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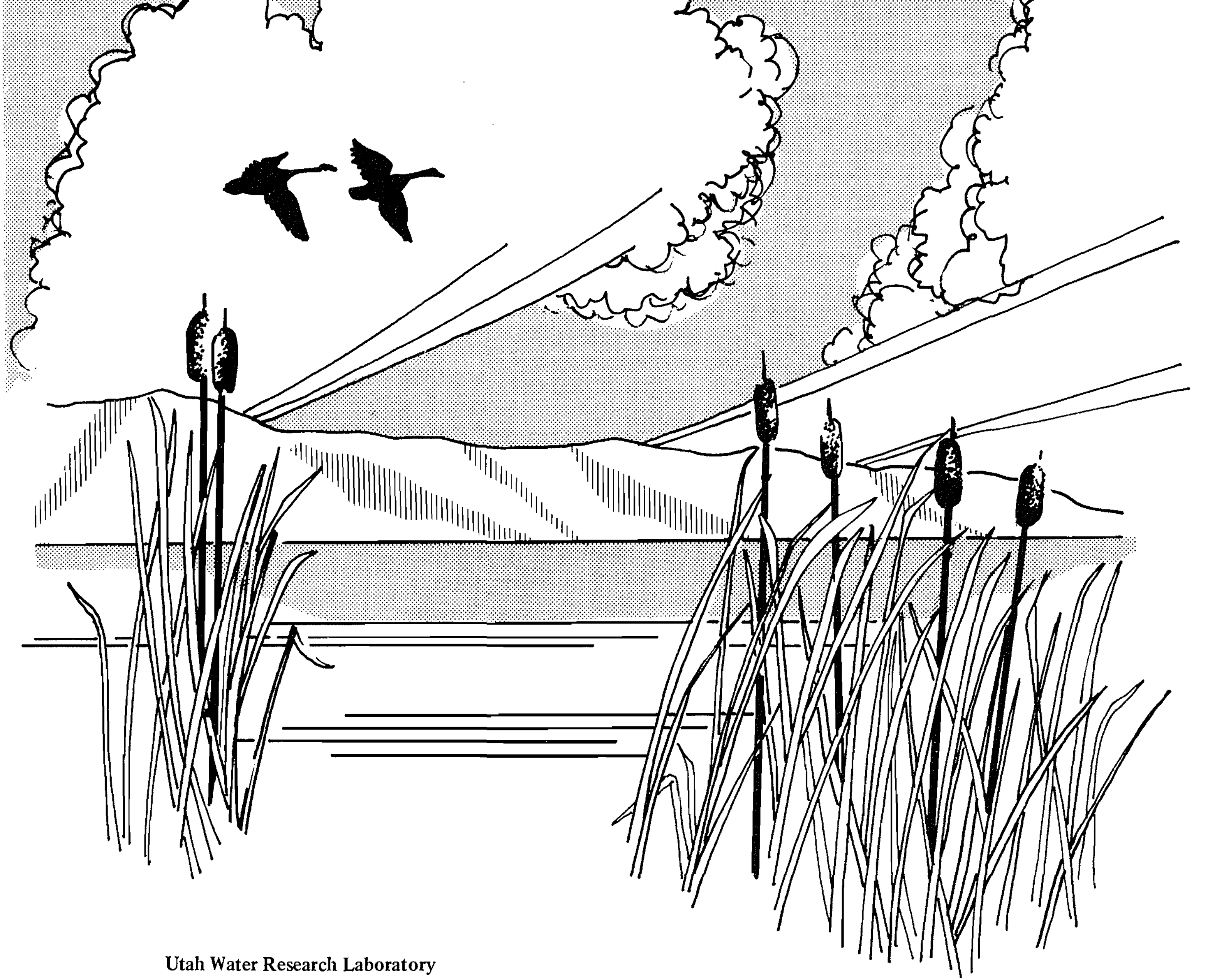

Utah Water Research Laboratory
College of Engineering
Utah State University
Logan, Utah 84322
June 1980

WATER RESOURCES PLANNING SERIES
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DEVELOPMENT OF PROCEDURES TO EVALUATE
SALINITY MANAGEMENT STRATEGIES IN
IRRIGATION RETURN FLOWS

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ABSTRACT

The salinity added by irrigation return flows is a major problem in rivers draining agricultural lands throughout the arid regions of the world, and many irrigation water management alternatives have been proposed for reducing downstream salinity problems. The merits of these alternatives, however, can only be judged from reliable information on their actual effects on the salinity in rivers receiving the drainage water and the water withdrawn from the river by downstream users. Hydrosalinity models are widely used to estimate these effects to guide the selection of a policy on management of irrigation return flows. The purpose of this research was to assess the state-of-the-art of hydrosalinity modeling in order to develop a practical management tool for predicting how the salt outflow from irrigated agriculture is affected by various farm management practices.

A review of the state-of-the-art of hydrosalinity models identified one of the major gaps in modeling as inadequate understanding and representation of the quantity and quality interrelationships between surface water, drainage water, and groundwater. Most models predict relatively constant levels of salinity over time in surface drains during the irrigation season and an increase in concentration in similar drains at other locations during the nonirrigation season.

The study also identified that a site specific equilibrium "threshold concentration" (TC) of dissolved solids can be adequately estimated and represented in a model. Salt concentration above the TC would result in precipitation of salts within the soil profile. Higher TC values would, however, exist in the unsaturated soil. Based on these new concepts, salinity in the return flows was modeled as a composite of individual component outflows from the unsaturated zones and the saturated groundwater zone, and represents the interrelationships among surface water, drainage water, and groundwater.

The model termed BSAM-SALT was tested using field data from irrigated areas in Grand Valley, Colorado, and the Circleville subbasin of the Sevier River Basin in Utah. A set of management runs was made to demonstrate the utility of the model in predicting the salt loading caused by irrigated agriculture in the Grand Valley, Colorado, area.

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CHAPTER 1. INTRODUCTION

Project Background

Rivers draining arid basins increase in salinity content as the flow moves downstream. In nature, much of the salt content of the water flowing from the mountain basins accumulates in the soil as the waters infiltrate or evaporate. When these lands are used for agriculture, the applied water leaches more salt through the soil and into the river.

As river water becomes more saline, its value in agricultural and urban uses declines. Andersen et al. (1978) found that in the Colorado River Basin, damages are becoming major as salinity levels at Imperial Dam pass 1000 ppm and now amount to approximately $20 per ton of salt removed ($200,000 for an average reduction in salt content of 1 ppm). They also found on-farm irrigation water management practices in upstream irrigated areas to be the least expensive salinity control method. Narayanan et al. (1979) also found methods to improve irrigation efficiency to be quite promising as salinity control measures.

These and the many other studies which could be cited that identify irrigation water management as important for salinity control, however, do not indicate exactly what management practices should be used. That issue has yet to be resolved through an improved understanding of how salt loadings from irrigated areas vary with farm water management practices employed. A model is needed to capture the essence of this relationship with mathematical expressions for quantitative prediction. Data are needed to calibrate such salinity-control applications. Two very important needs are a good model and good data for calibrating it.

In selecting a good model, one must recognize that the options come with wide variation in complexity of equations used, amount of descriptive input information required, and reliability of results predicted. For overall irrigation water management system design, one needs a tool that can be applied to a fairly large area without requiring an unreasonably extensive amount of data and that is accurate enough to lead to salinity control practices that work.

These river basin hydrosalinity models are powerful tools for 1) finding alternatives to existing water management procedures to improve and control salinity levels in irrigation return flows, and 2) predicting future salinity levels at critical reaches within a river system. Such predictions make it possible to assess achievements toward complying with the salinity level requirements of the Water Pollution Act Amendments of 1972 (PL 92-500).

Hydrosalinity modeling uses the modeler's present understanding of the various hydrologic and salinity transport processes that occur within an irrigated area to predict how the system as a whole will respond to changes, such as alterations to farm water management policy, that may directly affect only a few processes. Model predictions depend on the assumptions made in representing the various processes as well as on how the model portrays interactions among the processes to produce aggregate results.

Hydrologic modeling provides the basic framework for river basin hydrosalinity models, but several key processes must be added, or at least given additional emphasis. Three of the most critical are:

1. The chemical reactions and interactions that occur as waters containing varying combinations of salt species move, in an irregular time pattern, through soils having a variety of chemical properties.

2. The respective sources and degrees of mixing among surface runoff, natural groundwater, and irrigation return flow. In hydrologic modeling one simply adds these flows and does not need to be so careful in getting the correct mixture between them because errors can offset. In hydrosalinity modeling, the correct ratios among flows originating from various sources are important to portraying the chemical reactions correctly. Since gaged stream flow data seldom indicate flow sources, this requirement poses major difficulties for model verfication.

3. Salt pick up in effluent groundwaters is site specific depending upon the presence of residual salts and the extent of mineral dissolution in the groundwater zone. It is important to represent well the processes controlling salt pickup as well as those controlling the depositing and subsequent repeat

pickup of salts within the channel as flows rise and fall.

These issues represent some of the more important which must be reviewed in assessing the state-of-the-art of hydrosalinity modeling before applying it to evaluate any specific management strategy.

Purpose

The hydrosalinity models available in the literature vary in the degree of complexity used to represent the chemical reactions that occur within the soil-water system. Some models such as those described by Dutt et al. (1972) are detailed dynamic models which attempt to portray many complex phenomena that occur within the soil-water system including nitrogen transformations. At the other extreme of soil-water-salinity modeling are the steady state TDS models based on the conservation of mass principle formulated by many researchers including Terkeltoub and Babcock (1971).

The complex hydrosalinity models need an enormous amount of data, much of which is usually not available in the real world, and large amounts of computer time. The simple models, based on steady-state conditions, require less data but provide less detail. For a particular application, a model must be chosen from an appropriate point within this range. The key issues in the selection are:

1) The reliability, in terms of the desired use, of the simpler steady-state models.

2) The additional reliability achieved for the desired use with the complex hydrosalinity models.

3) The possibilities for upgrading a relatively simple hydrosalinity model to predict the impact of management practices on the salinity of return flows without having to collect data for the other features of the more complex models. This model should be capable of utilizing minimum field data.

The purpose of this research was to assess the state-of-the-art of hydrosalinity modeling in representing the salt pickup process within the soil-groundwater system. The assessment is geared to developing a practical management tool for predicting how the salt outflow from irrigated agriculture is affected by various farm water management practices.

Scope

In order to achieve the purpose of the study, the research was directed to the following tasks.

1) A review of pertinent hydrosalinity models, soil-water system models, and groundwater models with particular reference to the model capabilities in representing the various processes controlling salt and water flows in an irrigation water supply system, assumptions in the models, and modeling gaps, if any.

2) An inventory and analysis of field data from selected irrigated areas so as to have information available for verifying quantitative relationships proposed to depict salt pickup mechanisms.

3) The development of a relatively simple hydrosalinity model and a demonstration application based on calibration with field data from selected irrigated areas.

4) Demonstrative application of the model to examine the effects of alternative irrigation water management policies on return flows and salt loadings.

CHAPTER 2.
REVIEW OF PROCESSES
CONTRIBUTING TO SALINITY
IN RIVER SYSTEMS

Introduction

In order to quantify how the salinity management options would affect the stream flow quantity and quality in a particular river basin, the physical processes controlling salt and water movement within the basin would have to be identified, evaluated, and represented in the management model. The model would have to represent these processes sufficiently well to indicate how they would respond to the range of alternatives. The following review of chemical processes that occur in soil water systems, mineral weathering in the groundwater system, and salt loading in streams, provides the theoretical background for an overview of the available hydrosalinity models discussed later in this report.

Processes occurring in the soil water system

Levels of soil water salinity vary by location and over time in irrigated soils with differences in the quality, quantity, and application patterns of irrigation and natural waters and in the chemical, geohydrologic, and biological properties of the soil.

Salt pickup from the soil varies with interrelated physical, chemical, and biological factors. The physical factors include the soil type, quantity of water percolating through the soil, travel path within the soil profile, depth of groundwater table, and the hydraulic gradient causing the flow. The salt transport depends upon mass of flow, ionic diffusion, and dispersion in the soil. Chemical factors are the partial pressure of CO_2, complex inorganic and organic chemical reactions involving ion exchange, ionic adsorption, dissolution and precipitation, and formation of complex ion pairs changing the ionic strength of water and the concentration gradient. Biological activity depends on the extent and nature of microorganisms and their substrate present in the soil.

All in all, many physical processes interact in determining soil salinity levels, and the dominating ones vary greatly with local conditions. Generally, however, the three major processes are 1) precipitation of the salt content of the water within the soil or dissolution of the salts in the soil by the water with the two principal salts being $CaCO_3$ (calcium carbonate) and $CaSO_4 \cdot 2H_2O$ (gypsum), 2) concentration of salts within the soil water as the water is lost by evapotranspiration, and 3) spatial and temporal differences that impart spatial and temporal variability to the salt content of the soil water.

Precipitation of $CaCO_3$ and gypsum in the soil. Both the qualitative and quantitative aspects of the salt precipitation phenomena occurring within the soil profile under various conditions of leaching have been described (Willardson et al. 1979 and Swarez and Rhoades 1977). Willardson et al. (1979) showed that chemical precipitation occurs in the soil profile during cycles of evaporation and water additions which reduce the effluent salt content below what one would expect theoretically from the leaching fraction (LF).

The data in Table 2.1 indicate that when irrigation water is applied, $CaCO_3$ and gypsum precipitate in the soil in varying amounts depending upon the leaching fraction, depth within the root zone, and the type of irrigation water. The precipitation of salts in the root zone was less when the applied irrigation water was initially undersaturated with $CaCO_3$ as compared with the salt precipitation when the applied water was saturated with $CaCO_3$. Swarez and Rhoades (1977) contend that salt is deposited in the soil with high leaching (LF=0.4) because $CaCO_3$ is dissolved in the upper layers and is deposited in the lower layers of the root zone, although there is no net deposition in the soil.

Concentration of salts within the soil-water system. The salt content of the soil-water system is concentrated through consumptive use by irrigated crops and phreatophytes. The amount of increase in concentration depends on the LF, the type of soil, and the quality of the applied water (Table 2.4). Reduced leaching would: 1) increase precipitation of $CaCO_3$ and $CaSO_4$ in the soil, 2) reduce soil mineral weathering and dissolutions of salts previously deposited in the soil, and 3) increase the amount of soluble salt in the soil profile because less salt would be returned in the drainage water.

At high leaching fractions, salt is added to percolating waters, passing through

Table 2.1. Precipitation of $CaCO_3$ and gypsum in soil. (After Swarez and Rhoades 1977.)

Di-ver-sion (1)	Quar-ter depth (2)	Type 1 Water			Type 2 Water			Type 3 Water					
		LF = 0.1,	LF = 0.4		LF = 0.1	LF = 0.4		LF = 0.1		LF = 0.4		0.1 LF-0.4 LF	
		$CaCO_3$ (3)	$CaCO_3$ (4)	$\Delta CaCO_3$ (5)	$CaCO_3$ (6)	$CaCO_3$ (7)	$\Delta CaCO_3$ (8)	$CaCO_3$ (3)	Gypsum (4)	$CaCO_3$ (5)	Gypsum (6)	ΔCaCO (7)	Δ Gypsum (8)
2	1	-15.01	-24.39	9.38	-5.49	-13.67	8.18	-5.14	0	-12.82	0	7.73	0
	2	6.74	4.40	2.34	6.88	4.38	2.50	5.97	0	3.84	0	2.13	0
	3	5.35	2.53	2.82	5.47	2.41	3.06	4.18	18.63	1.94	0	2.24	18.63
	4	3.33	1.25	2.00	3.49	1.26	2.23	2.22	30.58	0.99	0	1.23	30.58
5	1	-13.04	-12.98	-0.06	-5.49	-14.02	8.53	-5.22	0	-12.30	0	7.08	0
	2	6.72	4.39	2.33	6.86	4.46	2.40	5.91	0	3.44	0	2.47	0
	3	5.28	2.49	2.79	5.52	2.40	3.12	3.94	30.65	1.83	0	2.11	30.65
	4	3.11	1.25	1.86	3.57	1.18	2.39	2.21	30.19	0.92	0	1.29	30.19
7	1	-10.89	-12.90	2.01	-5.71	-14.24	8.54	-5.40	0	-11.84	0	6.44	0
	2	6.62	4.37	2.25	6.87	4.25	2.62	5.96	0	3.33	0	2.63	0
	3	5.08	2.43	2.65	5.61	2.35	3.26	3.66	42.26	0.72	50.08	2.94	-7.82
	4	2.83	1.25	1.58	3.67	1.18	2.49	2.21	29.53	0.17	32.30	2.04	-2.77
9	1	-6.72	-11.96	5.2	-6.23	-15.4	9.17	-5.75	0	-11.95	0	6.20	0
	2	6.16	4.11	2.01	7.00	4.35	2.70	6.08	1.57	1.60	89.63	4.48	-88.04
	3	4.26	2.23	2.03	5.83	2.29	3.54	3.12	59.92	0.19	64.29	2.93	-4.37
	4	1.93	1.18	0.75	3.85	1.14	2.71	2.25	27.72	-0.13	32.42	2.38	-4.70

Note:

(1) Values are in metric tons (x 10^{-3}) per project (25,000 acres) in the rootzone; includes in-situ dissolution and subsequent precipitation.

(2) Negative values indicate dissolution; quarter depth is 1 ft.

(3) TYPE 1 water: River initially under saturated with $CaCO_3$ (Feather River)
TYPE 2 water: River initially saturated with $CaCO_3$ (Rio Grande River)
TYPE 3 water: River initially saturated with $CaCO_3$ and nearing saturation with gypsum (Pecos River)

(4) Diversion numbers in column 1 indicate consecutive downstream irrigation projects. The irrigation waters vary in salinity among these points, as the drainage flow is completely returned to the river below each project and the resulting water was reequilibrated before it arrived at the next downstream project.

(5) Composition of soil water is contained in Swarez and Rhoades (1977).

the root zone, as a result of silicate mineral weathering and dissolution of soil lime (Rhoades et al. 1974). Depending on the drainage of the soil-water system, the above processes may redistribute the storage of salts within the soil profile and develop, over long periods of time, a high capacity for "buffering" salts. For soil types such as those encountered in Vernal, Utah (King and Hanks 1975 and Melamed 1975), a characteristic soil solution concentration profile emerged. A unique salt concentration developed at each position in the soil profile, and it remained nearly unchanged as waters of various concentrations passed through.

Salt balance studies based on outdoor experiments with lysimeters (Rhoades et al. 1973 and 1974) indicated that, depending on the LF, there could be either a net gain of salts (by dissolution), and a net loss of salts (by precipitation). Both calcarious (with 1% $CaCO_3$) and noncalcarious Pachappa sandy loam soils were investigated (Figures 2.1 through 2.4). The fact that these lines representing drainage water salinity tend to become horizontal with large leaching fractions suggests that for these soils a threshold leaching fraction (LFT) could be identified beyond which there is neither a net gain nor a net loss of salt relative to the applied water. The estimated drainage water composition, "Threshold Salt Concentration" (TSC), corresponding to the LFT for several irrigation waters is shown in Table 2.2. Since the data represent tile drain water or the drainage waters as they emerge from the root zone, the above estimates of EC_T values correspond to the concentration of the soil-solution at the bottom of the root zone. The estimated values of TSC for noncalcarious (without lime) soils are higher than the corresponding TSC values for calcarious soils.

TSC values are highly site specific and depend on the quality of irrigation water, soil type, and the leaching status of the soil at the time of determination. For example, soils recently brought into irrigation may have a higher TSC than do soils irrigated for many years. Field experiments of the type described in Rhoades et al. (1973) provide in situ TSC for the usual soil types encountered in most parts of the western states where the soils provide a buffering capacity with respect to $CaCO_3$. Further investigations are necessary to extend this concept to other soils. Nevertheless, the TSC concept is useful for deriving a simple but reasonably reliable hydrosalinity model as described in a subsequent chapter.

Spatial and temporal variability of drainage water concentrations. Observed salinity concentrations in drainage waters from various irrigated areas change with the time and location of individual drains. The results of a number of research projects shed light on these patterns.

1. The eight studies of drainage water salinity concentrations reported in Table 2.3 indicate that concentrations remained similar throughout the year, suggesting that salt outflow was proportional to the corresponding water flow. Those observations appear,

Table 2.2. Estimated threshold salt concentrations (TSC) for Pachappa sandy loam. (Based on Figures 2.1, 2.2, 2.3, and 2.4.)

