}$ (dS m^{-1})	b (%/dS m^{-1})
Onion	1.2	16.0
Tomato	0.9-2.5	9.0
Potato	1.7	12.0
Beans	1.0	19.0
Sweet Corn	1.7	12.0
Sorghum	6.8	16.0
Asparagus	4.1	2.0
Cotton	7.7	5.2
Wheat	6.0	7.1
Wheat, semidwarf	8.6	3.0
Wheat, durum	5.7-5.9	3.8-5.5
Alfalfa	2.0	7.3
Maize	1.8	7.4
Rye-grass	5.6	7.6
Wheatgrass, tall	7.5	4.2

Appendix E: Simulated Crop Consumptive Use in the Arkansas River Basin from January 1986 to December 1990 Due to Shallow Water Table and High Soil Salinity Problems

units: acre-ft

Year	Bess-emer	Booth	Excel-sior	Collier	Colora-do	Highline	Oxford	Otero	Catlin	Holbrook	Rocky Ford
1986/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1986/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1986/03	70.1	7.5	24.3	1.7	327.6	114.9	9.1	61.6	114.2	111.1	131.3
1986/04	413.7	40.1	158.4	11.8	1643.0	646.8	77.4	288.9	591.3	575.9	591.1
1986/05	849.1	76.8	269.0	21.0	2821.8	1151.5	154.6	477.0	1033.6	988.2	986.7
1986/06	1302.1	116.6	390.5	30.7	4035.5	1654.3	225.9	685.8	1493.8	1419.4	1423.8
1986/07	1305.6	116.8	469.0	36.9	4853.5	1988.7	269.8	827.5	1801.8	1707.1	1720.2
1986/08	1301.1	116.9	401.4	31.5	4331.3	1771.7	233.4	742.4	1597.4	1527.9	1525.5
1986/09	809.9	79.6	244.4	17.7	2377.4	878.6	95.1	441.6	853.2	817.0	932.6
1986/10	384.3	41.1	135.3	9.2	1454.9	512.2	42.0	278.7	509.7	483.3	586.5
1986/11	16.5	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1986/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/03	69.8	7.5	24.2	1.6	327.6	115.0	9.1	61.8	114.6	111.1	132.3
1987/04	412.2	39.9	157.2	11.7	1643.1	647.0	77.5	288.9	592.1	575.9	593.1
1987/05	848.4	76.8	269.0	21.0	2821.4	1151.3	154.6	477.0	1032.9	987.9	986.3
1987/06	1302.1	116.6	390.2	30.6	4035.5	1654.3	225.9	685.7	1493.5	1418.8	1423.2
1987/07	1305.6	116.8	469.0	36.9	4853.5	1988.7	269.8	827.5	1801.6	1706.7	1719.7
1987/08	1301.1	116.9	401.3	31.5	4331.1	1771.6	233.4	742.4	1597.0	1527.8	1524.9
1987/09	809.9	79.6	244.4	17.7	2377.3	878.6	95.1	441.6	853.1	817.0	932.5
1987/10	384.3	41.1	135.3	9.2	1454.9	512.2	42.0	278.7	509.7	483.3	586.5
1987/11	16.5	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/03	69.9	7.5	24.2	1.6	327.3	115.4	9.2	61.8	115.1	109.1	133.2
1988/04	413.1	39.9	158.3	11.7	1646.5	647.6	77.7	286.1	594.6	567.8	596.1
1988/05	852.1	76.8	269.0	20.9	2836.6	1157.4	155.5	483.6	1046.1	999.1	1004.6
1988/06	1302.1	116.6	390.5	30.6	4035.5	1652.6	225.6	686.1	1491.9	1418.3	1422.5
1988/07	1305.5	116.7	469.4	36.9	4830.7	1986.1	269.2	788.2	1794.6	1703.1	1712.7
1988/08	1302.2	116.9	401.8	31.5	4330.6	1770.7	233.2	738.0	1599.2	1521.0	1529.0
1988/09	808.7	79.6	244.4	17.7	2323.6	874.4	94.6	434.2	845.7	814.2	924.3
1988/10	383.8	41.1	135.3	9.2	1443.9	506.7	41.7	260.7	503.2	475.4	580.0
1988/11	16.6	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/03	69.9	7.5	24.2	1.6	328.1	115.4	9.2	62.8	115.3	108.9	134.1
1989/04	412.5	39.9	158.2	11.7	1649.1	647.4	77.3	288.7	592.6	574.3	596.4
1989/05	850.9	76.8	268.1	21.0	2836.6	1153.7	154.5	483.4	1037.0	993.8	991.4