	River type	Estimated LF at which no salt precipitation or dissolution occurred	Estimated Threshold Salt Concentration, TSC (mmhos/cm)	
			calcarious lysimeters (with 1% $CaCO_3$)	noncalcarious lysimeters
1	Feather	-	-	-
2	Grand	0.285	2.75	
		-		-
3	Missouri	0.17	3.75	
		0.185		4.25
4	Salt	0.185	7.2	
		0.145		10.5
5	Colorado	0.235	4.25	
		0.225		4.10
6	Sevier	-	-	-
7	Gila	0.230	11.75	
		0.235		12.25
8	Pecos	-	-	-
		-	-	-

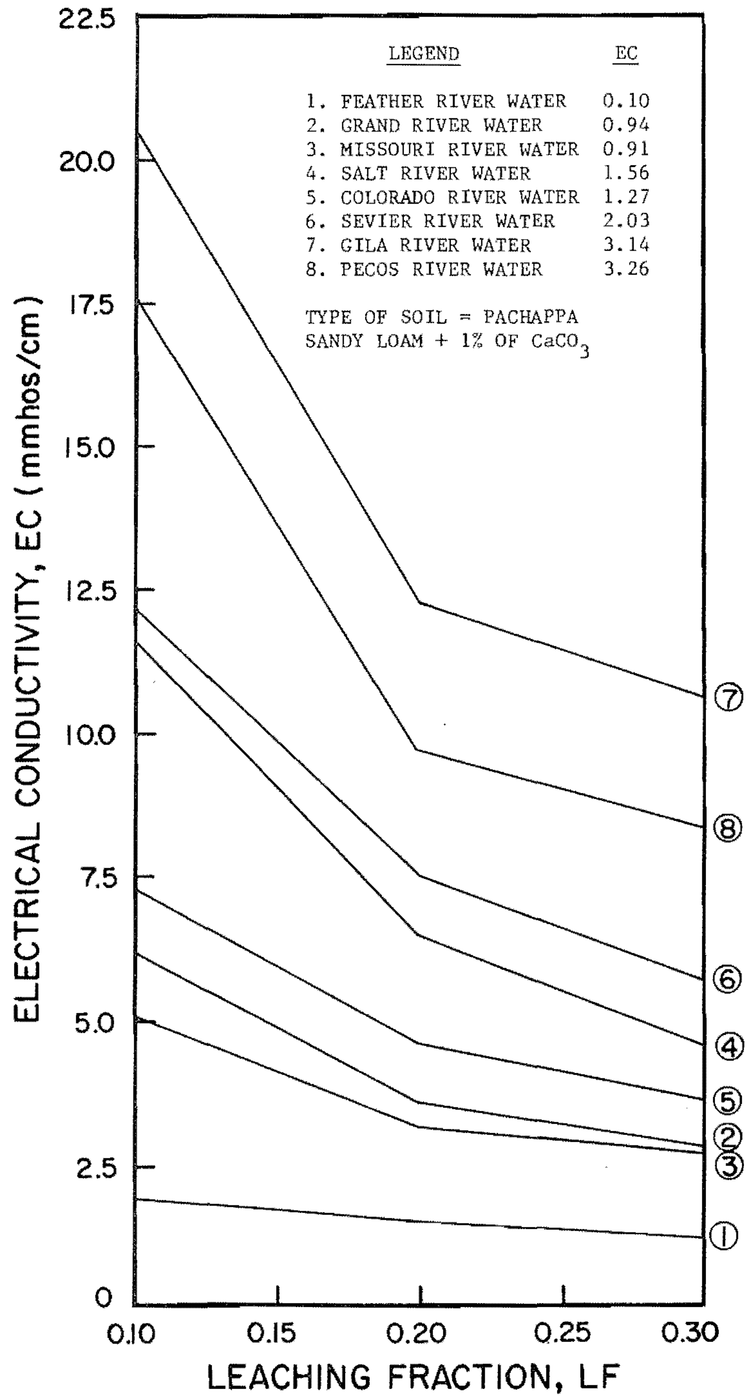


Figure 2.1. Average composition of drainage waters from calcarious lysimeters, spring of year. (After Rhoades et al. 1973.)

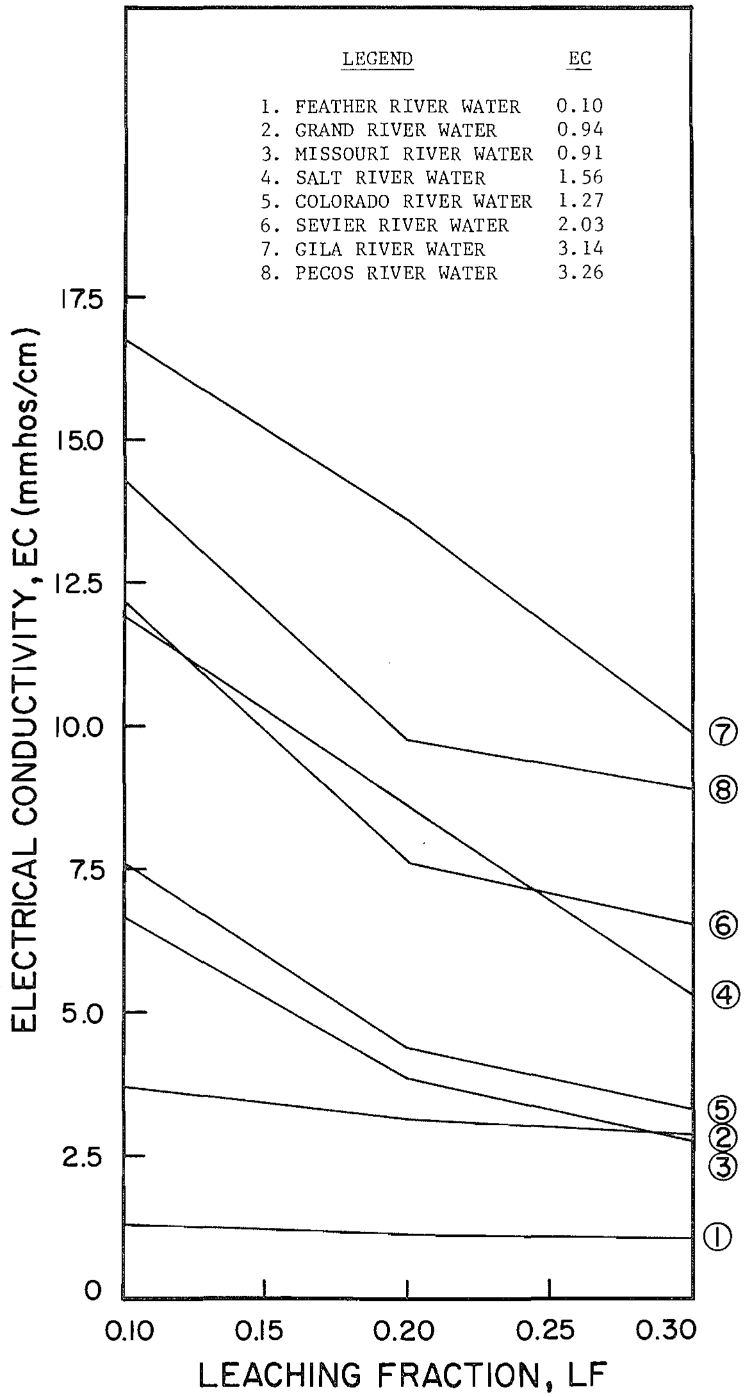


Figure 2.2. Average composition of drainage waters from noncalcarious lysimeters, spring of year. (After Rhoades et al. 1973.)

however, to be site specific. As evidence, concentrations in the La Mesa drain in the Massilla Valley, California, (Figure 2.5) have a cyclic trend with an EC lower during the summer than in the winter months. A possible explanation for this phenomena would be that the drain could still be leaching the residual salts and had not reached steady state conditions.

2. The drainage water salinity concentration time patterns in three drains in the Palo Verde Irrigation District (Figure 2.6), vary considerably from one another. While the outfall drain showed little variation in concentration (measured in tons/ac-ft) with time, the Anderson drain showed large fluctuations. Two possible causes exist, 1) Anderson drain could still be leaching residual salts (in a pattern varying with the rate of irrigation) or 2) the amount of effluent groundwater mixing with the drainage waters could be higher and diluting the drainage water during the low flow season.

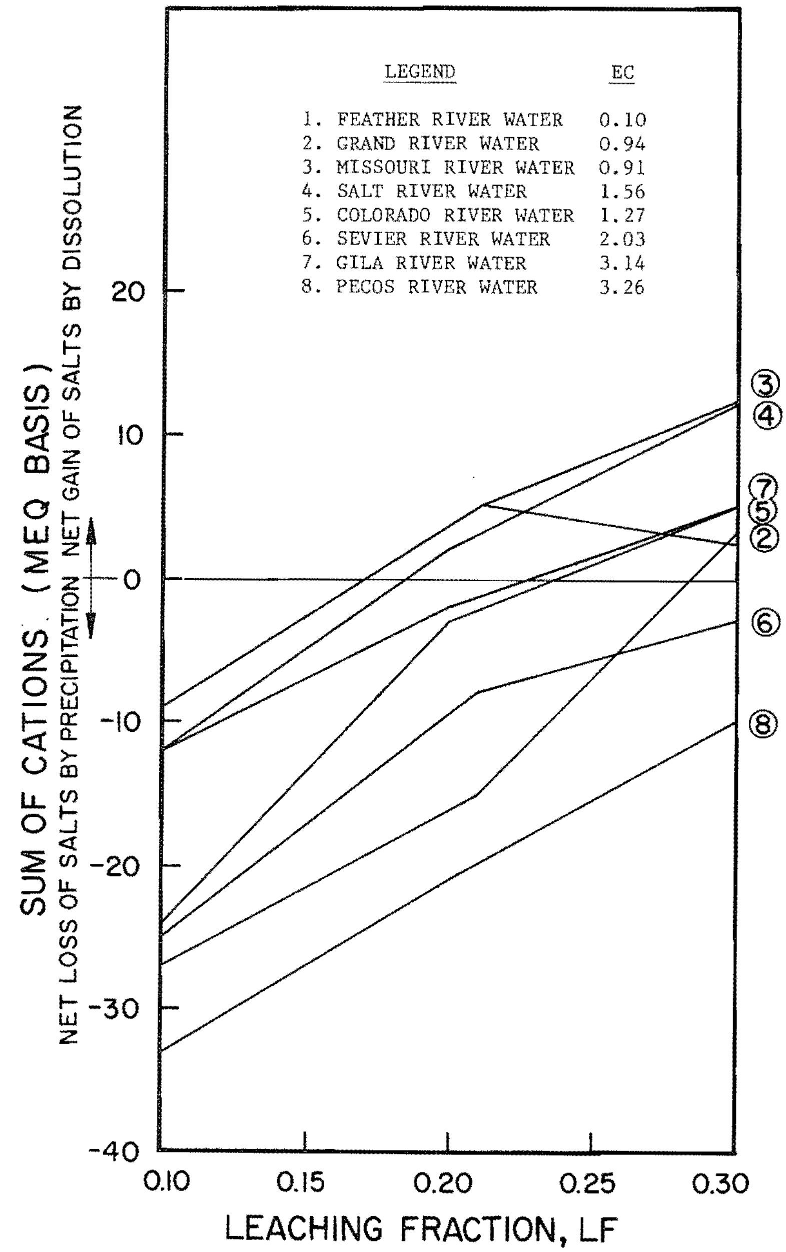


Figure 2.3. Net contribution of mineral weathering and salt precipitation expressed as percent of salts applied in irrigation waters in calcarious soils. (After Rhoades et al. 1974.)

3. Flows in the open drains in the Uintah Basin exhibit a high degree of spatial variability. When the flow of water reduced sharply to 0.5 - 2.0 gpm, however, there was a sharp increase in the EC, suggesting an important base flow contribution to total salinity in the drains. The scatter of the concentrations in the larger flows (Figure 2.7) suggests that one should consider the possibility of using spatial variability in TSC values when modeling a basin like the Uintah.

4. A plot of salt outflow versus total drainage outflow for the Grand Valley drains (Figure 2.8) displays considerable scatter. The pattern, however, generally fits an hypothesis that groundwaters contributing about 20 tons/day per cfs (7420 ppm) mixes with variable amounts of surface and irrigation water causing the scattered points to the right. The 7420 ppm is the approximate slope of an envelope curve on the left side of the plot. From these trends, two hypotheses provide reasonable assumptions for modeling the salt loading of drainage flow.

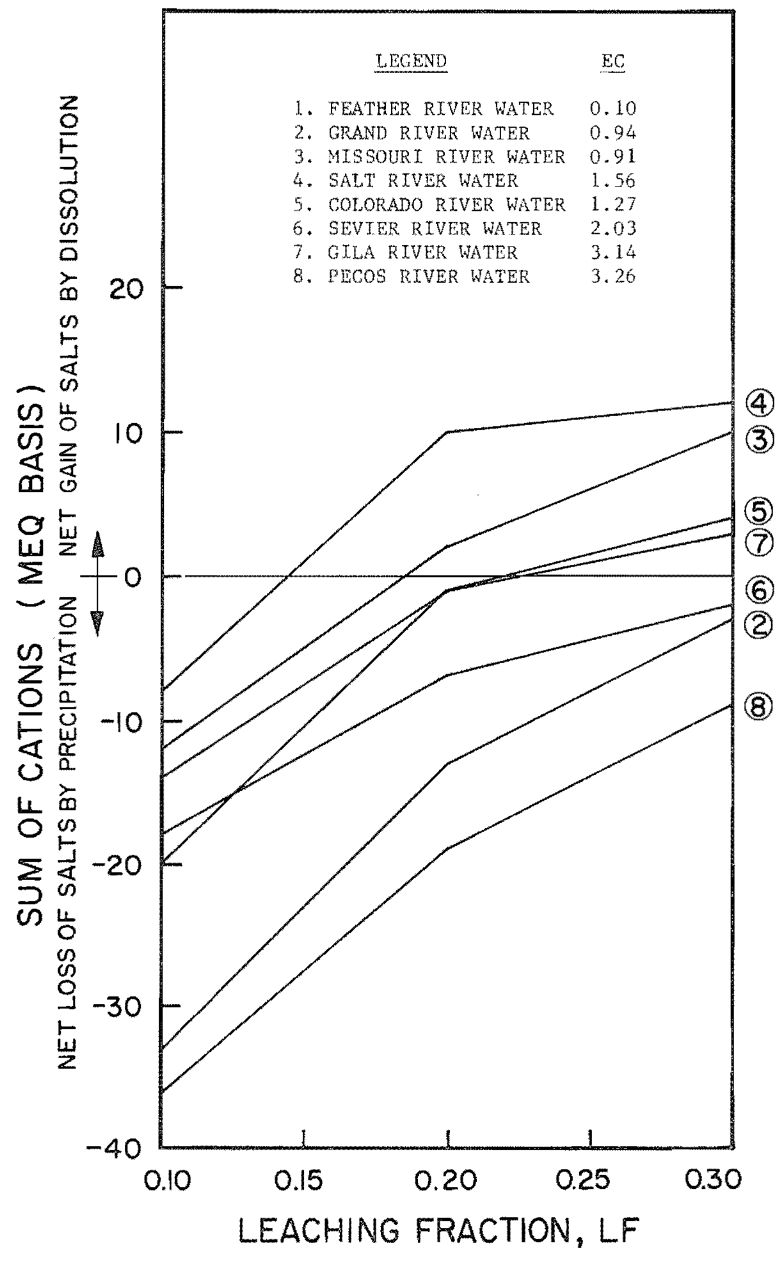


Figure 2.4. Net contribution of mineral weathering and salt precipitation expressed as percent of salts applied in irrigation waters in noncalcarious soils. (After Rhoades et al. 1974.)

Table 2.3. Results of studies of drainage water salt concentrations.

Reference	Period of data	Crops grown	Soil type	Depth of drains	Results	Location
Ayars, J. E. 1976	1975	Alfalfa, wheat, and crested wheat grass	Grand Valley soils	Approximately 3 meters	Salt concentration of the leachate at the bottom of the soil profile is independent of the volume of water. TDS profile at the beginning and end of growing season show the concentration of salt in the profile below the root zone relatively constant. This region acts as a buffer and causes salt concentration of the return flow to be relatively constant, i.e. reduction in salt loading is directly proportional to reductions in the volume of return flow.	Grand Valley, Colorado
Brown, Kirk W. et al. 1978	1974 and 1975	Rice	Beaumont clay soil (Typic pelludert)	Surface runoff	Floodwater was not percolating through the soil profile. The salts were evidently adsorbed and not readily solubilized, i.e. the soil served more as a sink than a source. Year / Irrigation Technique / Salt in irrigation water Kg/ha / Salt in return flow Kg/ha 1974 Impounded 528 559 1974 Continuous 993 575 1975 Impounded 428 433 1975 Continuous 712 587.9 (i) Since more salt-bearing water is added to the continuous flow plots than is removed in the outflow during the growing season, it is apparent that this management practice could lead to excess salt in the soil during years which do not receive much rainfall between growing seasons (ii) Concentration of salts in the outflow from the plots were less and the water would more easily meet rigid water quality standards.	Texas
Carter, et al. 1979	2 years (1973-74)	Not Cropped	Port neuf silt loam	Variable	(1) Once residual salts are removed by irrigation, there is no rapid salt accumulation to high concentrations from dissolving minerals, when lands are no longer irrigated. (2) The quantity of salt outflow from soils, after residual salts are removed, depends almost entirely upon the amount of leaching water passing through the soil. With less water passing through the soil, smaller	Twinfalls, Idaho

Table 2.3. Continued.

Reference	Period of data	Crops grown	Soil type	Depth of drains	Results	Location
Carter (Cont.)					quantities of salt dissolved from soil minerals and slightly soluble salts will be removed by leaching water. (3) There was no measurable buildup in soils from mineral dissolving for at least 10 years after irrigation was terminated.	
Chichester, F. W. et al. 1979	May 1974-April 1976	Pasture and meadow	Silt loam	To 130 cm	Chemical concentrations remained similar throughout the year. Salt quantity is proportional to the flow of water.	Coshocton, Ohio
Sammis, T. W. and C. M. Hohn, 1977	1976	Wheat, tomatoes, cotton, lettuce, alfalfa		2 - 2.5 meters	Negative correlation existed between water quality in the drain and flow rates. flow (m^3/sec) — EC (mmhos/cm) summer 0.8-0.9 — 1.3 winter 0.25 — 2.0	Messilla Valley, N.M.
Swarez, D. L. and J. D. Rhoades 1977	Hypothetical river waters			4 ft 120 cm	(i) The concentration of the soil waters at the bottom of the root zone (assumed at 4 ft depth) were lower for all river types with high leaching. The salt load in return flow, however, reduced with low leaching. (ii) The data showed that the concentrations of the drainage waters and salt loads all increase with distance downstream. (iii) Salt was deposited in the soil even with high leaching because $CaCO_3$ was dissolved in the upper quarters and was deposited in the lower quarters, even though there was no net deposition in the soil with respect to that derived from irrigation water per se. (iv) Reduced salt load in return flows may or may not reduce salt load in the river waters. The reduction depends on the degree of saturation of river water with respect to $CaCO_3$ and gypsum.	
USDA. SCS Salinity report Jan. 1979	April 1976 to October 1977	Pasture, alfalfa, small grain, and corn for silage	Variable. Marine shales to gravelly terraces, and glacial outwash materials.	7 - 8 ft	In most of drains EC remained nearly constant even though the flow of water doubled. This indicates relatively constant solubility of salts in these soils. However, where the flow of water reduced sharply to 0.5 - 2 gpm there was sharp increase in the EC.	Uintah Basin, Utah

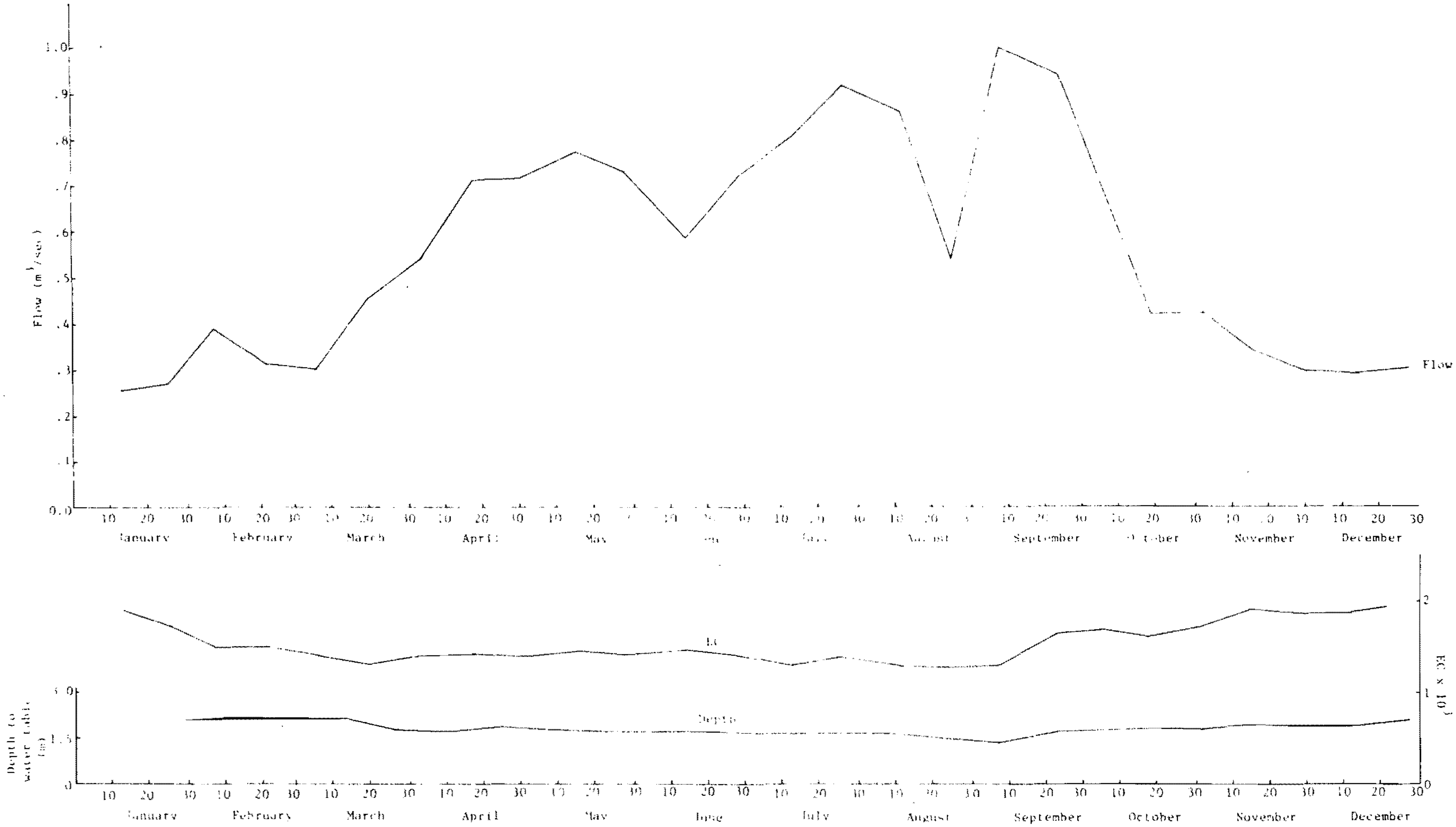


Figure 2.5. Flow rates and water quality in the La Mesa drain along with the associated groundwater depth fluctuations. (After Sammis and Hohn 1977.)

a. The surface and irrigation drainage water have a constant salt concentration or, as it may be equivalently stated, the salt loading is proportional to the flow.

b. Concentrations increase with low flows because of less mixing to dilute the relatively poor quality effluent groundwater. Some Grand Valley drains show concentrations as high as 20,000 ppm during low flows.