1989/06	1297.5	116.6	389.5	30.5	4024.5	1645.7	224.1	681.5	1478.2	1409.6	1404.5
1989/07	1304.4	116.8	469.4	36.8	4826.7	1983.8	268.7	817.0	1789.6	1704.2	1702.4
1989/08	1300.0	116.9	401.8	31.4	4237.1	1764.0	232.3	714.2	1584.7	1500.8	1508.6
1989/09	810.0	79.6	244.4	17.7	2365.9	873.9	94.5	432.9	840.1	817.5	918.2
1989/10	383.2	41.2	135.7	9.2	1451.8	502.0	41.3	273.9	494.4	493.5	566.8
1989/11	16.5	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/03	69.8	7.5	24.1	1.6	322.7	114.2	9.0	61.1	113.0	107.0	130.1
1990/04	412.6	39.9	158.3	11.7	1635.2	643.6	77.0	288.9	586.4	560.6	584.6
1990/05	850.3	76.8	269.6	21.0	2835.4	1153.8	154.5	483.3	1033.7	993.2	984.8
1990/06	1298.7	116.6	390.7	30.7	4029.0	1650.9	224.8	685.8	1483.2	1415.6	1406.4
1990/07	1303.8	116.8	469.4	36.9	4839.1	1985.3	269.0	822.6	1790.9	1695.6	1702.4
1990/08	1298.3	116.9	401.8	31.5	4337.4	1772.3	233.3	741.2	1597.1	1512.3	1522.6
1990/09	809.3	79.6	244.4	17.7	2381.6	877.1	95.0	443.1	846.0	817.5	925.5
1990/10	383.0	41.1	135.3	9.2	1443.9	502.3	41.3	278.7	493.9	493.5	568.8
1990/11	16.5	1.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

units: acre-ft

Year	Ft Lyon	Las Animas	Fort Bent	Keese	Amity	Lamar Manvel	Hyde	XY Graham	Buffalo	Sisson
1986/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1986/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1986/03	588.9	60.8	15.1	5.1	122.7	34.0	1.8	12.4	19.0	0.0
1986/04	3108.5	294.1	87.1	27.3	805.2	189.9	18.3	94.7	153.4	0.2
1986/05	5115.3	490.3	146.5	45.3	1333.5	315.3	31.5	148.6	240.7	0.7
1986/06	6794.0	664.2	213.9	65.7	2017.5	454.0	48.3	263.8	414.1	1.8
1986/07	9894.9	939.9	333.5	103.7	3088.4	723.1	70.5	347.5	580.2	2.2
1986/08	8362.0	797.0	286.9	88.9	2631.4	627.3	59.4	316.1	508.3	1.8
1986/09	4058.8	425.0	120.6	40.5	993.4	255.5	20.6	172.5	228.7	0.1
1986/10	1728.6	193.8	46.0	15.8	377.2	99.6	7.3	65.0	87.7	0.0
1986/11	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1986/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/03	599.2	64.1	18.3	6.2	144.1	39.4	2.0	13.9	21.2	0.0
1987/04	3125.9	301.3	104.3	32.7	934.7	223.1	20.6	108.0	173.4	0.2
1987/05	5118.3	493.0	169.5	52.6	1517.0	362.2	34.5	168.4	273.1	0.7
1987/06	6790.5	664.6	223.5	69.1	2043.8	481.3	49.0	268.3	422.9	1.8
1987/07	9891.5	939.6	349.5	108.3	3161.6	758.2	71.2	359.8	592.7	2.2
1987/08	8361.3	797.1	292.0	90.5	2644.8	635.8	59.7	317.4	511.2	1.8
1987/09	4058.6	425.1	124.2	42.0	1002.1	265.3	20.8	175.7	235.8	0.1
1987/10	1728.7	193.9	47.2	16.2	380.5	102.3	7.4	65.8	89.6	0.0
1987/11	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1987/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

1988/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/03	609.1	65.2	20.4	6.9	162.3	43.6	2.5	17.4	27.1	0.0
1988/04	3125.5	302.9	108.7	34.1	966.5	232.0	22.2	114.6	182.7	0.2
1988/05	5214.7	505.5	177.9	55.5	1598.3	389.3	36.3	185.2	300.5	0.7
1988/06	6808.4	666.9	224.1	69.6	2032.9	489.4	48.7	269.8	426.3	1.8
1988/07	9848.8	935.7	349.5	108.1	3138.6	757.4	69.6	358.3	580.0	2.2
1988/08	8342.6	797.7	289.9	90.0	2591.5	630.0	59.0	309.0	491.1	1.8
1988/09	4072.9	413.9	127.1	42.6	1011.2	278.6	20.5	179.0	245.9	0.1
1988/10	1667.9	178.5	46.7	15.9	362.8	100.7	7.0	60.1	83.9	0.0
1988/11	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1988/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/03	602.0	64.4	20.4	6.9	162.4	39.8	2.5	17.4	24.2	0.0
1989/04	3148.6	304.2	110.2	34.2	981.3	232.4	22.4	116.8	182.4	0.2
1989/05	5196.7	497.6	174.9	55.1	1583.2	372.2	35.6	182.7	293.4	0.7
1989/06	6716.8	652.1	218.4	67.5	2003.2	467.7	47.8	262.9	412.2	1.8
1989/07	9841.9	922.3	347.0	107.6	3104.4	746.8	69.8	350.5	562.1	2.2
1989/08	8149.2	769.6	279.3	86.5	2575.3	600.4	56.3	305.2	492.0	1.8
1989/09	4089.8	415.2	124.3	41.9	1001.8	275.1	20.6	178.1	246.9	0.1
1989/10	1632.8	177.7	44.8	15.3	350.9	94.8	6.6	59.5	80.2	0.0
1989/11	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1989/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/01	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/02	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/03	595.1	62.3	20.3	6.9	162.1	38.5	2.1	17.1	24.5	0.0
1990/04	3103.3	294.1	102.7	32.2	912.7	212.3	19.7	104.4	163.3	0.2
1990/05	5165.5	497.0	176.2	54.4	1593.9	375.5	35.6	184.4	301.2	0.7
1990/06	6737.5	651.6	222.2	68.7	2032.3	473.6	48.4	265.6	413.8	1.8
1990/07	9769.5	928.9	337.5	104.4	3075.2	721.7	66.9	342.2	545.8	2.2
1990/08	8290.1	781.8	284.9	88.8	2610.6	609.0	57.8	312.9	499.2	1.8
1990/09	4076.0	412.2	123.3	42.3	1021.9	265.1	19.9	172.4	234.5	0.1
1990/10	1641.4	179.6	42.1	14.4	388.3	90.5	6.2	54.8	76.4	0.0
1990/11	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
1990/12	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix F: Computer Code for Generating Response Functions

```
///////This Program is designed to calculate Response Functions

#include <stdio.h>
#include <string.h>
#include <stdlib.h>
#include <iostream.h>
#include <math.h>
#include <malloc.h>
#include "RSP.h"

//******************************** main program ********************************

void main()
{
 int i,j;

 //***************** memory allocation
 item_hint=new char[200][180];
 GWTableElev=new double[D_Max_X][D_Max_Y];//ground water table elevation(m)
 CellAttribute=new int[D_Max_X][D_Max_Y];//cell attribute:1 for boundary cells (no flows); 2 for free water cells
 HydraulicCond=new double[D_Max_X][D_Max_Y];//hydraulic conductivity (m/day)
 Porosity=new double[D_Max_X][D_Max_Y];//porosity
 TravelTime=new double[D_Max_X][D_Max_Y];//travel time for recharge cells

 End_X=new int[D_Max_X][D_Max_Y];//final location (X) for a recharge cell
 End_Y=new int[D_Max_X][D_Max_Y];//final location (Y) for a recharge cell

 RiverReach=new int[D_Max_X][D_Max_Y];//1,2,3..represent river reach number; -1 means
 Stress=new double[D_Max_X][D_Max_Y];//water quantity of a stress on a recharge cell(unit: water units)
 ReturnFlow=new double[D_Reach][D_Day];//return flows from stress cells to a reach at a day