Mineral weathering in the groundwater system

The salts within the groundwater zone are picked up more freely because of mineral weathering. The weathering rate is influenced by the soil moisture level, quantity of percolating water, composition of the parent material underlying the groundwater zone, and the salinity of the soil water. All these factors should logically be included in quantitative analysis of salt pickup in the soil where the two principal processes are the weathering (dissolution) of residual salts within the soil water - groundwater system and the resolubilization of previously precipitated salts (reverse weathering) within the soil layers.

Salt loading in streams

The major processes contributing to salt loading between two stream points are:

1. Instream salt pickup as it varies with: a) seasonal flow fluctuations including associated larger fluctuation in sediment movement, b) manmade fluctuations in flow associated with diversion and return flow patterns, and c) the salt content of the river water. Flow fluctuations that expose

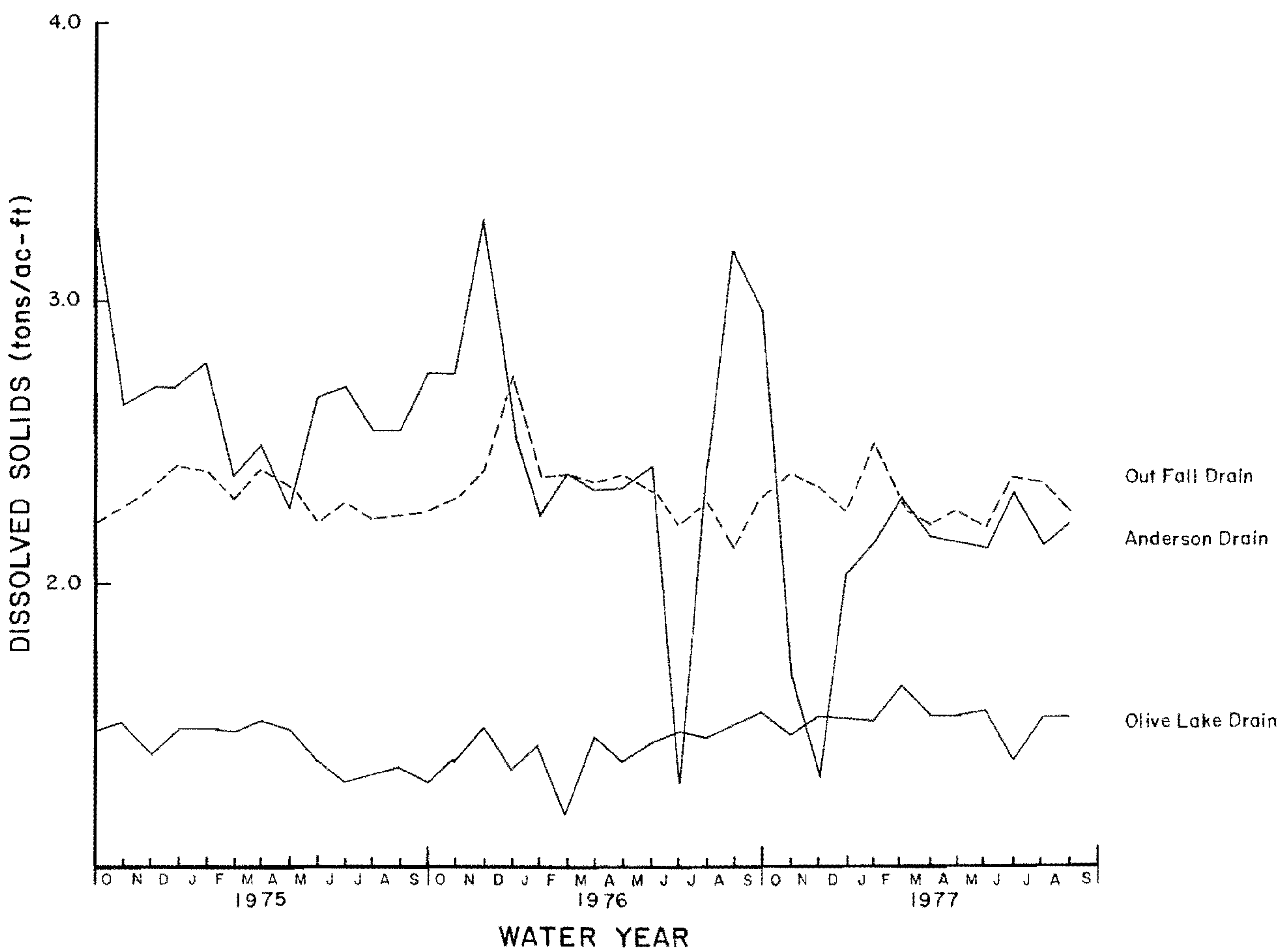


Figure 2.6. Dissolved solids concentration in the drains of the Palo Verde Irrigation District, California, 1975-1977 water years.

stream bed areas cause near surface water to rise to the surface and evaporate leaving appreciable deposits of salt along river or canal banks or bottoms.

2. Natural salt pickup from the contributing drainage area (overland process).

3. Salt loading from irrigated agriculture. The flow components are canal seepage, deep percolation, and field drainage. Salt loadings from field drainage depend on the soil water system as previously discussed.

Empirical salt loading equations

Two empirical methods were tried for representing overall salt loading from natural and agricultural sources.

Method 1: A linear regression model fit to empirical data (Fifield 1979) represented the incremental salt loading between any two points of a stream by:

$$\Delta \text{load} = m\,(\Delta \text{ flow}) + b \quad . \; . \; . \; . \; . \quad (2.1)$$

in which

Δ load = stream loading, applied either on the basis of an individual constituent or total dissolved solids (TDS), and m and b are regression coefficients.

The slope of the regression line, m, represents the net average concentration of the constituent ion or TDS. Equation 2.1 was used to estimate stream loading by "un-

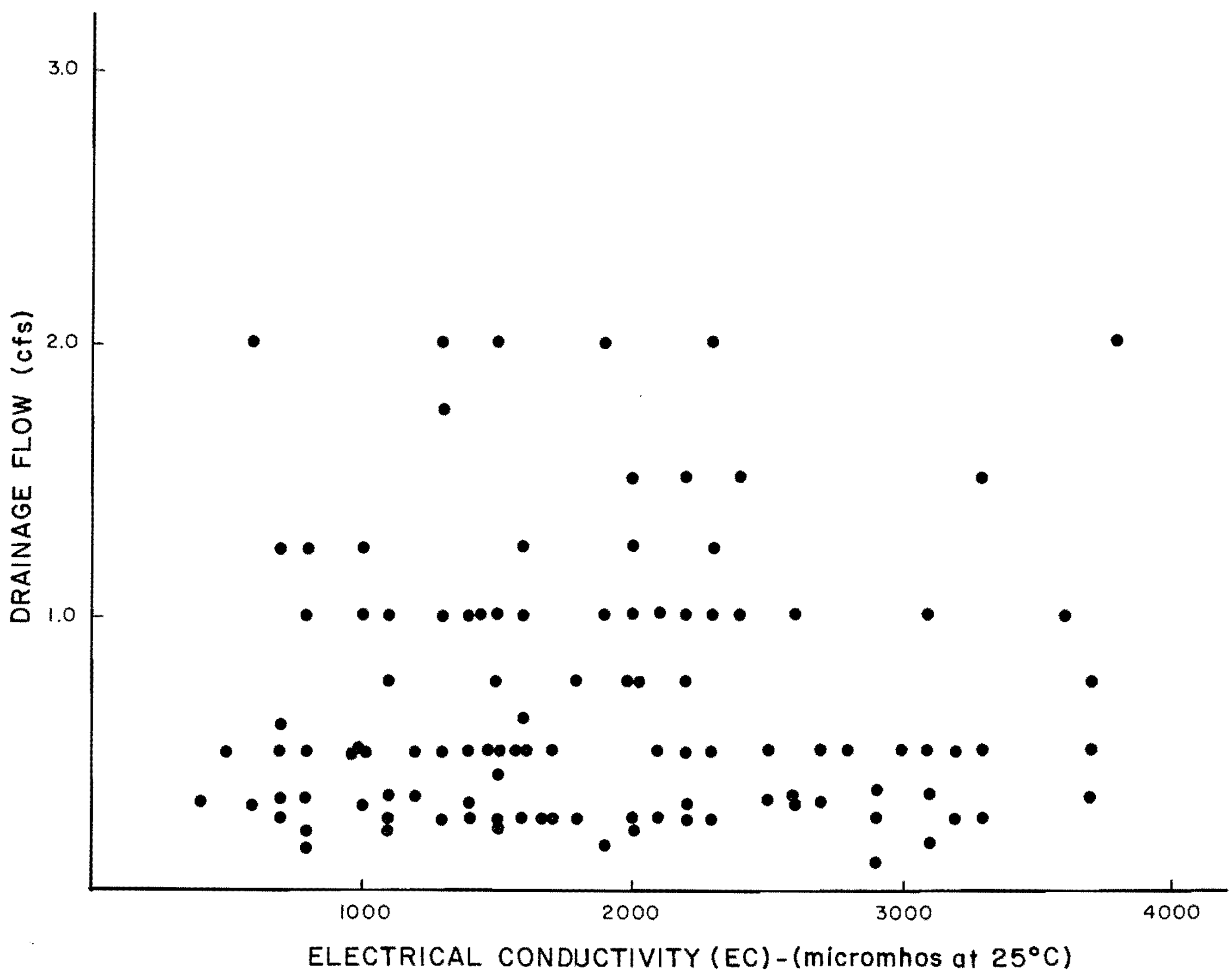


Figure 2.7. Drainage flow versus EC in the Uintah Basin drains (1976).

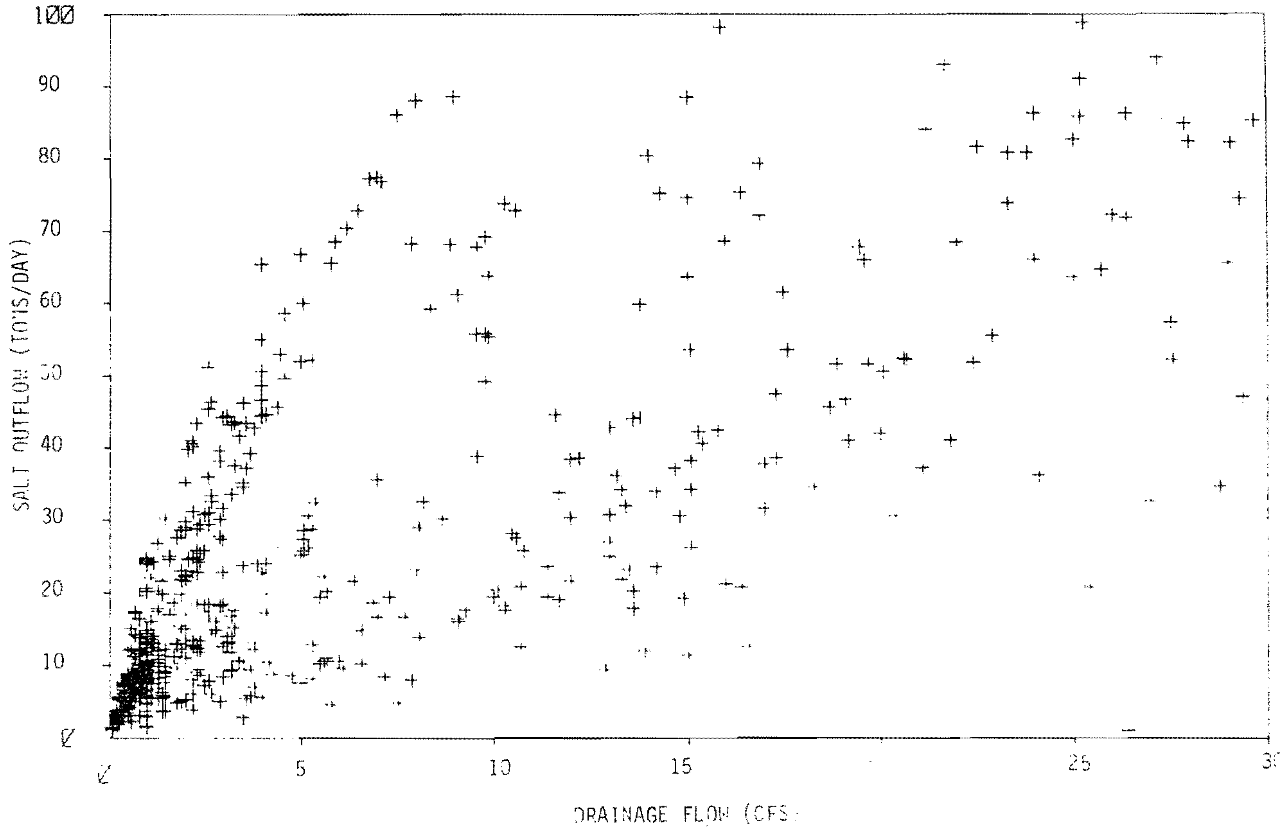


Figure 2.8. Salt outflow plotted as a function of drainage flow for 35 drains in the Grand Valley, Colorado, subbasin. (U.S. Bureau of Reclamation Grand Valley Salinity Control Project data.)

Table 2.4. Trends in changes of soil salt loading with river water type and leaching fraction. (After Swarez and Rhoades 1977.)

Type of river water	Changes in salt loading
Rivers undersaturated with $CaCO_3$.	Net removal of $CaCO_3$ from solution under low leaching (LF = 0.1) Net dissolution of $CaCO_3$ under high leaching (LF = 0.4)
Rivers saturated with $CaCO_3$.	Lost $CaCO_3$ by precipitation in the soil root zone under low leaching, and by precipitation in the river channel (after remixing the drainage water with undiverted river water) under high leaching. The total amounts of precipitation and the river compositions were uneffected by irrigation management.
Rivers saturated with $CaCO_3$ and nearing saturation with gypsum.	Lost substantially more salts by precipitation under low versus high leaching. LF = 0.1: Precipitation of salts was relatively constant through successive valleys. LF = 0.4: Initial valleys showed no gypsum precipitation, but subsequent valleys showed sharp rise in gypsum precipitation.

accountable" sources, both ungaged point and nonpoint sources in a model successfully applied to four agricultural basins, namely, Grand Valley and Arkansas River in Colorado, Palo Verde Irrigation district in California and Yakima Valley in Washington.

Method 2: Improved estimation of salt pickup was attempted by using separate estimations of loadings by salt source and summation in the form:

$$T_{load} = t_{instr} + t_{nat} + t_{ag} + t_{ps} + t_{ups} \quad \ldots\ldots\ldots (2.2)$$

in which

T_{load} = total salt loading between any two points of a stream.

T_{instr} = incremental salt loading due to instream salt dissolution and precipitation phenomena.

t_{nat} = Incremental salt loading due to natural (diffuse) sources.

t_{ag} = Incremental salt loading due to agricultural (diffuse) sources.

t_{ps} = Incremental salt loading due to known point sources.

t_{ups} = Incremental salt loading due to unknown point sources.

Salt loading by instream processes is important for storm or other short-period flows, but loading and deposition balance out over longer periods. If one wishes to model over long periods and can identify all the significant point sources, Equation 2.2 reduces to

$$T_{load} = t_{nat} + t_{ag} + t_{ps} \quad \ldots\ldots (2.3)$$

Of the three terms on the right side of Equation 2.3, the diffuse agricultural loading can be summed from its sources:

$$t_{ag} = S_{SRT} + S_{ART} + S_{GEF} + S_{SPL} + S_{TW} - S_{CNL} \quad \ldots\ldots\ldots\ldots (2.4)$$

in which

S_{SRT} = Salt in seepage returns.

S_{ART} = Salt in agricultural return flow from deep percolation.

S_{GEF} = Salt in effluent groundwater.

S_{SPL} = Salt returned to stream through administrative and operational spills.

S_{TW} = Salt returned to stream through tail water runoff from irrigated fields.

S_{CNL} = Salt taken from the stream in canal diversions.

Known point source loadings can be estimated from data on the sources. Riley and Jurinak (1979) postulated that the total salt load added within a subbasin could be apportioned between natural and agricultural sources on the basis of average quantity of water that was estimated to flow through the soils of each area:

$$\frac{S_n}{S_a} = \frac{W_n}{W_a} = \frac{\Delta Q + ET_{ag}}{W_d - ET_{ag}} \quad \ldots\ldots\ldots (2.5)$$

in which

S_n = rate of salt loading from natural sources.

S_a = t_{ag}.

W_n = rate of drainage from natural lands.

W_a = rate of drainage from irrigated lands.

ΔQ = change in measured water flow.

W_d = rate of water diversions for irrigation within subbasin.

ET_{ag} = evapotranspiration of water diverted for agriculture.

The salt loading from natural sources can then be estimated from Equation 2.3 since all terms but T_{nat} are now known.

CHAPTER 3.
REVIEW OF HYDROSALINITY MODELS

Introduction

Recent advances in the state-of-the-art of hydrosalinity modeling have been significant as many computer simulation models have been developed to predict salinity in return flows. Walker (1978) and Fifield (1979) have provided preliminary evaluations of different categories of models. Lewis (1976) demonstrated an evaluation procedure by application to many water quality models, some of which include conservative constituents. The above evaluations, however, did not state the assumptions made in the representations of salt pickup phenomena in the reviewed models. Since irrigation waters usually undergo significant quality deterioration in passing through the soil profile, the capability of a model can be better understood by evaluating 1) the basic assumptions in the model, and 2) the representations of the various processes that occur within the soil profile, that affect the salinity status of the soil-water. The review of hydrosalinity models discussed in this section focuses on these assumptions and representations as well as the criteria listed in Table 3.1.

Only deterministic models that portray the hydrology and chemistry of soil-water and groundwater regimes were reviewed. Available models predict water salinity (TDS and/or constituent ion) for specific situations and represent a wide range of capability and applicability. There exist many other models and versions of models, but those reviewed are representative of the range of those currently available for hydrosalinity modeling.

General Description of Models

Two major categories of hydrosalinity models exist, namely, 1) the simple water and salt budget models and 2) the more complex models that also depict chemical processes within the soil-water system.

Water and salt budget models

Lane (1975) and Dixon (1978) provide an extensive discussion of these mass balance models. A brief description of characteristics of eight pertinent models is outlined in Table 3.2. Both one-dimensional instream models and two-dimensional river basin models have been developed assuming steady state conditions and considering TDS as a conservative constituent. Neither the chemical reactions nor the precipitation/dissolution of salt within the soil profile is considered in these models, which generally operate on a monthly time increment.

Models depicting chemical processes

These models synthesize both hydrologic (water flow) and chemical (solute flow)

Table 3.1. Criteria for evaluating salinity management models.