 ReturnFlow_Month=new double[D_Reach][D_Month];//return flows from stress cells to a reach at a month

 Path_GW_Cell_Time=new double[Max_Paths][Max_Cells];//groundwater travel time in each [path][cell]
 DrainagePortion=new double[D_Max_X][D_Max_Y];//Percentage of groundwater drained in cell(i,j)
 Path_DrainagePortion=new double[Max_Paths][Max_Cells];//Percentage of groundwater drained in [path][cell]
 Path_Ret_Days=new double[Max_Paths][Max_Outcomes];//GW return time (days) for each outcome in a GW path
 Pre_Path_Ret_Days=new double[Max_Paths][Max_Outcomes];//created for the convinience of GW return time calculation
 Path_Ret_Months=new double[Max_Paths][Max_Outcomes];//GW return time (Months) for each outcome in a GW path
 Probability_Path_Outcome=new double[Max_Paths][Max_Outcomes];//Probability of an outcome generated by a stress
 Pre_Probability_Path_Outcome=new double [Max_Paths][Max_Outcomes];//created for the convinience of outcome probability
calculation

 //***************** inititalization for variables
 for(j=0;j<D_Max_Y;j++)
 {
  for (i=0;i<D_Max_X;i++)
  {
   TravelTime[i][j]=0;
  }
 }

 for (i=0;i<D_Reach;i++)
 {
  for(j=0;j<D_Day;j++)
  {
   ReturnFlow[i][j]=0;
  }
 }

 for (i=0;i<D_Reach;i++)
 {
  for(j=0;j<D_Month;j++)
  {
   ReturnFlow_Month[i][j]=0;
  }
 }
```

```
for (i=0;i<Max_Paths;i++)
{
 Path_Stress[i]=0.0;
}

//**************** setup input/output console
output=fopen("out.txt","w");
check=fopen("check.txt","w");

//**************** read in input file
int ihint=0;
//enter grid size (m)
input=fopen("GridInformation.txt","r");
fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
fscanf(input,"%lf",&grid_size);
fprintf(check,"%s\n",item_hint[ihint-1]);
fprintf(check,"%3.2lf\n",grid_size);
//enter total x grids and total y grids
fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
fscanf(input,"%d %d",&Total_X_Grid,&Total_Y_Grid);

//** read in GW cells attributes
input=fopen("CellAttribute.txt","r");
fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
for(j=0;j<Total_Y_Grid;j++)
{
 for (i=0;i<Total_X_Grid;i++)
 {
  fscanf(input,"%d",&CellAttribute[i][j]);
 }
}

//** read in GW table elevation
input=fopen("GWTableElev.txt","r");
fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
for(j=0;j<Total_Y_Grid;j++)
{
 for (i=0;i<Total_X_Grid;i++)
 {
  if (CellAttribute[i][j]!=1)//GWTableElev is not available for boundary cells
  {
   fscanf(input,"%lf",&GWTableElev[i][j]);
   fprintf(check,"i=%d j=%d GWTableElev=%4.2lf \n",i,j,GWTableElev[i][j]);
  }
  else GWTableElev[i][j]=9999; //without this setting, run time error msg appears!
 }
}

//** read in hydraulic conductivity
input=fopen("HydrauCon.txt","r");
fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
for(j=0;j<Total_Y_Grid;j++)
{
 for (i=0;i<Total_X_Grid;i++)
 {
  if (CellAttribute[i][j]!=1)//HydraulicCond is not available for boundary cells
  {
   fscanf(input,"%lf",&HydraulicCond[i][j]);
  }
  else HydraulicCond[i][j]=9999; //without this setting, run time error msg appears!
 }
}

//** read in porosity
input=fopen("Porosity.txt","r");
fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
for(j=0;j<Total_Y_Grid;j++)
{
 for (i=0;i<Total_X_Grid;i++)
 {
```

```
    if (CellAttribute[i][j]!=1)//Porosity is not available for boundary cells
    {
     fscanf(input,"%lf",&Porosity[i][j]);
    }
    else Porosity[i][j]=9999; //without this setting, run time error msg appears!

  }
 }

 //** read in river reaches
 input=fopen("RiverReach.txt","r");
 fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
 fprintf(check,"%s\n",item_hint[ihint-1]);
 for(j=0;j<Total_Y_Grid;j++)
 {
  for (i=0;i<Total_X_Grid;i++)
  {
   if (CellAttribute[i][j]==2)//assign a river reach # only for a free water cell
   {
    fscanf(input,"%d",&RiverReach[i][j]);
    fprintf(check,"%d ",RiverReach[i][j]);
   }
   else RiverReach[i][j]=9999; //without this setting, run time error msg appears!
  }
  fprintf(check,"\n");
 }

 //** read in stress on recharge cells
 input=fopen("Stress.txt","r");
 fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
 fprintf(check,"%s\n",item_hint[ihint-1]);
 for(j=0;j<Total_Y_Grid;j++)
 {
  for (i=0;i<Total_X_Grid;i++)
  {
   if (CellAttribute[i][j]==3)//stress could only be applied on a recharge cell
   {
    fscanf(input,"%lf",&Stress[i][j]);
    fprintf(check,"(%d)",CellAttribute[i][j]);
    fprintf(check,"%4.2lf ",Stress[i][j]);
   }
   else Stress[i][j]=9999; //without this setting, run time error msg appears!
  }
  fprintf(check,"\n");
 }
 fprintf(check,"\n");

 //** read in groundwater drainage percentage on recharge cells
 input=fopen("DrainagePortion.txt","r");
 fscanf(input,"%s",item_hint[ihint]); ihint=ihint+1;
 fprintf(check,"%s\n",item_hint[ihint-1]);
 for(j=0;j<Total_Y_Grid;j++)
 {
  for (i=0;i<Total_X_Grid;i++)
  {
   if (CellAttribute[i][j]==3)//stress could only be applied on a recharge cell
   {
    fscanf(input,"%lf",&DrainagePortion[i][j]);
    fprintf(check,"%4.2lf ",DrainagePortion[i][j]);
   }
   else DrainagePortion[i][j]=9999; //without this setting, run time error msg appears!

  }
  fprintf(check,"\n");
 }
 fprintf(check,"\n");

 fclose(input);

 //**************** begin the simulation **************************
```

```
//**calculate flow path (and also end cell) for each recharge cell
int iPath=0; //set groundwater flow path =0 initially
double total_stress=0;
for(j=0;j<Total_Y_Grid;j++)
{
 for (i=0;i<Total_X_Grid;i++)
 {
  if (CellAttribute[i][j]==3 && Stress[i][j]!=0)//calculate for recharge cells only
  {
   Path_Stress[iPath]=Stress[i][j];
   FlowPath(i,j,iPath);
   iPath=iPath+1;
   //accumulate stresses on the aquifer just for checking
   total_stress=total_stress+Stress[i][j];
  }
 }
}
Total_Paths=iPath;
fprintf(check,"Total_Path=%d \n",Total_Paths);

//Calculate total # of outcome for each path
int total_outcome[Max_Paths];
for (iPath=0;iPath<Total_Paths;iPath++)
{
 total_outcome[iPath]=int(pow(2,double(Path_Total_Cells[iPath])));
 fprintf(check, "iPath= %d; total outcomes=%d stress=%3.2f \n",iPath,total_outcome[iPath],Path_Stress[iPath]);
}