1. Model Capabilities
   - Applicable situations
   - Constituents modeled
2. Model Assumptions
   - Within root zone
   - Within the unsaturated zone below root zone
   - Within the saturated groundwater zone
3. Salt Pick Up Methodology
4. Representation of Groundwater Salt Component
5. Data Requirements
   - For model inputs
   - Additional. for model verfication
6. Model Costs
   - Initiation costs
   - Utilization costs
7. Model accuracy
   - Representation of physical system
   - Numerical accuracy
   - Sensitivity to input errors
   - Sensitivity of management options
8. Ease of Application
   - Adequacy of available documentation
   - Output form and content
   - Updateability of data decks
   - Modification of source decks
9. Model Credibility
   - Adapted research areas
   - Adapted practical locations

Table 3.2. Characteristic analysis of water and salt budget models.

Reference	Constituents modeled	Major assumption(s) in salinity model	Input data requirements	Time increment	Comment	Model Application
Hill, R. W. et al. 1973	TDS	Salt quantities leaving the soil profile were estimated by attaching the soil effluent concentration to any deep percolation water, determined from the soil moisture system. Deep percolation salt and water were routed through the same delay network before appearing in the groundwater system. Salt concentration in soil solution (TDS) is estimated by an accounting process at the end of each model time period, considering the salt from weathering as well.	TDS data for inflow/outflow streams.	monthly	Soil-salt process is simulated in a gross manner. The model essentially provided a link between the quality of applied water and subsurface return flow quality.	Bear River Basin, Utah
Hyatt et al. 1970	TDS	Salt balance exists within each subbasin. Deep percolation water percolating through the groundwater basin assumes a salt load by the groundwater concentrations. As with the hydrologic system, the input functions to the salinity system within an area are acted upon by the routing and storage functions of the system. Because dissolved solids are now degradable, the continuity of mass principle also applies to dynamics of flow within the salinity system. Salt content in the ungaged component of stream flow has constant concentration.	Salinity (TDS) data associated with stream flows, diversions, returns.	monthly	Salt loading from sediment is not represented	Upper Colorado River Basin
Jensen, A. R. 1976	TDS	Ionic composition of TDS is constant over all months and all tributaries, or that no chemical reactions or precipitation occur. (TDS is taken to be a conservative substance).	All the flow quantities and the respective TDS concentration.	monthly	The model predicts instream salinity based on statistical analysis.	Colorado River

Table 3.2. Continued.

Reference	Constituents modeled	Major assumption(s) in salinity model	Input data requirements	Time increment	Comment	Model Application
Jensen (cont.)		TDS output of each tributary basin can be adequately modeled using stream flow and TDS data recorded at the outflow gaging station of each subbasin. In stream TDS from late summer to late fall is derived entirely from groundwater flow. Ratio of groundwater flow component to the total flow is a function only of the total flow. No hysteresis phenomena is considered.				
Lewis, L. Delong 1977	TDS	Increase in flow tends to decrease the concentration (TDS) TDS concentrations and stream loads can be estimated from stream flow records using a regression model derived from chemical analysis. Constant year-round relation is assumed in the regression procedure.	Stream flow data and TDS concentrations	monthly	The model is one dimensional instream model. It does not identify salinity contribution from selected sources per se.	Green River Basin, Wyoming
Melamed, J. P. 1975	TDS (EC)	Precipitation and dissolution processes within soil profile are represented by source-sink term in the model. There is a particular concentration in the soil solution where neither precipitation nor dissolution occurs. Ion exchange is of minor importance; total salinity (TDS) was considered. Ion exchange capacity is relatively uneffected by salinity levels generally encountered in the area.	Experimental set-up of soil columns, and leaching water of variable quality	Fraction of a day	Computations using a constant parameter source-sink term for the whole soil profile improved the prediction of total salts in the entire profile, but still predicted very poorly the salt distribution with depth.	Laboratory analysis and field testing of soils from Vernal, Utah
Ribbens, Richard W., 1973	TDS	Flow and salinity are routed through the river system.		1 month	One dimensional instream model	Colorado River

Table 3.2. Continued.

Reference	Constituents modeled	Major assumption(s) in salinity model	Input data requirements	Time increment	Comment	Model Application
Ribbens (cont.)		Time dependent interactions of physical processes such as soil-water transformations due to an irrigation regime must be specified by the user. TDS parameter is considered to be conservative. Mass balance concepts are employed. So chemical precipitation, dissolution and reactions of individual constituents are not considered. Complete mixing of solute is assumed. Flow is assumed to be independent of quality, although quality depends on flow. Salinity computations are based on mean flow weighted concentrations.				
Utah Water Research Lab 1975 (SALT)	TDS	Under steady state conditions the model conducts flow and mass balance on a river system. It is a one dimensional instream model.	Water and salt flows, user options.	Annual	Unlike a dynamic model, this model does not show peak values which might appear for short periods of time after a change in conditions or periodic variations in existing conditions.	Colorado River Basin
Walker, W. R. 1970	TDS	Mass balance of water and salt must occur during the model time. Salt flows are assumed to depend on water flows. Salt obtained from an irrigated area is assumed to be the quantity of salt outflow which must occur to maintain annual salt balance minus quantity of salts indicated by groundwater outflows with measured salinity concentrations.	TDS data for inflow and outflow.	monthly	Root zone salt analyses require updating.	Grand Valley, Colorado

processes. The representation of the hydrology component varies substantially depending upon the watershed studied and the desired application. The state-of-the-art of modeling soil-water systems is found in the models outlined in Table 3.3 and described by Oster and Rhoades (1975).

The hydrologic and chemical components are normally modeled separately but integratedly. Huber et al. (1976) describe the Basin Simulation and Assessment Model (BSAM), a generalized hydrologic model that can be applied to any watershed and also can be coupled to a suitable salinity submodel. As classified in Figure 3.1, the complexity of solute flow models ranges from simple applications of plate theory assuming piston-flow movement of solute and water (Tanji et al. 1967), to detailed models which attempt to represent the complex chemical reactions within the soil profile by use of both hydrodynamic dispersion and diffusion principles.

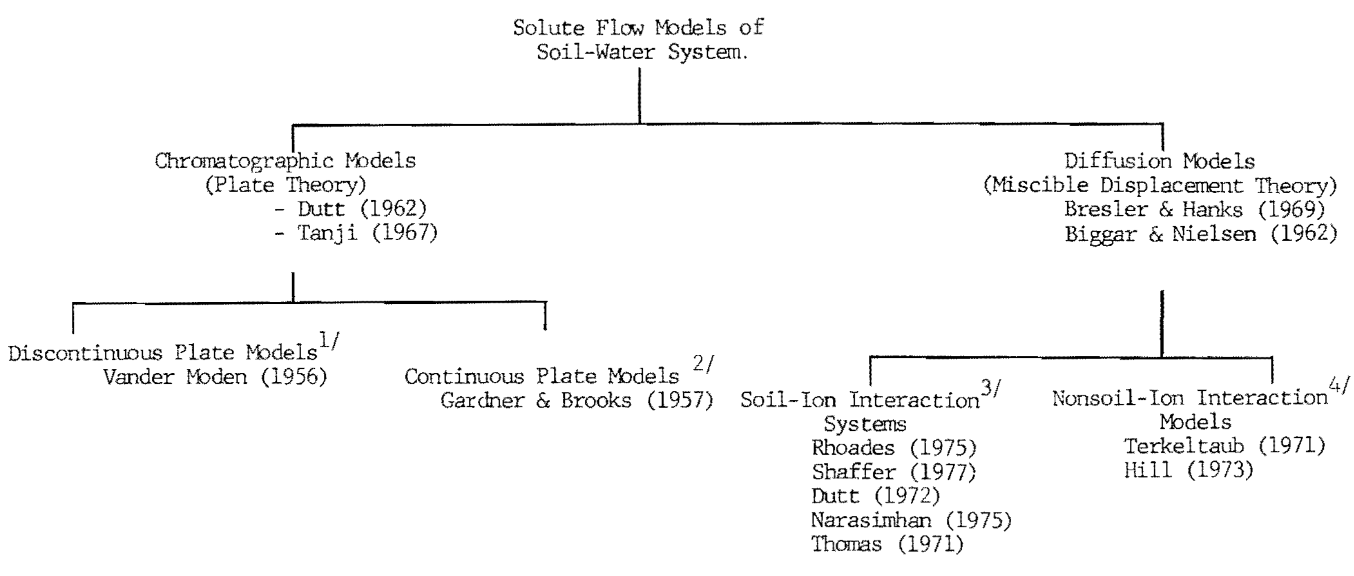
Models integrating solute transport with soil-water chemistry were initially pursued by Tanji et al. (1967), Thomas et al. (1971), Dutt et al. (1972), and Narasimhan (1975). Descriptions of specific characteristics of some more recent and more refined models are outlined in Table 3.4.

Models attempting to also integrate groundwater flow came even later. The first models (Konikov and Bredehoeft 1974) considered reactions within the water but not chemical reactions between the water and the aquifer. As a result, they did not rigorously define the relation of groundwater salinity to overlying soil salinity. Helweg and Labadie (1976) computed groundwater salinity (represented by TDS) by means of a regression equation using the electrical conductivity (EC) of the soil water and groundwater. Outlined in Table 3.5 are the general characteristics of these groundwater salinity models.

Assumptions in Salinity Modeling

Solute flow processes

Review of the models presented in Tables 3.2, 3.3, 3.4 and 3.5 indicated that solute flow is generally modeled from a few basic relationships. The principle of conservation of mass was generally used, and steady state conditions were commonly assumed. The only exception is the model of Willardson et al. (1979), which assumed transit state conditions in modeling the water and solute flows.



Note:
1. Solution remains in an affective plate of column until equilibrium with solid phase is obtained.
2. Based on rate theory and ionic equilibrium.
3. Equilibrium concentration of constitutent ions, ion pairs and cation exchange considered.
4. No chemical reactions are considered.

Figure 3.1. Development categories of selected solute flow models of the soil-water system.

Table 3.3. Characteristic analysis of soil-water system models.

Reference	Constituents modeled	Major Assumptions	Input data Requirements	Results	Model Applications
Oster, J. D. and J. D. Rhoades, 1975	Ca^{2+}, Mg^{2+}, Na^{+}, SO_4^{2-}, HCO_3^{-}, SAR, EC	Soil profile in steady state with irrigation water. pH of the solution was assumed to be governed by soil carbonate equilibria. Soil lime ($CaCO_3$) was assumed present in sufficient quantity to saturate the soil solution. Drainage waters were assumed to be in equilibrium with 0.13 atm of CO_2, the average partial pressure determined at the 80-cm depth in lysimeter studies.	Ionic composition of irrigation water, and drainage water, leaching fraction, appropriate equilibrium constants, pH, and PCO_2 within the soil profile.	(1) Adequacy of the computer model for predicting the drainage water composition was demonstrated by the EC, total salt burden, SO_4^{2-} and SAR. (2) Correlations between measured and calculated concentrations of Ca^{2+}, Mg^{2+}, and HCO_3^{-} were less satisfactory. (3) Measured Ca^{2+} ion concentrations tended to be greater than calculated values, whereas reverse was the case with Mg^{2+}. (4) Both salt precipitation and mineral dissolution significantly affect the EC of the drainage water, with relative magnitudes varying with the irrigation water. (5) The extent of salt pickup or loss was dependent upon composition of the irrigation water, leaching fraction, and the partial pressure of CO_2.	Study conducted with lysimeters filled with Pachappa soil and irrigated with eight typical western U. S. waters.
Oster, J. D. and B. L. McNeal, 1971	Individual ions, and EC.	Equilibrium concepts with respect to concentrations of constituent ions. Each micro equivalent of OH^- generated or consumed in a reaction for each unit of CEC (meq/100 gm), would change the soil pH, in the pH range studied, by 0.01 pH unit.	Composition of soil-saturation extract data, CEC of soil, percent of water at saturation and at field water content, and the estimated partial pressure of CO_2.	Model 1: Used Debye-Hückel equation and considered precipitation, of salts, CE, soil pH buffer capacity, ionic activities, SO_4, HCO_3 and CO_3 pairs of Ca, Na, Mg, and partial pressure of CO_2. Model 2: Same as above, except extended Debye-Hückel equation was used. Model 3: Included only ion pair of $CaSO_4$.	Data from Southwest Irrigation field station at Brawley, California

Table 3.4. Characteristic analysis of detailed hydrosalinity models.

Reference	Constituents modeled	Major Assumptions in salinity model	Input data requirements	Time increment	Comments	Model Application
Ayars, J. E. 1976	Ca^{+2}, Mg^{+2} Na^{+}, HCO_3^{-}, Cl^{-}, SO_4^{-2} TDS	Soluble species move freely with soil segment. Solute concentrations are constant for any soil segment. Nitrogen transformations are not considered (as it is not a major pollutant in Grand Valley). Inorganic reactions are based on equilibrium chemistry since the reaction times are less than the residence time of water in a soil segment. Water flow and content are independent of any chemical process. Complete mixing occurs at each increment in time and space. Each chemical process is independent of other processes over a time step with respect to availability of component masses. Rate of change of mass for each component is constant over a time step. Mixing cell concept is used to calculate salt transport. The length of cell remains constant.	Irrigation water chemical analysis. Number, sizes and depth of chemistry horizons in soil profile. Initial soil analysis of each horizon. Fertilization and irrigation dates. CEC, concentration of gypsum in soil, presence of lime.	Daily	TDS concentrations were adequately modeled, but the individual ionic constituent concentrations were not. $CaSO_4$ - $CaCO_3$ - $Ca(HCO_3)_2$ system was not adequately modeled for the soils in the Grand Valley.	Grand Valley, Colorado
Khan, I. A. and Labadie, J. W., 1979	Individual ions, TDS and SAR of the irrigation return flows.	The unsaturated zone model assumes: (i) steady state conditions, (ii) the pH of the solution is governed by the soil carbonate equilibria, (iii) soil lime is present in sufficient quantities to saturate soil solution, (iv) complete mixing is assumed. Temperature effects are not considered.	Average ionic concentration of irrigation water, soil-water, and groundwater.	Annual	The management strategy should be based on the tacit assumption that at least a portion of available groundwater is still usable for agriculture.	Lower San Luis Rey River Basin, Southern California

Table 3.4. Continued.

Reference	Constituents modeled	Major Assumptions in salinity model	Input data requirements	Time increment	Comments	Model Application
Khan (cont.)		The saturated zone model assumes a mass-transport equation which includes both convective transport and dispersion, with the latter assumed proportional to the concentration gradient. Chemical reactions and ion exchange in the saturated zone are ignored.				
Shaffer et al. 1977	NO_3^-, NH_4^+ Ca^{2+}, Na^+, Mg^{+2}, HCO_3^-, Cl^-, CO_3^- and SO_4^{2-}	Unsaturated flow Solutes contained in the water move with the water into and/or from adjacent segments. Each cell is spatially homogeneous, and solutes transferred into a segment or cell are mixed with those already in the cell. Saturated flow Water moves by piston displacement through successive segments until it reaches the drain. Lateral dispersion and diffusion are ignored. Kinetic approach is used for nitrogen transformation. Equilibrium conditions exist for salt or inorganic chemistry. Drain effluent No chemical reactions occur after the mixing process in the drains.	Many physical and chemical parameters are required.	0.001 day to 0.1 day	It may be uneconomical to collect necessary field data and make model runs. The model simulates many important physical and chemical processes which have been identified and expressed in equation form. Rate equations for nitrogen transformations are statistical and may not be valid outside derivation data set. User may not by completely aware of all the options in the model.	Vernal, Utah, Grand Valley, Colorado
Willardson, Lyman S., R. J. Hanks and J. J. Jurinak, 1979	Ca^{+2}, Mg^{+2} Na^+, K^+, Cl^-, $SO_4^=$, HCO_3^-, SAR, Θ_v	Salt moves with water. As the volume of water in the soil is decreased, the salt concentration increases and chemical precipitation may occur. Chemistry model assumes that (1) Soil contains lime. (2) Each soil depth increment	Variable depending whether SALT FLOW I, SALT FLOW II, or SALT FLOW III is selected.	one month	Model is designed for calcarious soils. Model capabilities include: a) Salt can be moved through the soil profile without chemical reactions with the soil (SALT FLOW I). b) Salt can be moved in combination with chemical precipitation and	Ashley Valley Northern Utah

Table 3.4. Continued.

Reference	Constituents modeled	Major Assumptions in salinity model	Input data requirements	Time increment	Comments	Model Application
Willardson (cont.)		is constant. (3) Each soil depth increment is an open system with respect to CO_2 exchange with soil atmosphere. (4) Henry law constant is independent of temperature and salt concentration. (5) Cation exchange capacity (CEC) is a constant for a given soil, independent of pH, ion type and concentration. (6) $CEC = X_{ca} + X_{mg} + X_{na} + X_{k}$. (7) $HCO_3 : CO_3$ ratio remains constant.			dissolution (SALT FLOW II). c) Same as above plus cation exchange equilibrium reactions. (SALT FLOW III) d) Model can handle transcient moisture flow.	

Table 3.5. Characteristic analysis of groundwater-salinity models.

Reference	Constituents modeled	Major assumptions in quality model	Input data requirements	Time increment	Comments	Model Application
Helweg J. Otto and John W. Labadie 1976	TDS	The groundwater is still reusable. The concentration of drainage water was based on the leaching formula. TDS is computed from EC measurements via regression equation: TDS = -2 + 0.683 EC (mg/ℓ) (micromhos/cm) Maximum allowed difference in concentration between the drainage water and groundwater was specified (DCON) as a basis for a management strategy. There is some way of ultimately removing salts from the basin, such as by pumping downstream groundwater into an outfall. If the basin is closed then a salt sink area must be identified. There were no sources or sinks of salts in the unsaturated zone of soil profile.	Annual average TDS in irrigation water, average groundwater concentration level for the beginning of the model period, and DCON.	Annual	The model is named accelerated salt transport (ASTRAN) method. It is used to obtain least-cost alternatives for distributing water over the basin, by adjusting the parameter DCON to produce a desired degree of salinity control.	Bonsall Subbasin in the San Luis Rey River Basin, California
Konikov, F. Leonard and John D. Bredehoeft 1974	TDS in groundwater	No chemical reactions occur between the water and the aquifer or soil materials that affect the dissolved solid concentration. The movement of dissolved solids due to hydrodynamic dispersion is proportional to the concentration gradient. Source or sink term is added to adjust the groundwater concentration. Dissolved solids concentrations in the flow leaving a stream cell was calculated from principles of mass balance. TDS concentration in recharge water was adjusted such that	Hydrologeologic data, observation well network, specific conductance of surface and groundwater, TDS	monthly	Dispersion equation is solved to describe the chemical concentration in the groundwater system.	Arkansas River Valley, SE Colorado

Table 3.5. Continued.

Reference	Constituents modeled	Major assumptions in quality model	Input data requirements	Time increment	Comments	Model Application
		increase in concentration in recharge water is proportional to the decrease in volume due to ET.				
Konikov, L. F. and J. D. Bredehoeft 1978	Solute concentration	1. Darcy's law is valid and hydraulic-head gradients are the only significant driving mechanism for fluid flow. 2. The porosity and hydraulic conductivity of the aquifer are constant with time, and porosity is uniform in space. 3. Gradients of fluid density, viscosity, and temperature do not affect the velocity distribution. 4. No chemical reactions occur that affect the concentration of the solute, the fluid properties, or the aquifer properties. 5. Ionic and molecular diffusion are negligible contributors to the total dispersive flux. 6. Vertical variations in head and concentration are negligible. 7. The aquifer is homogeneous and isotropic with respect to the coefficients of longitudinal and transverse dispersivity.	Boundary conditions, aquifer characteristics, stresses	variable (minutes to years)	The program solves two simultaneous partial differential equations, one is the groundwater flow equation, and the other is the solute transport equation. The assumption must be carefully evaluated before applying the model to a field problem.	A Theoretical model, demonstrated with analytical solutions to idealized problems.

Other principles or assumptions used in the more sophisticated models were chemical equilibrium, spatial homogeneity, complete mixing of the solutes, insignificant lateral dispersion and diffusion, presence of soil lime, constant pH at each depth increment, constant CEC for a given soil, insignificant temperature effects, and no salinity change from chemical reactions or ion exchange in the saturated zone. The above assumptions were common to many of the models, but there was considerable variation in the level of detail used in representing the soil-water-chemistry of the system.

A few general assumptions are inherent in the one-dimensional instream salinity models based on statistical analyses. Constant salinity concentration in the source groundwater was assumed by Pinder and Jones (1969). Fairly constant proportions of constituent ions were assumed by MacKinhan and Stuthmann (1969) and Jensen (1976). These models did not allow for the possibility of salt pickup.

Jensen (1976) made the two assumptions on p. 10 in stating that stream salinity concentration c can be represented by:

$$c = C_g \ (Q_T/Q_o)^B \quad \text{for } Q_T > Q_o \qquad (3.1)$$

and

$$c = C_g \qquad \text{for } Q_T \leq Q_o \quad . \ . \ . \quad (3.2)$$

in which

C_g = groundwater TDS concentration.

Q_T = total runoff.

Q_o = river flow at which the groundwater component is greatest.

B = an exponent varied between -1.0 and 0.0 for the Colorado River system.

Because data were lacking for estimating natural TDS concentration, Jensen (1976) assumed that the increase in salinity concentration over the period of record was the result of water depletions rather than increases in salt loading. As this assumption has not been verified, his results should be interpreted carefully.