//initialization for return time and return probability
for (iPath=0;iPath<Total_Paths;iPath++)
{
 for (i=0;i<Max_Outcomes;i++)
 {
  Path_Ret_Days[iPath][i]=0.0;
  Path_Ret_Months[iPath][i]=0.0;
  Pre_Path_Ret_Days[iPath][i]=0.0;
  Probability_Path_Outcome[iPath][i]=0.0;
  Pre_Probability_Path_Outcome[iPath][i]=1.0;
 }
}

//calculate return days for each path
int iCell;
for (iPath=0;iPath<Total_Paths;iPath++)
{
 int outcome=2;
 for (iCell=0;iCell<Path_Total_Cells[iPath];iCell++)
 {
  int k=0; double m=1;
  for (i=0; i<outcome; i++)
  {
   Path_Ret_Days[iPath][i]=Pre_Path_Ret_Days[iPath][k]+Path_GW_Cell_Time[iPath][iCell]*m;
   Probability_Path_Outcome[iPath][i]= Pre_Probability_Path_Outcome[iPath][k]*fabs(m-Path_DrainagePortion[iPath][iCell]);
   k=k+int(fmod(i,2)); //take the remainder of k
   m=int(fabs(m-1));
  }

  //delete the outcomes that have return days exceed Max_Return_Day
  int deleted_case=0;
  for (i=0;i<outcome;i++)
  {
   j=i+deleted_case;
   while(Path_Ret_Days[iPath][j]>Max_Return_Day)
   {
    deleted_case=deleted_case+1;
    outcome=outcome-1;
    j=j+1;
   }
   Path_Ret_Days[iPath][i]=Path_Ret_Days[iPath][j];
```

```
      Probability_Path_Outcome[iPath][i]=Probability_Path_Outcome[iPath][j];
     }

    for (i=0; i<outcome;i++)
    {
      Pre_Path_Ret_Days[iPath][i]=Path_Ret_Days[iPath][i];
      Pre_Probability_Path_Outcome[iPath][i]=Probability_Path_Outcome[iPath][i];
      //calculate return time as "month"
      if (Path_Ret_Days[iPath][i]==0) Path_Ret_Days[iPath][i]=0.001; //for convience, let 0 become a small number
      Path_Ret_Months[iPath][i]=ceil(Path_Ret_Days[iPath][i]/30)-1;
    }
    //set total_outcome for each path
    total_outcome[iPath]=outcome;
    outcome=outcome*2;
  }
 }

//print out outcomes of return days for each path
for (iPath=0;iPath<Total_Paths;iPath++)
{
 fprintf(check,"\n");
 fprintf(check,"iPath= %d \n",iPath);
 for (i=0;i<total_outcome[iPath];i++)
 {
  fprintf(check,"outcome= %d;  Return days= %3.2f; Probability= %f; Return months= %d \n",i
    ,Path_Ret_Days[iPath][i],Probability_Path_Outcome[iPath][i],int(Path_Ret_Months[iPath][i]));
 }
}

//calculate return flows as amount in reaches and months
for (iPath=0;iPath<Total_Paths;iPath++)
{
 int temp_reach=Path_Return_Reach[iPath];
 for (i=0;i<total_outcome[iPath];i++)
 {
  int temp_month=int(Path_Ret_Months[iPath][i]);
  ReturnFlow_Month[temp_reach][temp_month]=ReturnFlow_Month[temp_reach][temp_month]
    +Path_Stress[iPath]*Probability_Path_Outcome[iPath][i];
 }
}

//calculate sum of return flow units
double return_sum=0;
double return_sum_12Month=0;
for (i=0;i<D_Reach;i++)
{
 for(j=0;j<D_Month;j++)
 {
  if (ReturnFlow_Month[i][j]!=0)
  {
   return_sum=return_sum+ReturnFlow_Month[i][j];
   if (j<12)
   {
    return_sum_12Month=return_sum_12Month+ReturnFlow_Month[i][j];
   }
  }
 }
}
fprintf(check,"total stress=%f; total return flows=%f \n",total_stress,return_sum);
fprintf(output,"first_12_month_percentage= %5.4f \n", return_sum_12Month/total_stress);
fprintf(output,"total_return_flow_unit= %5.4f\n",total_stress);