Surface groundwater interrelationships

As water moves slowly in the groundwater aquifers, a long time might be required for the deep percolating (DP) water to emerge as effluent flow. Since the percolating water carries dissolved salts, mixes with the groundwater, and eventually joins the surface runoff from the basin, it is important to represent the surface-groundwater quality interactions accurately. Two major parameters are 1) the delay time of the subsurface flow to emerge as outflow, and 2) the proportion of subsurface flow that joins the stream. Hyatt et al. (1970), Thomas et al. (1971), and Hill et al. (1973) assumed both parameters to be constants. They estimated values for both as part of their model calibration procedure.

Since the runoff in winter months is predominantly from subsurface sources, an assumption that effluent groundwater is a constant proportion of the runoff from subsurface sources is not realistic. Narasimhan (1975) successfully modeled the subsurface contribution to total runoff by time variant parametric representation.

Salt pickup phenomena

The mechanisms governing salt pickup are highly complex. Reliable field data are a prerequisite to identify and quantify the salt pickup by the percolating waters. If total dissolved solids (TDS) is considered the salinity indicator, then the possible assumptions are that the rate of salt pickup is 1) proportional to the quantity of percolating water, 2) uniform over time, or 3) follows some more complicated relationships that needs to be derived for the agricultural drainage system from reliable field data.

The models reviewed in this study contained explicit parametric relationships for salt pickup. Hill et al. (1973) accounted for increases in salt flow in the surface and subsurface return flows separately. The parameter CF determined the proportional increase in agricultural surface return flow salt content, while the parameter SWS assigned a soil weathering rate in tons/acre/month. Hyatt et al. (1970) assigned a parameter C_{ga} to indicate the average salinity concentration within the soil solution beneath the agricultural lands. The value of C_{ga} was estimated during model calibration. The natural salinity contribution was estimated by assuming that within each basin substantial interchanges occur between surface and subsurface waters. The rate of salt flow resulting from the interchange process was estimated by the equation:

$$S_r^{NS} = k_p \ Q_r \ C_g \quad . \ . \ . \ . \ . \ . \ (3.3)$$

in which

S_r^{NS} = rate of salt flow contributed from natural sources within the basin.

k_p = percentage of the surface flow recirculating through the stream alluvium or groundwater basin.

Q_r = monthly average of inflow and outflow to a subbasin.

C_g = average water salinity level within the groundwater basin or stream alluvium. This quantity, assumed to be constant throughout the simulation period, was estimated from the average salinity level of the base flows of the streams within the sub-basin.

Melamed (1975) assumed a lumped "source sink" term to represent soil-salt interactions in both the dissolution and precipitation processes. He also assumed that the rate of the process was directly proportional to the difference between the concentration of the surrounding soil water solution, C, and some equilibrium concentration, R, for which the rate was zero. In equation form, the rate of the process is

$$fn(c,x,t) = K (R - C) \quad . \quad . \quad . \quad . \quad (3.4)$$

in which

K is a proportionality coefficient related to soil properties and salt composition.

Considering TDS as the salinity indicator, Riley and Jurinak (1979) assumed that, in extensive areas of the Upper Colorado River basin where the percolating water contacts saline marine shales underlying the soil, the volume of salt pickup is proportional to the volume of percolating water. By inference, then, this assumption implies that salt pickup is inversely proportional to irrigation efficiency.

The various assumptions in the detailed hydrosalinity models discussed in Table 3.4 are made to represent the complex solute reactions occurring in the soil-water system. The concentration of salts in the drain outflow is based on assumed chemical equilibrium conditions in the soil-water system. Shaffer et al. (1977) further assumed that no chemical reactions occurred in the outflow drains.

Limitations of the Existing Models

From the above overview of the basic relationships used in hydrosalinity models, the three relationships concluded to be the most limiting in accurate model representation are the chemical processes, surface-groundwater interrelationships, and the salt pickup phenomena. While better data are needed to establish just how limited the models are, some theoretical and empirical evidence follows.

Chemical processes

The common assumption that total dissolved solids is a conservative parameter may not apply for a wide range of values. For example, under large fluctuations in loading conditions (of the order of 10,000 mg/l), several significant mineral constituents may reach saturation. Precipitation and dissolution phenomena within the soil profile and along the stream may significantly alter the proportions of various ions, and as a result the relationship between constituent concentrations and the TDS or EC may become nonlinear beyond a certain range of concentrations.

Although TDS concentratons are adequately simulated by the detailed salinity models (Ayars 1976), the concentrations of certain individual ionic constituents are not. The inadequate representation of $CaSO_4$ $-CaCO_3$ - $Ca(CHCO_3)_2$ system in the models appeared to have significantly reduced prediction accuracy as illustrated in the following situations.

1) The low correlation between observed and simulated SO^{2-} concentrations in the models of Thomas et al. (1971), Narasimhan (1975), and Ayars (1976) is attributed to inadequate representation of the $CaSO_4$ $-CaCO_3$ systems.

2) Table 3.6 lists three models proposed by Willardson et al. (1979). Although the most sophisticated of the three models considered the precipitation, dissolution, and cation exchange of chemical constituents, the concentrations of Ca^{2+} and HCO_3 were underestimated by as much as 35 percent. All three models were observed to have inherent weakness in representing PCO_2 - HCO_3 - CO_3 - pH relationships.

Surface-groundwater interrelationships

Complex interrelationships between the surface water, soil water, and groundwater often exist. Representation of these relationships is quite general in most models and often based on calibrated percentage parameters. Huber et al. (1976) used the proportion of canal seepage that returns to the stream and the proportion of agricultural return flow that is available for rediversion.

Most of the TDS models assumed either ungaged surface or subsurface flows to achieve mass balance. The resulting freedom in model calibration could lead to major misrepresentation of the relative magnitudes of the component salt loadings within the system, particularly between seepage returns and subsurface return flows. The following situations illustrated the importance of identifying and accurately quantifying the surface-groundwater interrelationships.

1) Salinity modeling studies of the Duchesne River basin conducted by UWRL (1975) identified that significant recycling of stream diversions occurred within the basin. These findings subsequently were supported by Mundorff (1977).

2) Weston (1975) identified groundwater as the primary agent of salt pickup and transport to the Colorado River from the

Table 3.6. Comparison of results with more sophisticated modeling of chemical reactions (after Willardson et al. 1979).

Water Quality Constituent	Sampling day	SALT FLOW I (chemical reactions in the soil not considered)	SALT FLOW II (chemical precipitation and dissolution considered)	SALT FLOW III (chemical precipitation, dissolution and cation exchange considered)
EC	148	*satisfactory	satisfactory	satisfactory
	278	overestimated	overestimated	satisfactory
SAR	140	Too high at 25 cm depth.	Too high at 40 and 75 cm depth.	satisfactory
	278	Underestimated at 25 and 50 cm depth.	Overestimated at 25 and 50 cm depths, but underestimated at 75 cm depth.	satisfactory
Cl^-		satisfactory	satisfactory	satisfactory
Ca^{2+}		overestimated		underestimated the concentration in about one half of the cases. (Variation 0 to 35%)
Na^+			Not adequately predicted	satisfactory
Mg^{2+} and K^+		Predicted movement of these ions from the upper depth increments was too rapid.	Same as in SALT FLOW I.	satisfactory
SO_4^{2-}		SO_4^{2-} acted as an inert salt that was not affected by precipitation or dissolution of gypsum.	Prediction was in error inversely to the direction of Ca^{2+} error.	Satisfactory under both dissolution and precipitation conditions.
HCO_3^-				Generally underestimated
P_{CO_2} - HCO_3 - CO_3 -pH			weak	weak

*Only subjective judgments of results are provided.

Grand Valley. Available data indicated that the winter flows in all washes and drains in the Grand Valley gain salt as they move downstream, and a hydraulic connection between groundwater and the Colorado River appeared to exist through a Cobble aquifer. Therefore, inadequate representation of surface-groundwater interrelationships in a hydrosalinity model of the Grand Valley would have serious consequences in the model predictions.

3) Rhoades et al. (1974) found that leaching would reduce soil mineral dissolution and enhance precipitation of gypsum and lime, thereby reducing the salt load in drainage water. In order to quantify the effects of reduced leaching fractions on downstream river, soil, and groundwater compositions, the corresponding solute flow components require critical study and recognition.

Each of these examples illustrates the importance of covering the spectrum of possible seepage and groundwater return flow relationships in hydrosalinity modeling. If the relationships that are in fact most important in a given location are omitted, the model can often still be balanced. However, the results will not correctly identify salt sources. As to applications, the processes that it suggests for salinity control through improved water management may, when practiced, accomplish very little. The only way to avoid this situation is to have a valid model and reliable data to calibrate it.

Salt pickup phenomena

The soil-water system models developed by Rhoades et al. (1975) applied chemical equilibrium relationships to the soil solution to predict salt concentrations at or near the bottom of the root zone. Therefore, these models perform best where displacement of groundwater and pickup of previously deposited salts located in the groundwater flow path are eliminated as potential contributors to agricultural return flows. It is also important to consider differences between whether or not the drainage water is open to the atmosphere, as it is in open ditches, or closed to the atmosphere, as it is deep percolation water joining the groundwater reservoir. For example, Rhoades et al. (1973) provided experimental evidence that in Pachappa sandy loam soils the total solute burden of drainage waters is less in waters exposed to the atmosphere than in waters percolating down into the groundwater reservoirs. However, the drainage waters exposed to the atmosphere had a greater sodic hazard potential than did those waters closed to the atmosphere.

The way in which soil-water system models are synthesized into a general hydrosalinity model will therefore have significant limitations on the accuracy of predicting salt burden of drainage water. As an example, Shaffer et al. (1977) simulate the chemical and physical processes associated with agricultural lands drained by subsurface tile drainage systems. Their model can also be applied to areas with surface drainage systems and predict flow and quality parameters as additional points within the plant-soil-aquifer system. The assumption is made however that no chemical reactions occur after the mixing process in the drain. This last assumption appears to be a significant limitation to its application because drain water quality can be unstable from a chemical standpoint.

Discussion

Based on the above review, the major capabilities, model limitations, and gaps by category of the models are abstracted in Table 3.7. The indications are that all of the available salinity models have inherent limitations and cannot be considered reliable when applied to a specific area for evaluating management options for salinity control. The simple TDS models do not adequately represent the various hydrologic and salinity processes of the system, while the detailed models require extensive field data of types usually available only in research plots under controlled conditions. Although the detailed models address the chemical processes that occur within the soil profile, they usually do not adequately represent the $CaCO_3$ - $Ca(HCO_3)_2$ subsystem. In addition, a failure by many models to adequately represent the interactions between surface and groundwater components frequently results in a loss of accuracy in predicting the constituent ions in the return flows. The degree of accuracy depends, of course, on the degree to which the model actually represents the particular soil-water being modeled.

The most constructive approach to this situation (simple models being unreliable because key physical processes were omitted and the more complex models being unreliable because adequate data were not available for their calibration) was taken as trying to develop a model which generally would be consistent with data availability and yet provide sufficient accuracy to permit realistic evaluation of various salinity control management options. The following chapters describe the procedures followed in developing such a hydrosalinity model. Limited management runs are also included to demonstrate the utility of the proposed model for evaluating various possible management options. A review of relevant research results concerning the various processes that occur within agricultural systems formed the basis for representing them in the hydrosalinity model developed for this study.

Table 3.7. Capabilities by model categories to evaluate salinity management alternatives (for short term and long range predictions).

Model category	Capability	Limitations and gaps with respect to using the models for management purposes
Water and salt budget models (Table 3.2)	Both one dimensional instream models and two dimensional river basin models exist that are capable of predicting short term responses and long trends in development changes.	Representation of the component hydrologic processes such as canal seepage, tailwater, irrigation efficiency, and surface-groundwater relationships and salt pickup processes are not adequate resulting in loss of prediction accuracy.
Soil water system models (Table 3.3)	Capable of predicting short term responses of constituent salts within the root zone soil profile on account of treatment changes.	These are mostly process identification models and are not intended for studying effects of basinwide salinity management. Require extensive chemical data to operate the models.
Detailed hydrosalinity models (Table 3.4)	Consider the several physico-chemical processes within the agricultural system and are generally capable of predicting both short term and long term trends affected by management alternatives.	Although the predictions of TDS concentrations are reliable, there is significant loss of accuracy in the predictions of individual ions. The models require extensive data and take excessive computer time to operate.
Groundwater system models (Table 3.5)	Predict long term status of groundwater quality.	Chemical reactions that affect the solute concentrations are not considered.

CHAPTER 4.
DEVELOPMENT AND TESTING OF A NEW HYDROSALINITY MODEL

Introduction

In order to quantify how the various salinity management options would affect the streamflow quantity and quality in a particular river basin, the physical processes controlling salt and water movement within the basin must be modeled. The model should represent these processes sufficiently well to indicate how they would respond to the range of management options. More specifically, the desired capabilities of a water management hydrosalinity model are:

1. To simulate, based on present conditions, the water quantity and quality (represented by TDS) at specific locations and over desired time intervals, considering a) the interrelationships between the surface water, soil water, and groundwater systems, and b) salt precipitation within the soil profile and subsequent salt pickup by percolating waters through the shallow soil profile and the deeper groundwater zones.

2. To estimate the immediate and long-term effects on water quantity and quality of the salinity management alternatives.

Assessment of the immediate effects of applying alternative management options requires the development of a dynamic simulation model which accurately represents the significant processes taking place in the irrigation system. From a salinity management standpoint, monthly or seasonal predictions are adequate. Of course, the same model is capable of predicting long term effects provided the operating rules, system definition, and management options remain unchanged. Generally, the assessment of long term effects (over decades or centuries) has not been very accurate. Over these longer time spans, advancing technology, political expediency, or changing attitudes drastically alter soil and water uses in ways which can never be completely anticipated. Therefore, long term means the longest time period for which current rules and operating procedures apply, seldom more than 50 years and often less than 5 years.

The goal here is to develop a hydrosalinity management model capable of representing, in the degree needed for water management purposes, water and salt movement processes within the soil profile. The starting point was the assessment of the gaps and the validity of inherent assumptions in the existing models discussed earlier in this report. It was envisioned that the management tool would still use total dissolved solids as the salinity indicator, be relatively simple, and be able to utilize generally available data for calibration and prediction.

Model Development

The result of this effort was that a salinity component (SALT) was developed and combined with the hydrology component (BSAM) developed by Huber et al. (1976) to form a hydrosalinity model, BSAM-SALT. The model uses monthly time increments and permits variation in spatial resolution to match the requirements of specific applications. A schematic of the flow paths modeled is shown in Figure 4.1.

Modeling Concepts

The basic concept used in formulating BSAM-SALT was that the runoff cycle can be represented by various storages and flows between storages. Each flow and storage potentially carries or is associated with a salinity concentration, except for the precipitation and evaporation processes where salinity content is negligible. The conceptual breakdown is shown in Figure 4.1.

In the model, salinity concentrations are read from measured data or estimated for each flow quantity shown in Figure 4.1, and rates of salt outflow are estimated by combining salt and water movements through the system. The overall salt balance is represented by the equation:

$$Q_{so}C_{so} = \sum_{j=1}^{n} Q_{sj}C_{sj} + \sum_{k=1}^{\ell} Qg_kCg_k \pm \sum_{p=1}^{m} \Delta S_pC_{scp} \quad \ldots\ldots (4.1)$$

in which

Q_{so} = water outflow rate from hydrologic unit or system

C_{so} = salt concentration in the outflow water

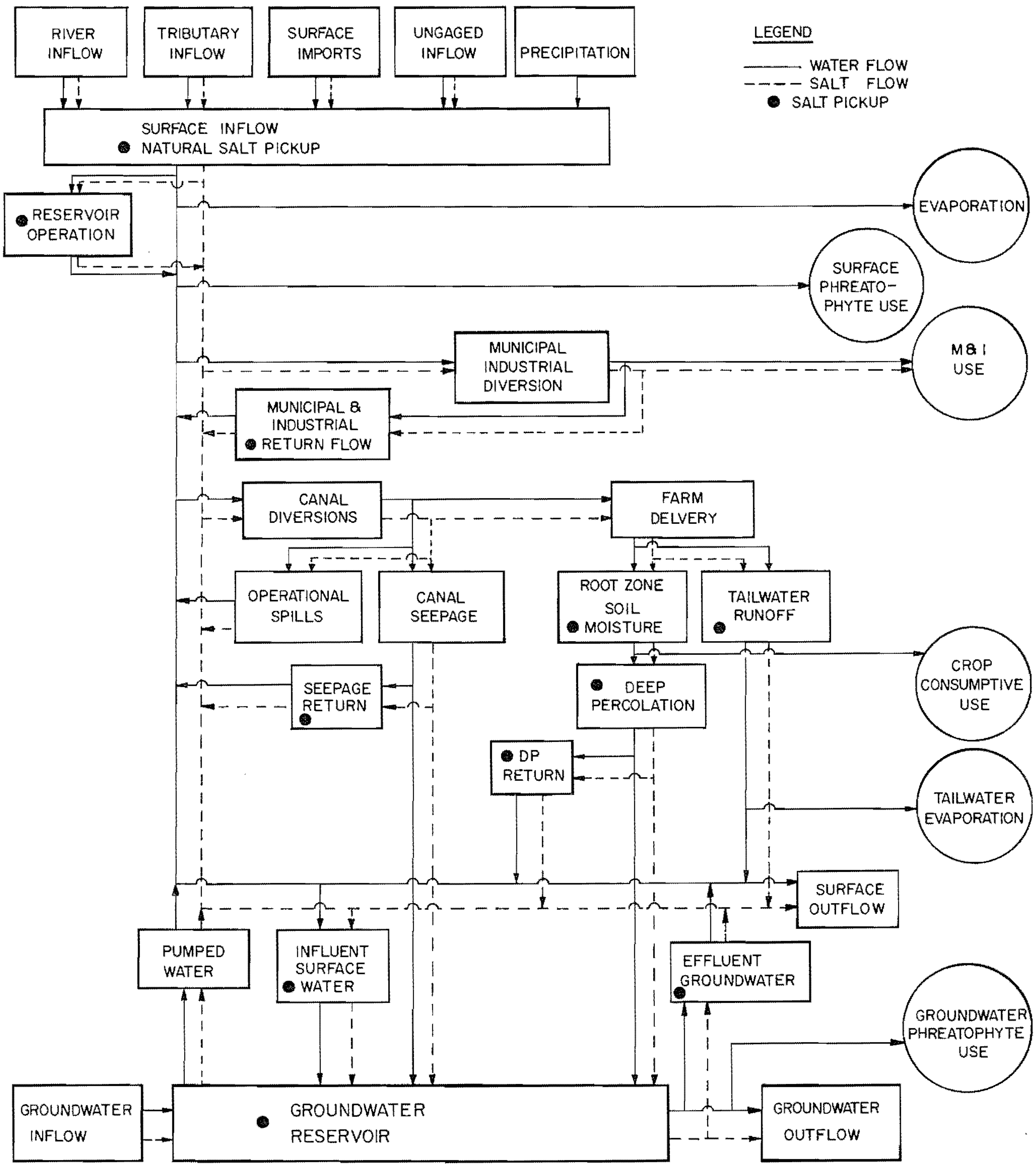


Figure 4.1. Schematic diagram of the flow paths in the hydrosalinity model, BSAM-SALT.

Q_{sj} = rate of water inflow from surface source j

C_{sj} = salt concentration of surface source j

Qg_k = rate of water inflow from groundwater source k

Cg_k = salt concentration of groundwater source k

ΔS_p = change in water stored in storage element p

C_{scp} = salt concentration associated with storage element p

The terms on the right side of Equation 4.1 represent the respective salt contributions to the outflow from surface and subsurface sources and changes in the salt content of the various storage elements of the system. As suggested by Figure 4.1, salts may also transfer among the various (n + ℓ + m)terms of Equation 4.1.

The modeled hydrologic processes can be grouped into 1) inbasin consumptive uses, 2) the flow components susceptible to management manipulation, 3) runoff components, 4) interrelationships between surface and groundwaters.

Inbasin use processes

Six inbasin uses are shown in the circles along the right side of Figure 4.1. Consumptive use may be partly municipal and industrial (M & I), but it is mostly agricultural in areas where hydrosalinity models are applied. The three principal losses are consumptive use by agricultural crops (AET), evapotranspiration by phreatophytes (ETPH), and evaporation from tail water (ECUTW).

The hydrologic processes that deliver water for inbasin use are canal diversions, defined to include groundwater pumping, and effluent groundwater movement. The pumped water and return flows from canal seepage and deep percolation represent the recycling of water within the basin. The sources of water diverted through the canals consist of 1) surface and subsurface flows from developed and undeveloped lands, 2) streamflows (gaged and ungaged), 3) reservoir releases, 4) pumped groundwater, 5) seepage returns, and 6) return flows rediverted from deep percolation. These components are considered separately by the model to facilitate the estimation of the salt concentration associated with each.