//print out return flows in reaches and months
for (i=0;i<D_Reach;i++)
{
 for(j=0;j<D_Month;j++)
 {
  if (ReturnFlow_Month[i][j]!=0)
  {
   fprintf(output,"\n");
```

```
      fprintf(output,"Reach= %d Month= %d Unit= %5.4f",i,j,ReturnFlow_Month[i][j]);
     }
    }
   }

  //**print out river cell for each stress cell to return to
  fprintf(check,"\n");
  for(j=0;j<Total_Y_Grid;j++)
  {
   for (i=0;i<Total_X_Grid;i++)
   {
    if (CellAttribute[i][j]==3 && Stress[i][j]!=0)//calculate for recharge cells only
    {
     fprintf(check,"GW in cell(%d,%d) flows to river at cell(%d,%d),in reach %d taking maximum %3.1f days\n",i,j,
      End_X[i][j],End_Y[i][j],RiverReach[End_X[i][j]][End_Y[i][j]],TravelTime[i][j]);
    }
   }
  }
  fprintf(check,"\n");
  fclose(check);
  fclose(output);
 }

//************FlowPath() is used to find flow path of a given recharge cell
void FlowPath(int x, int y, int iPath)
{
 int i;

 //find gradient by from surranding 8 cells
 double gradient=0;//set initial gradient=0
 int original_x=x;
 int original_y=y;
 int next_x=x;
 int next_y=y;
 //open a big dimension of steps in whicht a recharge cell can find its free water cell
 int max_steps=D_Max_X*D_Max_Y;

 //cell_time is time (unit: days) taken to travel from cell to cell by following its gradient;
 double cell_time=0;//set initial time=0
 int iCell=0;//set iCell=0 in the begining of the "travel" loop
 fprintf(check,"\n        flow path for cell(%d, %d) is: \n",original_x,original_y);
 for (i=0;i<max_steps;i++)
 {
  Next_Cell(x,y,&next_x,&next_y,&gradient,&cell_time);
  //assign GW travel time for each flow path at each cell
  Path_GW_Cell_Time[iPath][iCell]=cell_time;
  //assign drainage portion to each flow path at each cell
  Path_DrainagePortion[iPath][iCell]=DrainagePortion[x][y];
  x=next_x; y=next_y;
  TravelTime[original_x][original_y]=TravelTime[original_x][original_y]+cell_time;
  fprintf(check,"i=%d (%d %d) gradient=%f cell_time=%3.2f day(s)\n",i, next_x,next_y,gradient,cell_time);
  fprintf(check,"***iPath=%d iCell=%d GW_Cell_Time=%3.2f DrainagePortion=%3.2f\n",iPath, iCell,
   Path_GW_Cell_Time[iPath][iCell],Path_DrainagePortion[iPath][iCell]);
  //if GW enter free water cell, record the number of cells in that path and the river reach that GW returns to
  if (CellAttribute[x][y]==2)
  {
   Path_Total_Cells[iPath]=iCell+1;
   Path_Return_Reach[iPath]=RiverReach[x][y];
   fprintf(check,"****Total_Cells=%d Return_Reach=%d \n",Path_Total_Cells[iPath], Path_Return_Reach[iPath]);
  }
  //end calculation if a free water cell is reached or gradient can't be found
  if (CellAttribute[x][y]==2||gradient==0) break;
  iCell=iCell+1;
 }
 End_X[original_x][original_y]=next_x;
 End_Y[original_x][original_y]=next_y;

// fprintf(check,"%d %d",End_X[original_x][original_y],End_Y[original_x][original_y]);
}
```

```
//*************find steepest slope
void Next_Cell(int x, int y, int* next_x, int* next_y, double* gradient, double*cell_time)
{
 *gradient=0; //set initial gradient=0

 //time for GW to travel from cell to cell by following its gradient;