The flow components susceptible to management manipulation. The flow components explicitly controlled in the management of the irrigation system are the canal diversions and the water applied on the farms. The operating system is characterized by operational spills, canal seepage, tail water runoff, and deep percolations. These flows are all interrelated and are a function of the irrigation methods and practices. These in turn determine the two efficiency factors identified and used in the model, the canal conveyance efficiency and the farm or field application efficiency. The conveyance efficiency, ECV, is defined as the complement of the normalized seepage loss rate of the canal i.e., ECV is

$$1 - \frac{\text{canal seepage rate}}{\text{canal diversion rate}}.$$

The farm application efficiency is defined as the proportion of the irrigation water delivered to the farm that enters the root zone and remains available for evapotranspiration by the irrigated crops. It is a function of the moisture infiltration and holding properties of the root zone, antecedent rain plus snowmelt, and the amounts, time pattern, and application methods of water delivered to the farm. As a result of this interaction between weather conditions and farmer irrigation practices and the influence of spatial soil characteristic variability, irrigation application efficiency fluctuates widely. It is, therefore, not modeled as a constant but, rather, a value is set in model calibration and printed out.

Runoff components. The runoff flowing in the stream is modeled in two components. One accounts for the flow measured by the gage, and the other accounts for subsurface flow in the alluvium that is not being measured by the gage. The unmeasured flow is taken as a fixed fraction of the measured component.

Interrelationships between surface and groundwater. The capability of considering separately the lag time responses in the component flows, namely canal seepage, deep percolation, and groundwater, represented in a minimal way a pseudo relationship between the surface and subsurface flows. The effluent groundwater flow also is apportioned between groundwater outflow from the basin and its contribution to surface runoff from the basin.

Major salinity processes considered in the model are, 1) the concentrating effects of evapotranspiration in the root zone soil-solution to the threshold salt concentration, 2) the salt pickup by percolating waters, namely, canal seepage, deep percolation, and groundwater, 3) mixing of the soil water of varying concentrations.

Concentrating effect. The concentrating effect is produced by evaporation, consumptive use by natural vegetation and by irrigated crops, and by diversion of high quality water from the basin. In the irrigated area, however, the concentrating effect is limited to the threshold soil solution-concentration (TSC) discussed in the previous sections.

Salt pickup processes. The rate of salt pickup by the water percolating through the soil profile is one of the most important factors in developing the salt component of the model. A parametric representation of the pickup process was used in the model because the mechanisms describing the salt pickup involve complex chemical reactions which are often found to be unique to a particular basin. The parameters associated with the salt pickup are determined during the model verfication or calibration stage. Both the natural sources (instream and undeveloped land) and the agricultural sources are parametrically represented. The agricultural processes contributing to salt pickup include canal seepage, tailwater runoff, operational spills, deep percolation, and mineral weathering of deep groundwater.

Mixing effects of soil water concentrations. The composition of the drainage waters from an agricultural subbasin is a composite of the component sources. These components include the canal seepage, deep percolation, and groundwater; and each has different concentrations due to complex physical, chemical, and biological factors. The blending of these component concentrations is accomplished with a parametric linear reservoir routing or mixing algorithm.

Assumptions

The major assumption for modeling the hydrologic component is continuity of mass through the various processes within the system. Other assumptions are discussed in Huber et al. (1976). The major assumptions in the salinity component include:

1. To facilitate use of the model for evaluating salinity management alternatives, it is considered sufficient to represent TDS as the salinity indicator, thus avoiding the complex chemical reactions that occur within the soil water system.

2. There exists an equilibrium Threshold Salt Concentration (TSC) within the root zone soil-water system at which there is neither a net precipitation nor a net dissolution of salts in the soil profile. It is considered that TSC is highly site specific and depends on the type of soil, quality of applied irrigation water, and the degree of leaching already taken place at the time of consideration. The accumulation of salt within the soil profile due to concentrating effects are thus limited to TSC.

Modeling Procedures

The procedures adopted in modeling the salinity system are the following:

1) Salt pickup in overland flow and percolating waters can be modeled as partly being proportional to water flow and partly as having a constant value in the time interval chosen in the model. Represented mathematically,

$$SNAT = QIN * PUNAT + PNATU \quad . \quad . \quad (4.2)$$

in which

SNAT = salt pickup from natural sources

QIN = water inflow

PUNAT = natural salt pickup in QIN proportional to flow

PNATU = natural salt pickup in QIN proportional to time

Either PUNAT or PNATU or both can be used in the simulation. The choice is between assuming a constant volume of salt pickup or a constant equilibrium concentration in the percolating water. Similar relationships are adopted for salt pickup in tail water, canal seepage, and deep percolation.

2. Salt precipitation within the soil water system due to concentrating effects is modeled using the 'Threshold Salt Concentration' (TSC) concept. This concept is given in equation form as follows:

$$SPT = (SSG - SSI) * PSMSPT$$

$$\text{if } CSS > TSC \quad . \quad . \quad . \quad . \quad . \quad (4.3)$$

in which

SPT = precipitated salt

SSG = end of month (or time increment of model) salt storage in the root zone

SSI = initial salt storage in the root zone

PSMSPT = proportion of salt precipitated when TSC is exceeded

CSS = end of month concentration of soil salt in the root zone

Similar relationships are used to predict the salt precipitation in the soil profile lying below the root zone and above the groundwater level, as applicable.

3. Mineral weathering in the groundwater system is parametrically represented as occurring at a constant rate over the time

interval of the model. The concentration of salt in the groundwater reservoir is computed by the mathematical representation:

$$SGGW = SGWIN + SDUMSP + PGWU + SQPUM \quad (4.3)$$

and

$$CGGW = SGGW/GGW \quad (4.4)$$

in which

SGGW = end of month salt in groundwater zone

SGWIN = initial salt storage in groundwater zone

SDUMSP = salts added to groundwater zone during the month from canal seepage, deep percolation, infiltration from urban and undeveloped areas, and the infiltration from main stream

PGWU = weathering rate of minerals in groundwater, in tons/acre/month (or time interval of the model)

SQPUM = salt pumped from groundwater storage

CGGW = composite concentration of groundwater

GGW = end of month groundwater storage

4. In order to represent a) the mixing of salts within the soil water system, and b) the blending of groundwater concentrations emerging as effluent flows and joining the surface runoff, the salinity concentrations can be routed independent of any routing of the corresponding water flow components, using the linear reservoir routing technique in Huber et al.(1976).

Hydrology and Salt Model Interface

BSAM required little modification in order to accommodate the salt processess. The change was accomplished by adding a set of salinity subroutines which simulate the salt processes described above and attach the proper salinity values to the associated hydrologic system components to satisfy the salt mass balance conditions.

The expansion required collecting additional input data to represent salinity as well as calibrating the parameters used to model the salt transport processes. In addition, the objective function used to calibrate the model to a specific basin had to be expanded to include the salt components of the model output. The two additional scalar measures inserted to do this were 1) a scalar measure of the weighted differences between the computed and simulated salt loads and 2) a scalar measure of the weighted differences between the computed and simulated concentration.

Model Calibration

The simulation model may be thought of as a complex nonlinear function. The model parameters which affect the system response include model coefficients, initial conditions, and storage capacities. Simulation models are generally calibrated by a trial and error process where a run is made with all parameters set to an initial estimate and the input data given for a system with a known output. The simulation results are then compared with the observed output. If they show too much divergence, the run is repeated with a new set of parameters and the results again compared. This is repeated until the error has been reduced to acceptable limits. The modified pattern search algorithm (Huber 1970) already incorporated within the hydrology model was retained to aid in the calibration process and a new subroutine was included to allow interactive computation.

The objective function aiding in the calibration of BSAM-SALT combines three separate measures. One for the response of the hydrologic system, one for the response of the salt outflow system and the third combines the first two by measuring the response to the salinity concentration of the outflow. In addition, the algebraic sum of the differences between the computed and observed responses for each of the three measures is given to help identify the existence of any accumulating bias within the model.

The area chosen to test the model was Grand Valley in the Colorado River Drainage Basin. The Colorado River enters the Grand Valley from the east, joins with the Gunnison River at the city of Grand Junction, Colorado, and flows west into Utah. The salt loading of the river as it flows through the valley is significant and has been the subject of intensive study during the last few years. In order to calibrate the model, the stream flows, canal diversions, climatological data, and cropping land use patterns were collected for the water years 1970, 1971, and 1972. Other data needed for model application included information concerning the conveyance and application efficiencies associated with the irrigated agriculture of the valley.

A review of the available literature indicated that major variations existed among the efficiency estimates published by the researchers who have or who are now studying the area. In trying to replicate the data available, one would have to postulate different irrigation management systems as characterizing the area. The irrigation system efficiencies published by the different researchers are summarized in Table 4.1. Further field research would need to be undertaken to ascertain which data set most closely represents the actual system to be modeled. This aspect will be further discussed in the section on sensitivity analysis.

The efficiency values selected to represent the valley were those adopted by the Grand Valley Salinity Coordinating Committee. Another problem, which is not atypical, was that the water quality data for most of the stations were very limited. This required using regressions between specific conductivity and the corresponding stream flow at each station to extend the available salinity record (TDS). The hydrologic and climatologic data stations adopted for model calibration are listed in Table 4.2. The results with the calibrated model applying BSAM-SALT to the Grand Valley are shown in Figure 4.2. The simulated stream and salt outflow for the calibration period were within one percent of the gaged records. The predicted values of salt during low flow months, April in particular, however, differed by as much as 28 percent. Inadequate water quality data are a possible explanation for such large differences in the salinity predictions. The calibration results indicated that about 3 percent of the surface flow was unmeasured by the outflow gage, probably as a result of underflow. Predicted values of actual and potential consumptive use were within 2 percent.

The calibration process identified the following assumptions about the system:

1. Salt pickup values for canal seepage and for deep percolation water were the same.

2. Salt pickup in the groundwater zone was the result of mineral weathering.

3. The concentration of the seepage and percolating waters was unaffected by antecedent conditions.

4. Salt pickup from the undeveloped land area was negligible.

The predicted salinity concentrations during low flows ranged between 8,000-10,000 mg/l, which agreed with the observed concentrations in some of the drains in the area. The results also indicated that during the calibration period, there was no precipitation of salts in the soil profile, assuming a threshold salt concentration value of 4,000 mg/l. This result, however, is based on the assumption that salt pickup by the percolating waters was proportional to the amount of water passing through the profile. It is not difficult to calibrate the model with observed quantity and quality (TDS) data of surface outflows based on other assumptions. However, the predictive results under imposed management options would be different. In addition to calibrating the model for the Grand Valley area, data from the Circleville subbasin in Utah was used to illustrate model calibration with a set of different conditions.

Calibration results of Circleville subbasin, Utah. Although the data for calibration at Circleville are less reliable than those for the Grand Valley area, the simulated stream flow and salt were within 0.1 percent of the gaged records for the calibration period. However, the monthly predictions differed by as much as 57 percent for water and 10 percent for salt. The calibration results were obtained by assuming:

1. No operational spills, tail water, or groundwater outflow from the basin.

2. Unequal amounts of salt pickup in canal seepage and deep percolation water.

3. No salt pickup due to mineral weathering in the groundwater zone.

4. Significant natural salt pickup from undeveloped land.

5. Antecedent conditions did affect the salinity concentration of seepage and deep percolation water.

The values resulting from the calibrations for Grand Valley and Circleville Basin are shown in Table 4.3 for the more important parameters. The predicted groundwater salinity concentrations for the Circleville Basin during low flow months were 400 - 1,300 mg/l, which appeared reasonable according to available data.

Sensitivity Analysis

In order to study how the model reacts to variations in selected parameters, sensitivity studies were run on the calibrated model for Grand Valley. Figures 4.3 through 4.6 show the sensitivity of simulated stream flow, salt outflow, and salinity concentrations to variations in canal conveyance efficiency, canal diversions, tail water, and operational spills. The effects are summarized in Table 4.4.

Sensitivity to canal seepage

Total seepage losses from main canals, laterals, and ditches were considered for this study. The results showed that improving canal conveyance efficiency from 81 percent to 100 percent would reduce the total salt loading for the 3 years by about 204,000 tons and the salinity (TDS) concentration by 9 mg/l. Increase in stream flow, however, was not appreciable. The reduction in salt loading may be attributed to less seepage through the soil profile.

Table 4.1. Irrigation system efficiencies adopted by different researchers in the Grand Valley area, Colorado.

Component Item	ARS Studies (1977)	Grand Valley Salinity Coordinating Committee (1977)	Walker (1979)	UWRL (1975)[1]
1. Canal, lateral and ditch seepage losses (percent irrigation diversions)	22	19	13	24
2. Operational spills (percent irrigation diversions)	34.5	38	18	14
3. Tailwater runoff (percent of field delivery)	44	33	52	14
4. Evapotranspiration (percent of field delivery)	46	46	39	*[2]
5. Deep percolation (percent of field delivery)	10	21	16	*[2]

[1]Based on information given by Canal Companies.

[2]Computed during model calibration.

Table 4.2. Hydrologic and climatologic data stations used in this study.

Component Item	Grand Valley Area, Colorado	Circleville Subbasin, Utah
Stream Inflows (main stem)	Colorado River at Cameo, CO (09095500)	Sevier River at Hatch, UT (10174500)
Tributary Inflow	Gunnison River near Grand Junction, CO (09152500) Plateau Creek near Cameo, CO (09105000)	Panguitch Creek near Panguitch, UT
Stream Outflow	Colorado River near Colorado - Utah State Line (09163500)	Sevier River in Circleville, UT (10180000)
Canal Diversions	Government highline canal Grand Valley Canal Redlands Canal	West Hatch Ditch East Hatch Ditch Long & East Bench Canal East Panguitch Canal Tebbs Ditch McEwan Canal Bear Creek Canal Marshall Ditch & Slough Whittaker Ditches
Precipitation and Temperature	Grand Junction & Fruita	Circleville & Panguitch
Reservoir Storage Considered	Nil	Nil
Pumped diversion from Groundwater	Nil	Estimated values based on studies of ARS
Imports	Nil	Nil
Exports	Nil	Nil

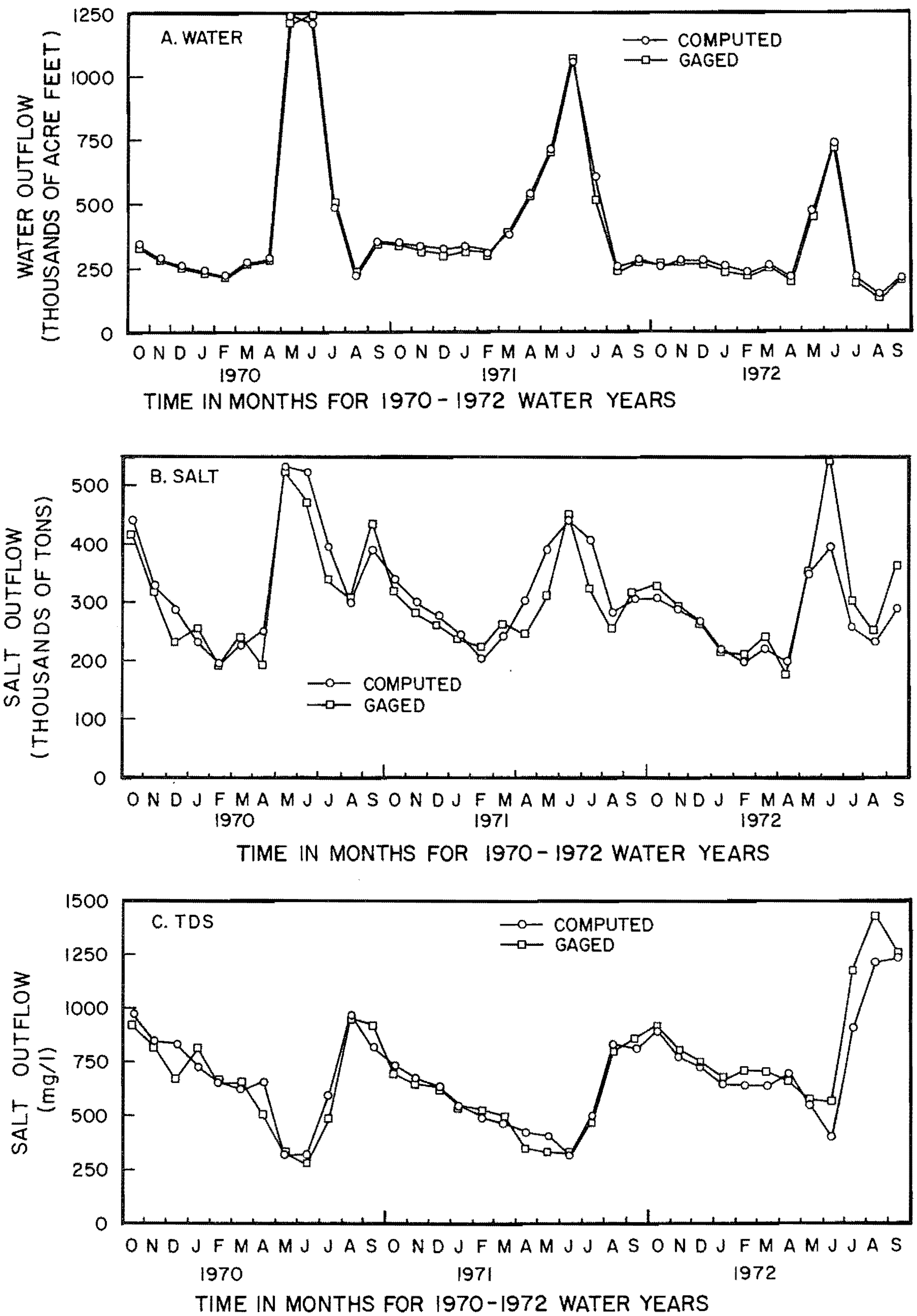


Figure 4.2. Calibration results from applying BSAM-SALT to the Grand Valley, Colorado, area.

Table 4.3. Selected parameter values adopted for BSAM-SALT calibration.

Flow Component	Parameter Values: Grand Valley, Colorado	Parameter Values: Circleville Basin, Utah
A. Water:		
---Canal seepage	81 percent	54 percent
---Operational spills	38 percent	0
---Tailwater runoff	33 percent	0
Predicted flow under the measured gage	3 percent	0
---Groundwater outflow	Nil	0
---Effluent groundwater Joining the streamflow	100 percent	100 percent
---Constant proportion of soil moisture joining D.P.		13 percent
B. Salt:		
---Salt pickup in canal seepage	2.6 tons/ac-ft	.025 tons/ac-ft
---Salt pickup in deep percolation (DP) water	2.6 tons/ac-ft	.004 tons/ac-ft
---Salt pickup due to mineral weathering in saline ground-water	0.13 tons/acre/mo	0
---Natural salt pickup from undeveloped land	Nil	.04 tons/ac-ft and 42 tons/acre/mo
---Predicted groundwater con-centrations during low flows	8,000-10,000 mg/1	400-1300 mg/1
---Routing coefficients for concentrations in seepage, DP, and groundwater	0	.88, 1.8, .4
Predicted root zone (RZ) salt precipitation	Nil	Nil
Predicted salt precipitation below RZ	Nil	Nil

Table 4.4. Summary of sensitivity studies for Grand Valley, Colorado.

Parameter		Change in Outflow (1970-1972)			
	Description	Change in Value	Water (ac-ft)	Salt (tons)	Concentration (mg/1)
Increase	Conveyance efficiency	81-100 percent	+ 28,459	- 204,563	- 9
Reduce	Operational spills	38-5 percent	+ 12,153	+ 1,140,752	+60
Reduce	Canal diversions	100-60 percent	+ 150,042	+ 88,846	- 1
		100-70 percent	+ 105,427	- 259,726	-17
		100-80 percent	+ 61,786	- 260,259	-16
Reduce	Tailwater	33-5 percent	- 38,438	+ 615,359	+31

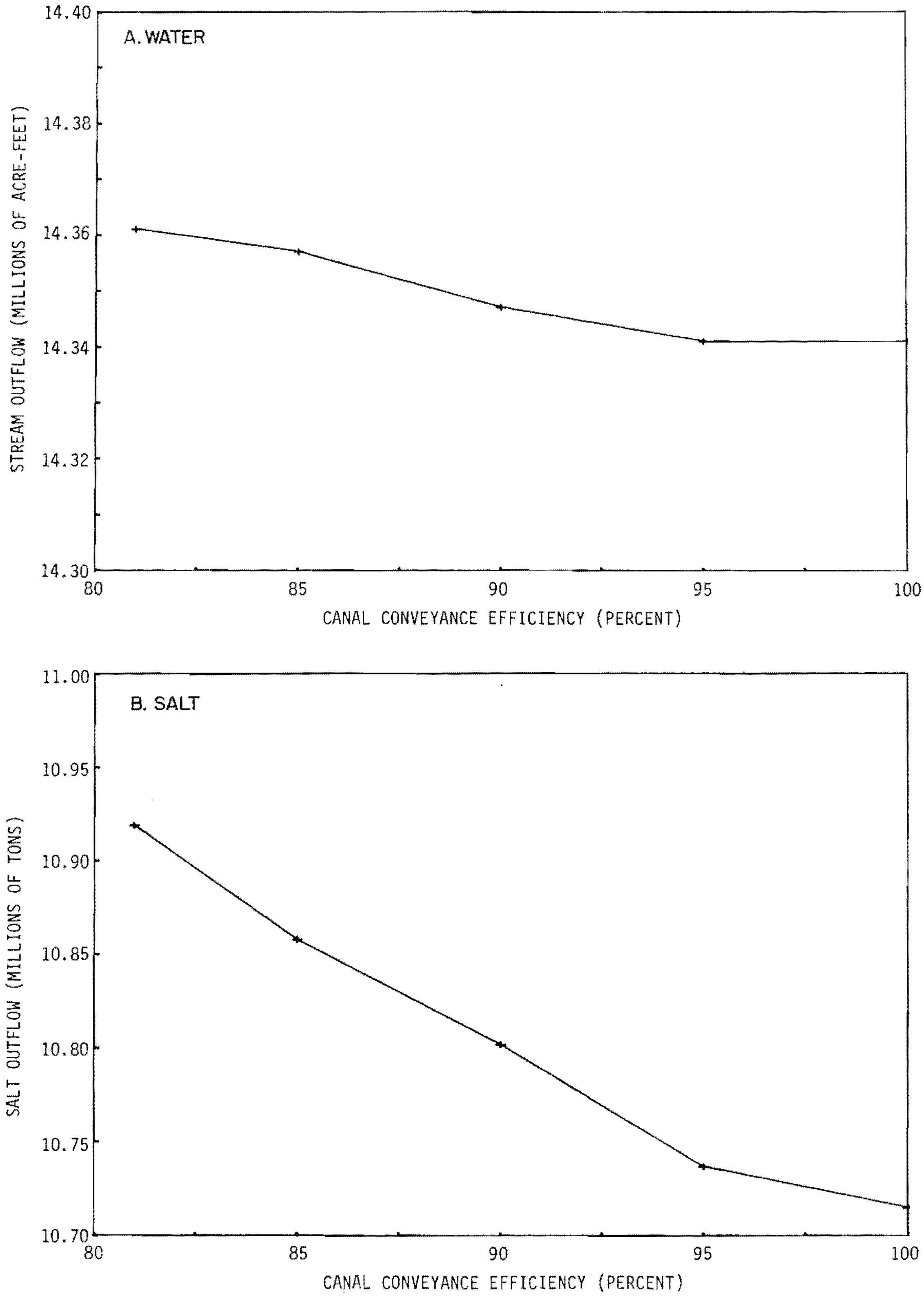


Figure 4.3. Sensitivity of a) stream outflow, b) salt outflow, and c) outflow salinity concentrations of the Colorado River below Grand Junction, Colorado, to canal conveyance efficiency. (Calibration based on component efficiencies adopted by the Grand Valley Salinity Coordinating Committee, i.e. spills = 38% and tail water = 33%)

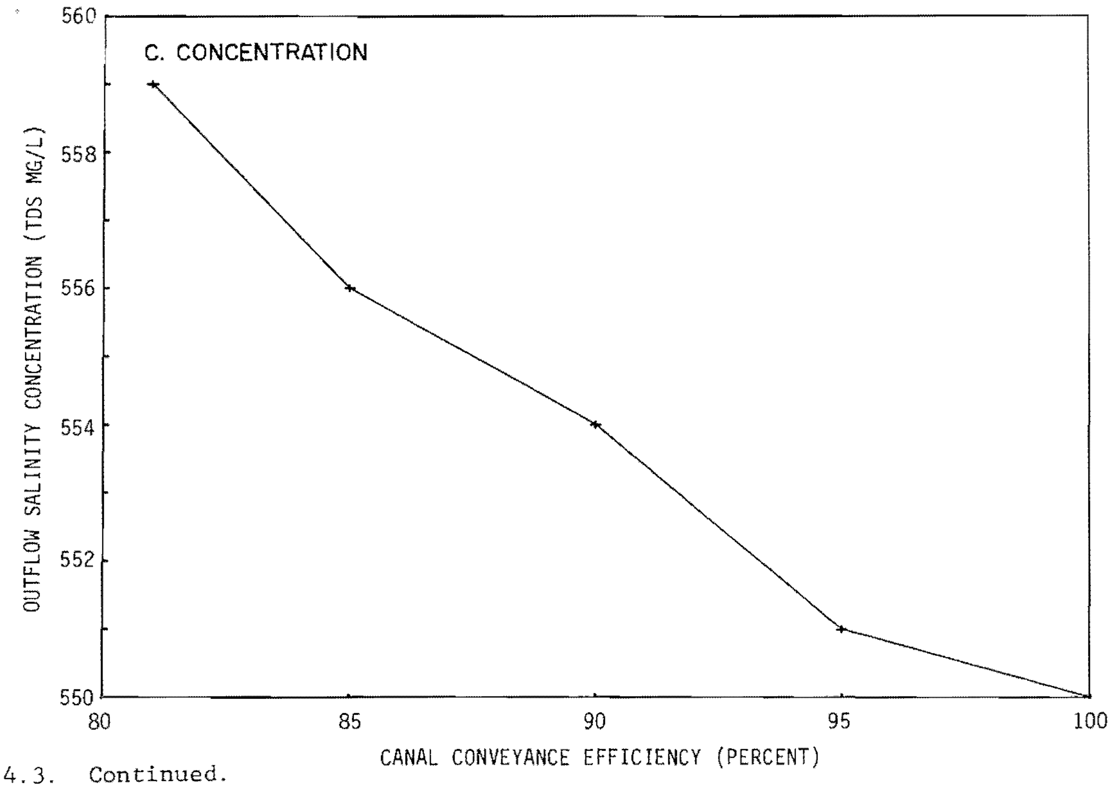


Figure 4.3. Continued.

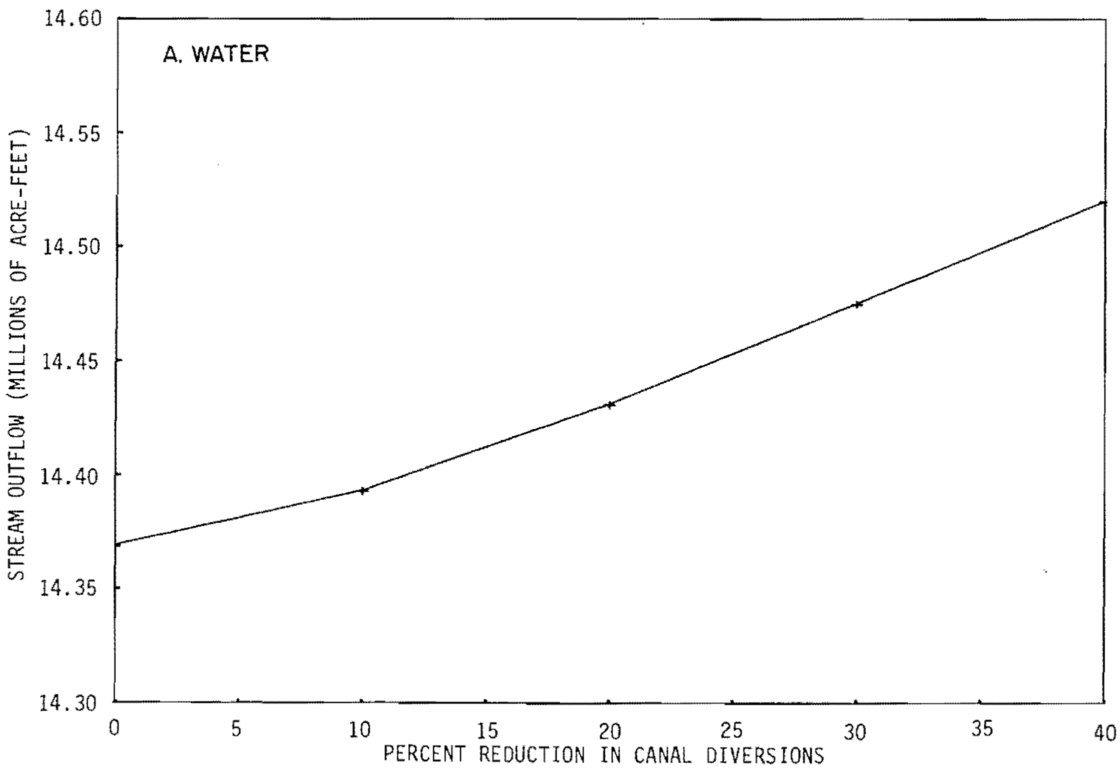
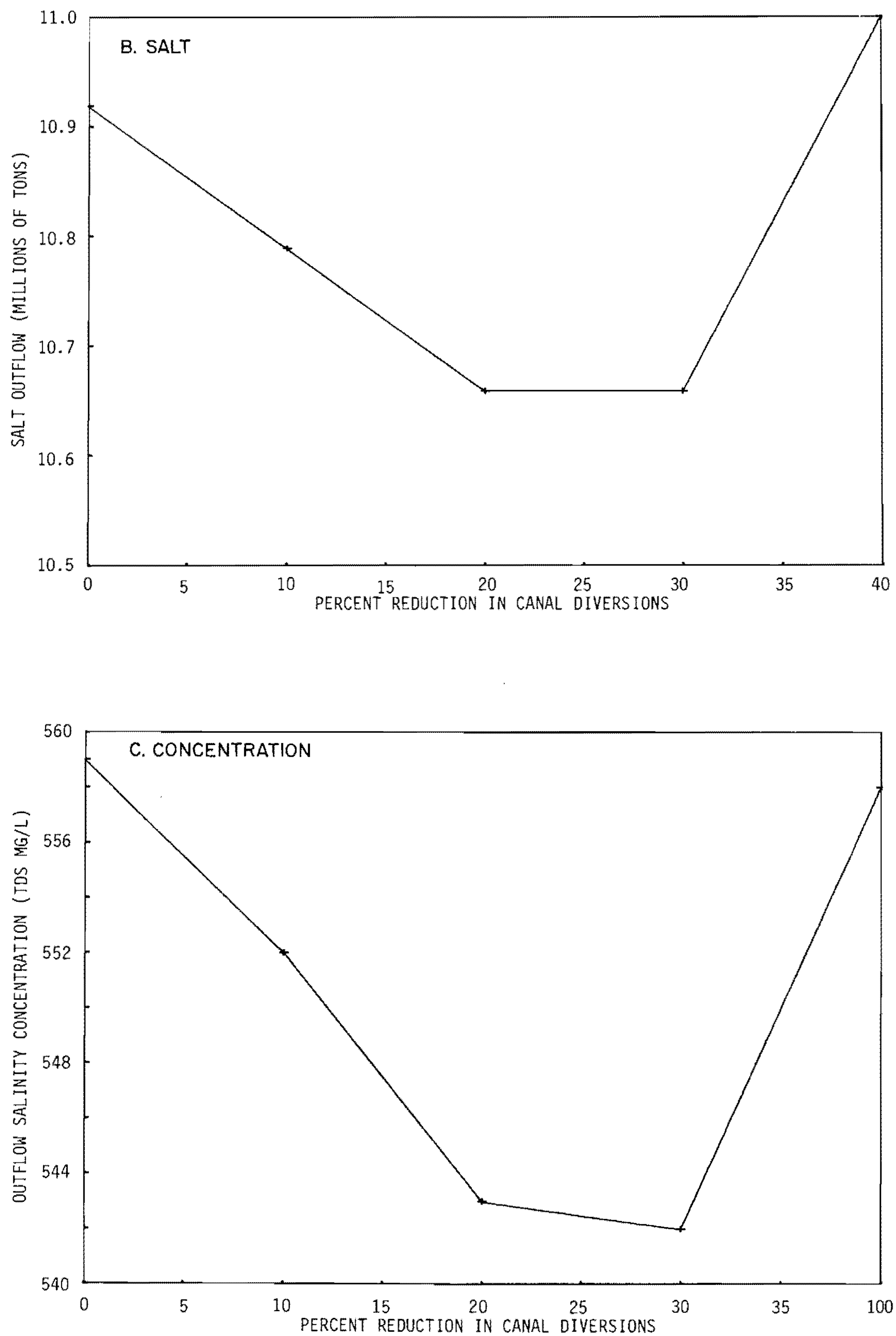


Figure 4.4. Sensitivity of a) stream outflow, b) salt outflow, and c) outflow salinity concentrations of the Colorado River below Grand Junction, Colorado, to a reduction in canal diversion. (Calibration based on component efficiencies adopted by the Grand Valley Salinity Coordinating Committee, i.e. spills = 38%, tailwater = 33%, and conveyance efficiency = 81%.)



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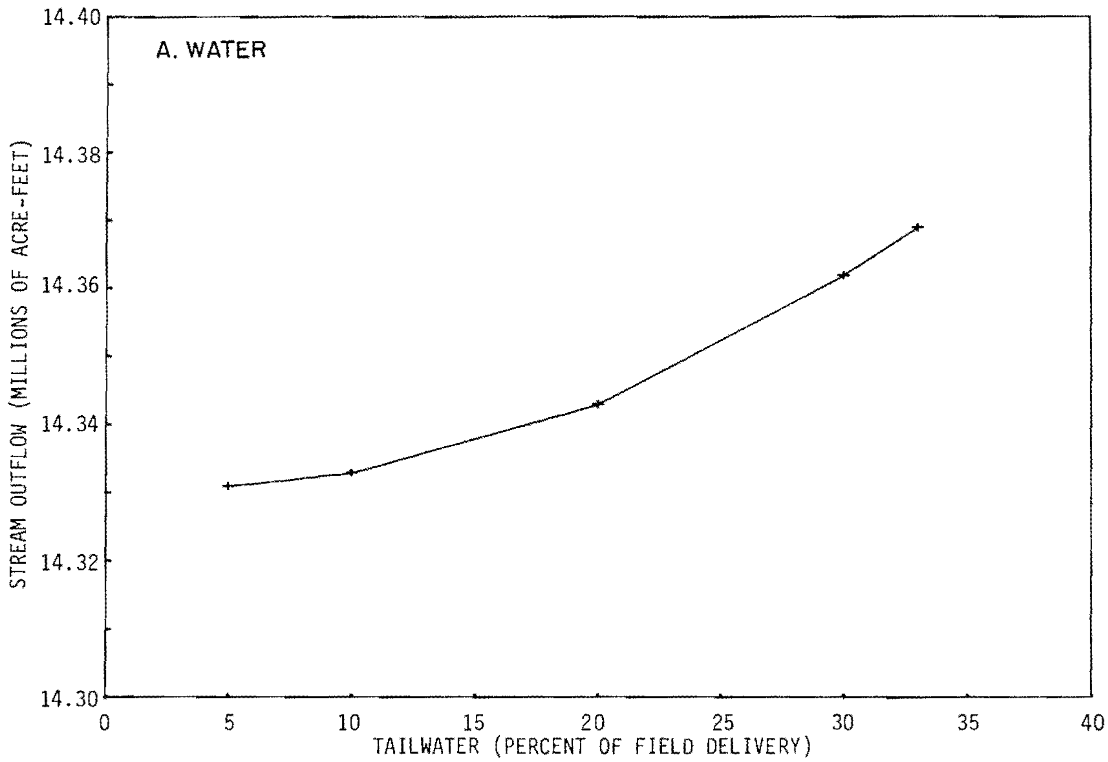


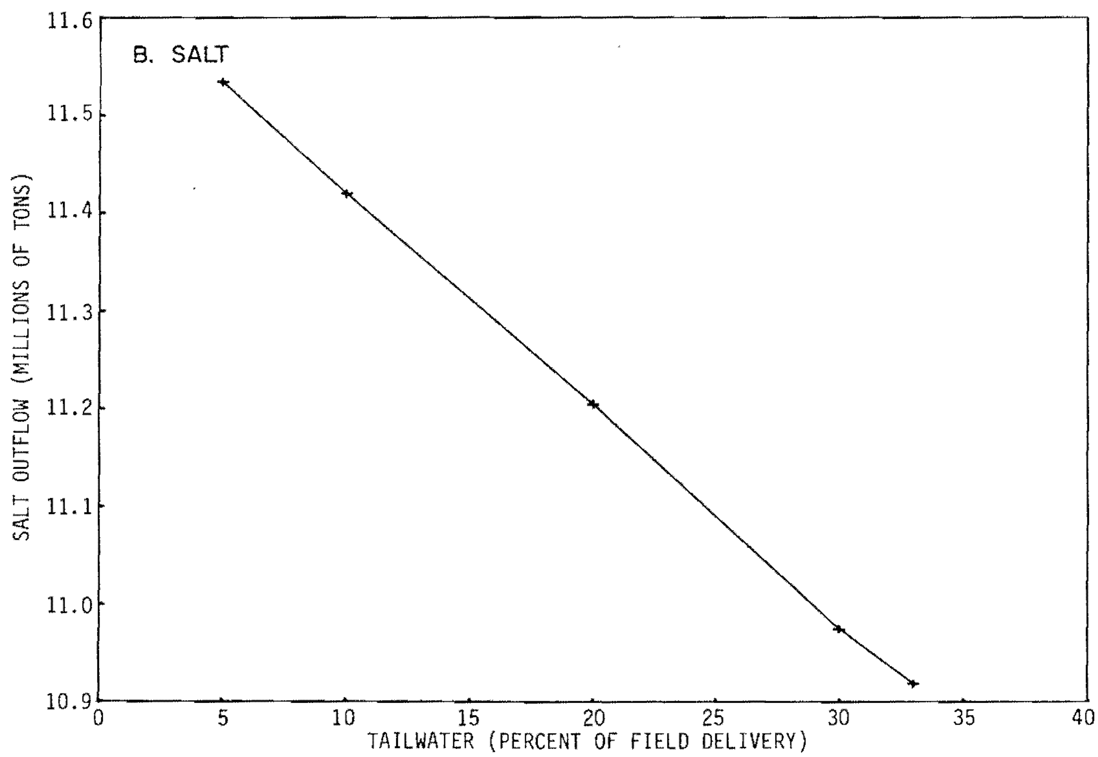


Figure 4.5. Sensitivity of a) stream outflow, b) salt outflow, and c) outflow salinity concentrations of the Colorado River below Grand Junction, Colorado, to the tail water runoff. (Calibration based on component efficiencies adopted by the Grand Valley Coordinating Committee, i.e. spills = 38% and canal conveyance efficiency = 81%.)

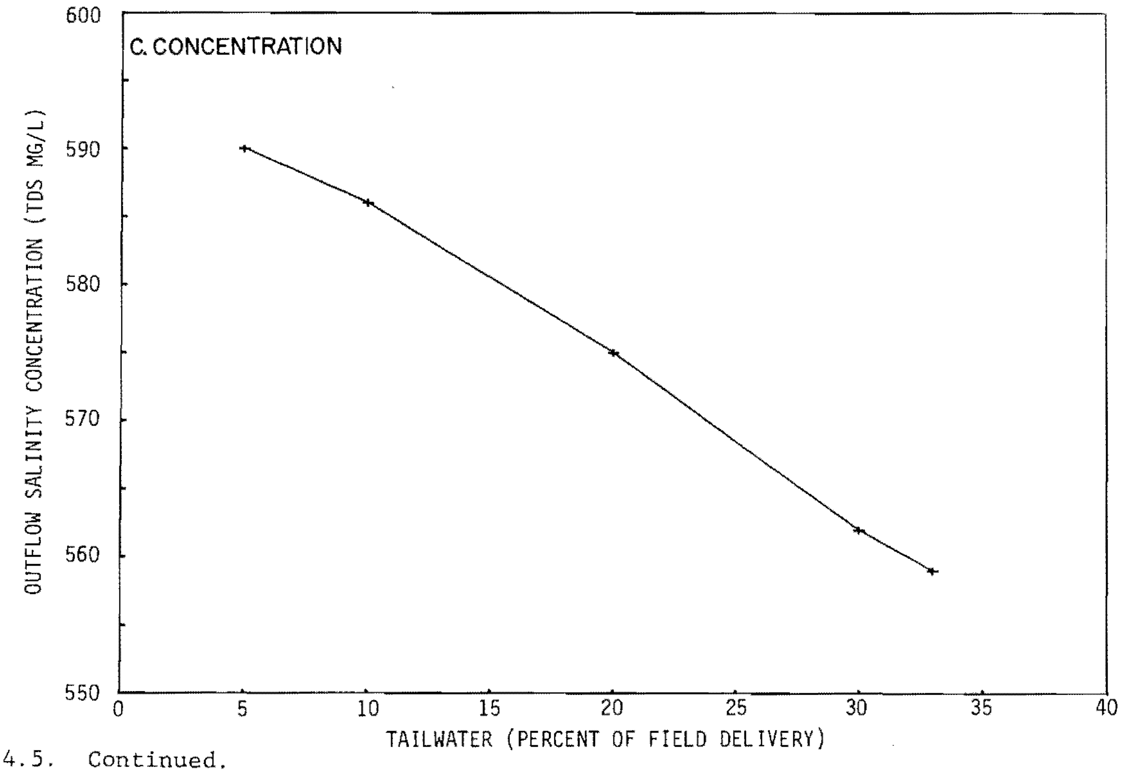


Figure 4.5. Continued.