 //-- Darcy V=k*gradient; time=distance/(Darcy V/porosity)
 *cell_time=0; //unit: days

 double slope;
 //compare with cell(x-1,y-1)
 if (x-1>=0 && y-1>=0)
 {
  if (CellAttribute[x-1][y-1]!=1)
  {
   slope=(GWTableElev[x][y]-GWTableElev[x-1][y-1])/(grid_size*pow(2,0.5));
   if (slope>*gradient)
   {
    *gradient=slope;
    *next_x=x-1;
    *next_y=y-1;
    *cell_time=grid_size*pow(2,0.5)/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
   }
  }
 }

 //compare with cell(x-1,y+1)
 if (x-1>=0 && y+1<=Total_Y_Grid-1)
 {
  if (CellAttribute[x-1][y+1]!=1)
  {
   slope=(GWTableElev[x][y]-GWTableElev[x-1][y+1])/(grid_size*pow(2,0.5));
   if (slope>*gradient)
   {
    *gradient=slope;
    *next_x=x-1;
    *next_y=y+1;
    *cell_time=grid_size*pow(2,0.5)/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
   }
  }
 }

 //compare with cell(x+1,y-1)
 if (x+1<=Total_X_Grid-1 && y-1>=0)
 {
  if (CellAttribute[x+1][y-1]!=1)
  {
   slope=(GWTableElev[x][y]-GWTableElev[x+1][y-1])/(grid_size*pow(2,0.5));
   if (slope>*gradient)
   {
    *gradient=slope;
    *next_x=x+1;
    *next_y=y-1;
    *cell_time=grid_size*pow(2,0.5)/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
   }
  }
 }

 //compare with cell(x+1,y+1)
 if (x+1<=Total_X_Grid-1 && y+1<=Total_Y_Grid-1)
 {
  if (CellAttribute[x+1][y+1]!=1)
  {
   slope=(GWTableElev[x][y]-GWTableElev[x+1][y+1])/(grid_size*pow(2,0.5));
   if (slope>*gradient)
   {
    *gradient=slope;
    *next_x=x+1;
    *next_y=y+1;
    *cell_time=grid_size*pow(2,0.5)/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
```

```
    }
   }
  }

  //compare with cell(x,y-1)
  if (y-1>=0)
  {
   if (CellAttribute[x][y-1]!=1)
   {
    slope=(GWTableElev[x][y]-GWTableElev[x][y-1])/(grid_size);
    if (slope>*gradient)
    {
     *gradient=slope;
     *next_x=x;
     *next_y=y-1;
     *cell_time=grid_size/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
    }
   }
  }

  //compare with cell(x,y+1)
  if (y+1<=Total_Y_Grid-1)
  {
   if (CellAttribute[x][y+1]!=1)
   {
    slope=(GWTableElev[x][y]-GWTableElev[x][y+1])/(grid_size);
    if (slope>*gradient)
    {
     *gradient=slope;
     *next_x=x;
     *next_y=y+1;
     *cell_time=grid_size/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
    }
   }
  }

  //compare with cell(x-1,y)
  if (x-1>=0)
  {
   if (CellAttribute[x-1][y]!=1)
   {
    slope=(GWTableElev[x][y]-GWTableElev[x-1][y])/(grid_size);
    if (slope>*gradient)
    {
     *gradient=slope;
     *next_x=x-1;
     *next_y=y;
     *cell_time=grid_size/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
    }
   }
  }

  //compare with cell(x+1,y)
  if (x+1<=Total_X_Grid-1)
  {
   if (CellAttribute[x+1][y]!=1)
   {
    slope=(GWTableElev[x][y]-GWTableElev[x+1][y])/(grid_size);
    if (slope>*gradient)
    {
     *gradient=slope;
     *next_x=x+1;
     *next_y=y;
     *cell_time=grid_size/(HydraulicCond[x][y]*(*gradient)/Porosity[x][y]);
    }
   }
  }

  if (*gradient==0)
   fprintf(check,"\n******************Warning!!  Gradient at cell(%d, %d) is 0!!****************\n\n",x,y);  }
```