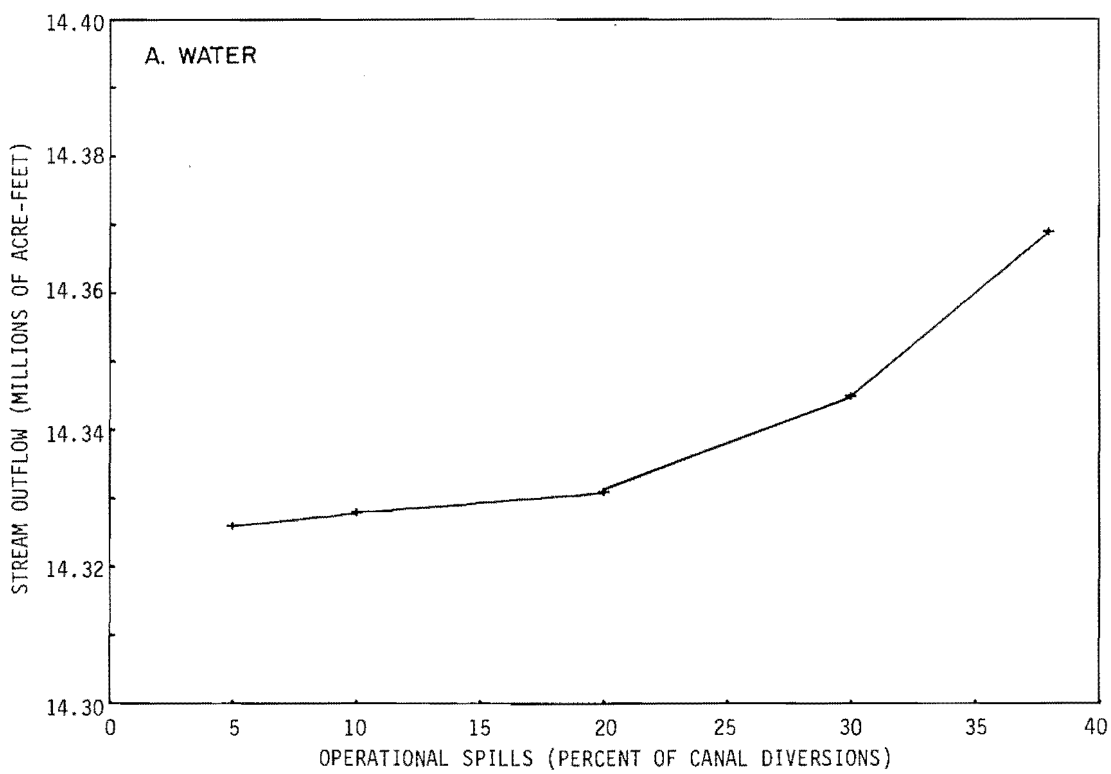


Figure 4.6. Sensitivity of a) stream outflow, b) salt outflow, and c) outflow salinity concentrations of the Colorado River below Grand Junction, Colorado, to operational spills. (Calibration based on component efficiencies adopted by the Grand Valley Salinity Coordinating Committee, i.e. conveyance efficiency = 33%.)

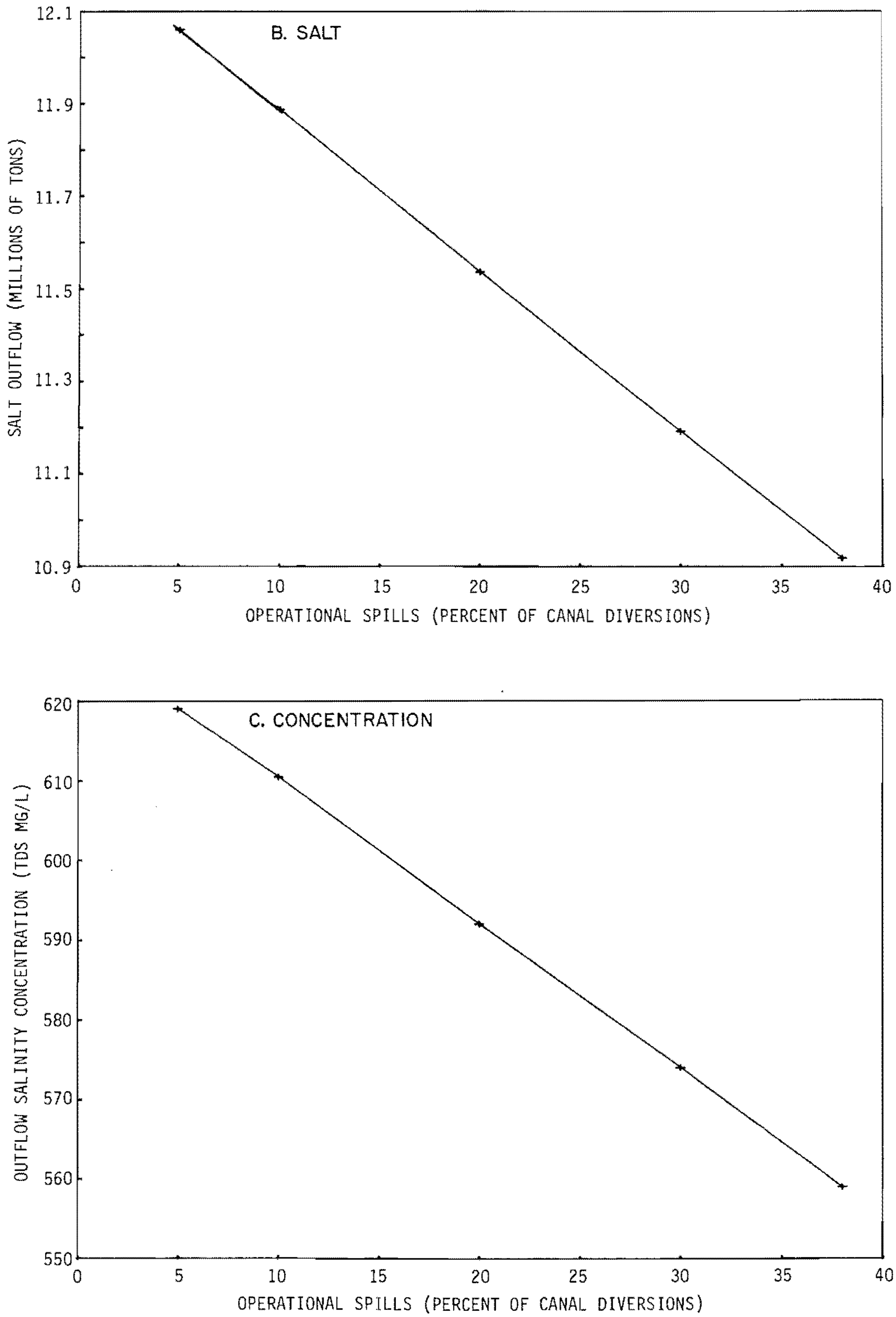


Figure 4.6. Continued.

Sensitivity to canal diversions

The response to reduction in canal diversions is shown in Figure 4.4. As was expected, a reduction in canal diversions resulted in a corresponding increase in river outflow. However, the salt outflow responded in an unexpected manner. Because less water was delivered to the farm when the diversions were reduced, it was expected that the salt loading would be correspondingly reduced. This proved true for a reduction of up to 30 percent. For greater than a 30 percent reduction, the salt outflow increased dramatically because of the concentrating effect of evapotranspiration on water in the root zone. With more than a 30 percent reduction in canal diversions, salt precipitated within the soil profile, soil moisture stress was placed on the crops resulting in reduced growth, and finally there was a net increase of salt loading in the stream.

Sensitivity to tail water runoff

The response sensitivity to the tail water runoff is shown in Figure 4.5. A reduction in tail water increased the amount of water infiltrated, resulting in more salt pickup from the soil profile and groundwater zones. The predicted results showed that by reducing tail water runoff from 33 percent to zero there would be an increase in TDS concentration of 31 mg/l during the 3-year period.

Sensitivity to operational spills

Waters spilled from the canal into the river were considered as operational spills. In practice, a reduction in operational spills would reduce the total diversions. Based on an assumption that the carrying capacity of the main canals and laterals is adequate, reductions in operational spills would deliver more water to the fields, resulting in more salt pickup. Reducing operational spills from 38 percent to 5 percent of the canal diversions increased the salt load by about 1,141,000 tons corresponding to an increase in TDS concentration of 60 mg/l. These results are shown in Figure 4.6.

Sensitivity of farm application efficiency

Since efficient farm application requires that the water stay in the root zone where the plants can use it later, application efficiency (EAP) is the ratio of the amount of water stored in the root zone to the amount of water applied. The efficiency is computed on the basis of a specified time interval. The model has the capability of computing the monthly and average annual values of EAP. Any change in the values of conveyance efficiency, canal diversions, operational spills, and tailwater runoff could change the total water applied to fields resulting in a corresponding change in EAP. The sensitivity study on the computed application efficiency was based on this concept. Figure 4.7 shows the sensitivity of EAP to corresponding changes in other parameters. The results showed that 1) EAP was more sensitive to reductions in operational spills than to increases in conveyance efficiency. Since the operational spills were larger than the seepage losses from the canals, a reduction in spills would provide more water to the farm which would result in a lower application efficiency. This result is rather trivial and of little practical value in the actual operation of the irrigation system. 2) EAP had an inverse relationship to changes in conveyance efficiency. Based on Figure 4.7, the average reduction in EAP was 7.0 percent for a 10 percent increase in conveyance efficiency. 3) There was no significant change in EAP for a reduction in canal diversions beyond 10 percent.

A comparison of results obtained from the sensitivity studies relating to changes in conveyance efficiency and canal diversions indicated that: 1) a 10 percent reduction in canal diversions would reduce streamflow concentrations by 7 mg/l (Figure 4.4), although the corresponding increase in application efficiency would be 4.5 percent (Figure 4.7). 2) The conveyance efficiency would have to be increased from 81 percent to 93 percent in order to achieve a similar reduction of 7 mg/l in streamflow concentration (Figure 4.3).

Sensitivity to system identification

A disturbing feature of the effort to use hydrosalinity modeling to examine the consequences of various irrigation water management options in the Grand Valley is that the results vary greatly with the published data used for model calibration (Table 4.1). Two distinctively different characterizations of the flow system and of the effects on the system of alternative management practices emerge from calibrating on the basis of different data sets. Both calibrations were made, and the results are compared below.

Calibration system 1 was based on the irrigation efficiency values used by the UWRL in the 1975 National Commission on Water Quality (NCWQ) assessment study. This system postulated values of 14 percent for the operational spills, 14 percent for the tail water runoff, and 76 percent for the canal conveyance efficiency. Calibration system 2, based on the Grand Valley Salinity Coordinating Committee data were 38 percent for the operational spills, 33 percent for the tail water runoff, and 81 percent for the canal conveyance efficiency. The two calibrations varied significantly in the values

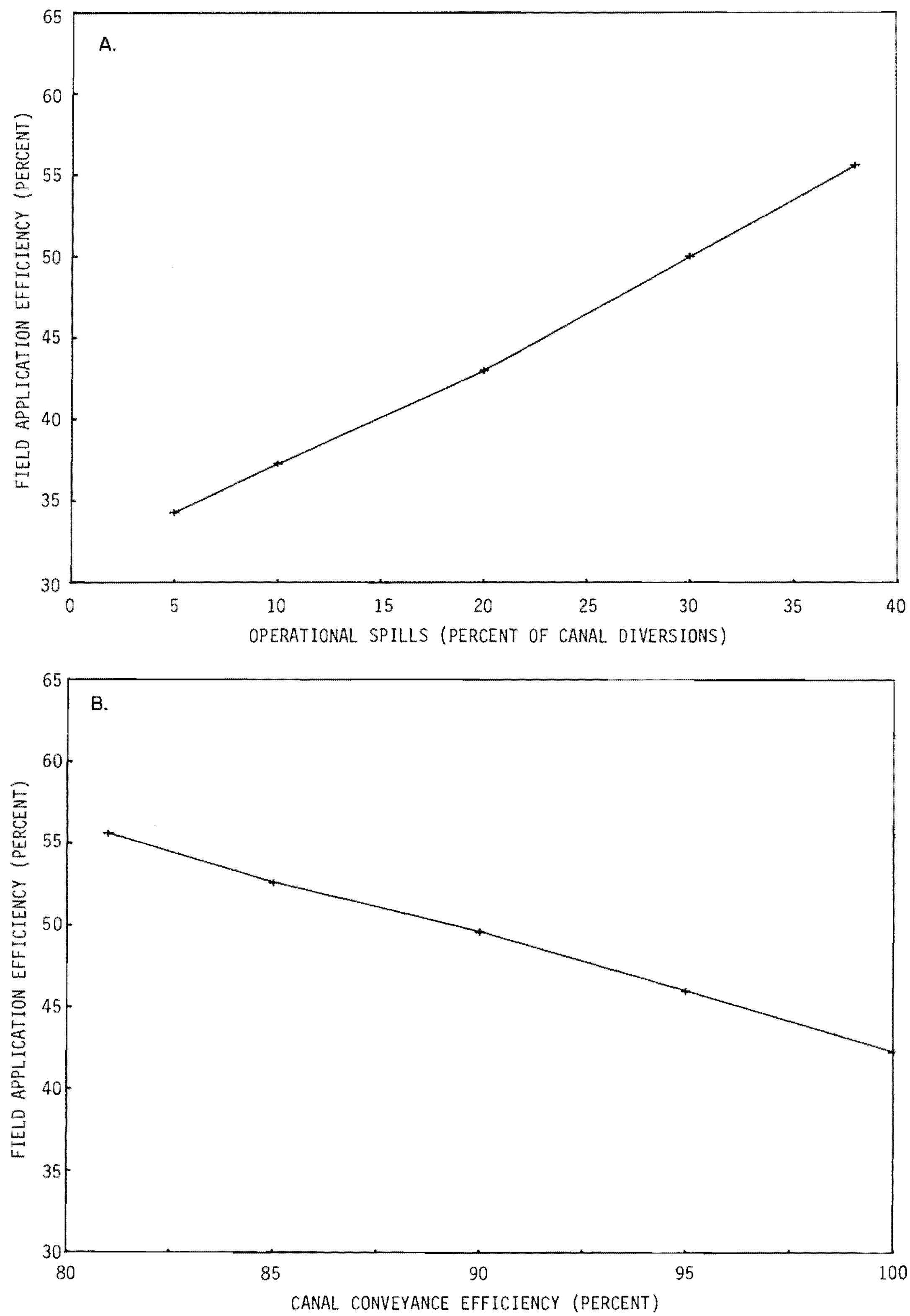


Figure 4.7. Sensitivity of the field application efficiency of the Grand Valley, Colorado, irrigation system to changes in a) operational spills, b) canal conveyance system efficiency, c) reduction in canal diversions, and d) tail water runoff. (Calibration based on component efficiencies adopted by the Grand Valley Salinity Coordinating Committee, i.e. conveyance efficiency = 81%, and tail water = 33%.)

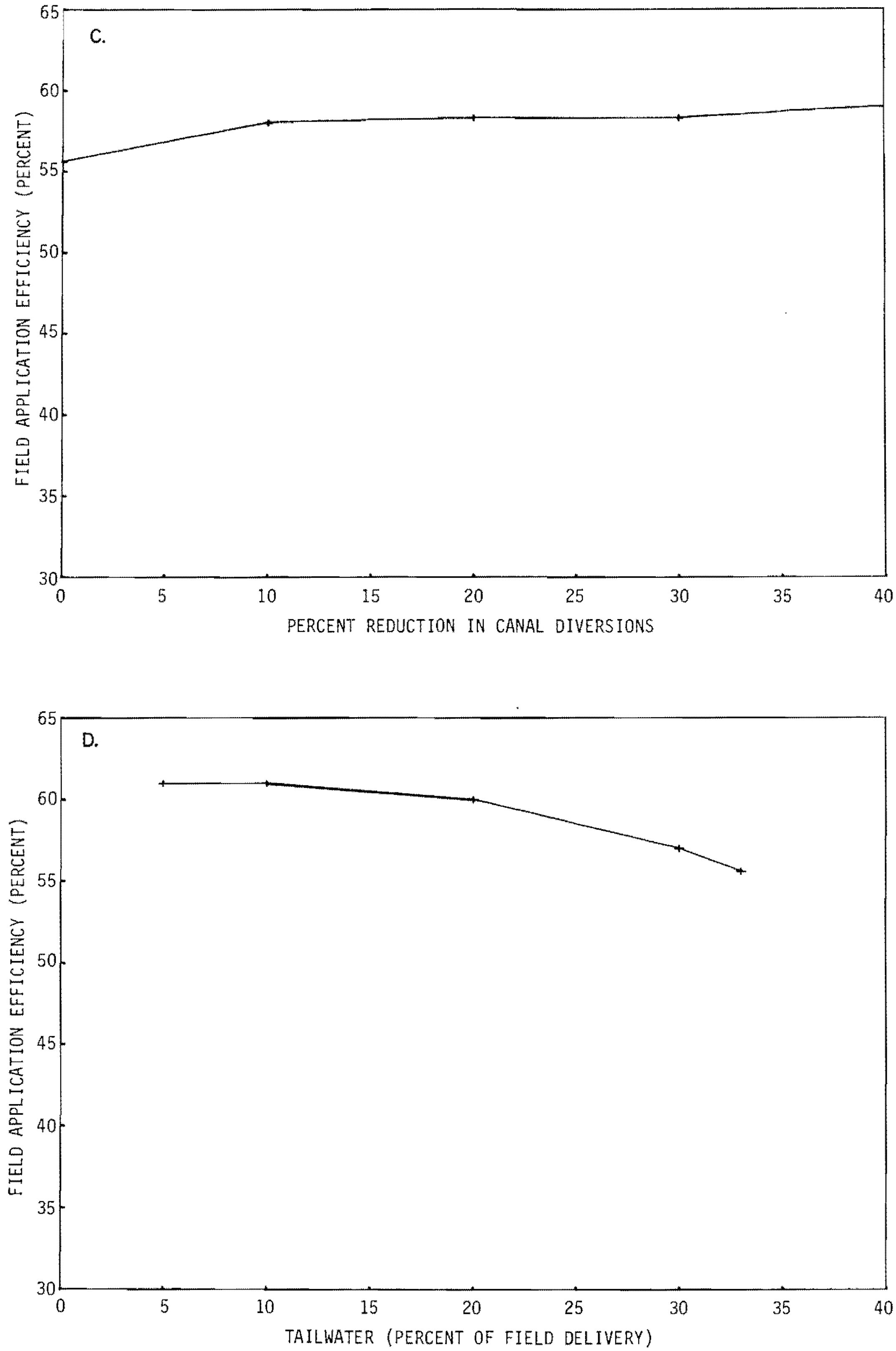


Figure 4.7. Continued.

of parameters (Table 4.5) needed to best fit the reported data. As shown in Figure 4.8, even though the two calibrations matched canal diversions exactly and consumptive use quite closely, other flows varied drastically. Particularly important for salinity management, these differences lead to opposite conclusions as to the dominant source of salt loading and what needs to be done by way of salinity control.

As a beginning, deep percolation and outflow concentration (TDS) were chosen to demonstrate system sensitivity to changes in conveyance efficiency and canal diversions. The results in Table 4.6 and Figures 4.9 and 4.10 show that:

1. A reduction in canal diversions had greater effect on System 1 than System 2, while an increase in conveyance efficiency had a greater impact on System 2 than on System 1. Thus, the water management method selected as more efficient depends on how the system was described in available data and identified by the modeler.

2. Reductions in operational spills and tail water runoff did have effects with the same directional trend in predicting outflow concentrations. However, the increase in streamflow concentrations due to these changes were consistently lower for System 1 than for System 2, because of the proportional increases in the corresponding flows.

Table 4.5. Variations in parameters from model calibration with data for system 1 and system 2, Grand Valley, Colorado.

#	Parameter Description	Units	Best fit value obtained for model calibration: System 1	System 2	Remarks
29	Proportion of groundwater outflow from basin		42	0	Refer to Fig. 4.8 for description of the systems
47	Unmeasured surface runoff		0	3 percent	
83	Salt pickup in DP	tons/ac-ft/mo.	3.0	2.6	
87	Salt pickup in canal seepage	tons/ac-ft/mo.			
93	Mineral weathering in groundwater	tons/acre/month	0	0.15	

Table 4.6. Relative sensitivity analysis of system 1 and system 2 to deep percolation and TDS concentration in the outflow, Grand Valley, Colorado.

Parameter: Description	Change in Value (percent)	Change in Outflow (1970-1972): System 1 DP water (ac-ft)	System 1 TDS (mg/l)	System 2 DP water (ac-ft)	System 2 TDS (mg/l)
Increase Conveyance Efficiency	76 to 100	+ 385,205	- 5		
	81 to 100			+ 201,956	- 9
Reduce Operational Spills	14 to 0		+ 20		
	38 to 0				+ 60
Reduce Canal Diversions	100 to 75	- 248,779	- 33		
	100 to 75			- 57,207	- 17
Reduce tailwater	14 to 0		+ 14		
	33 to 0				+ 31

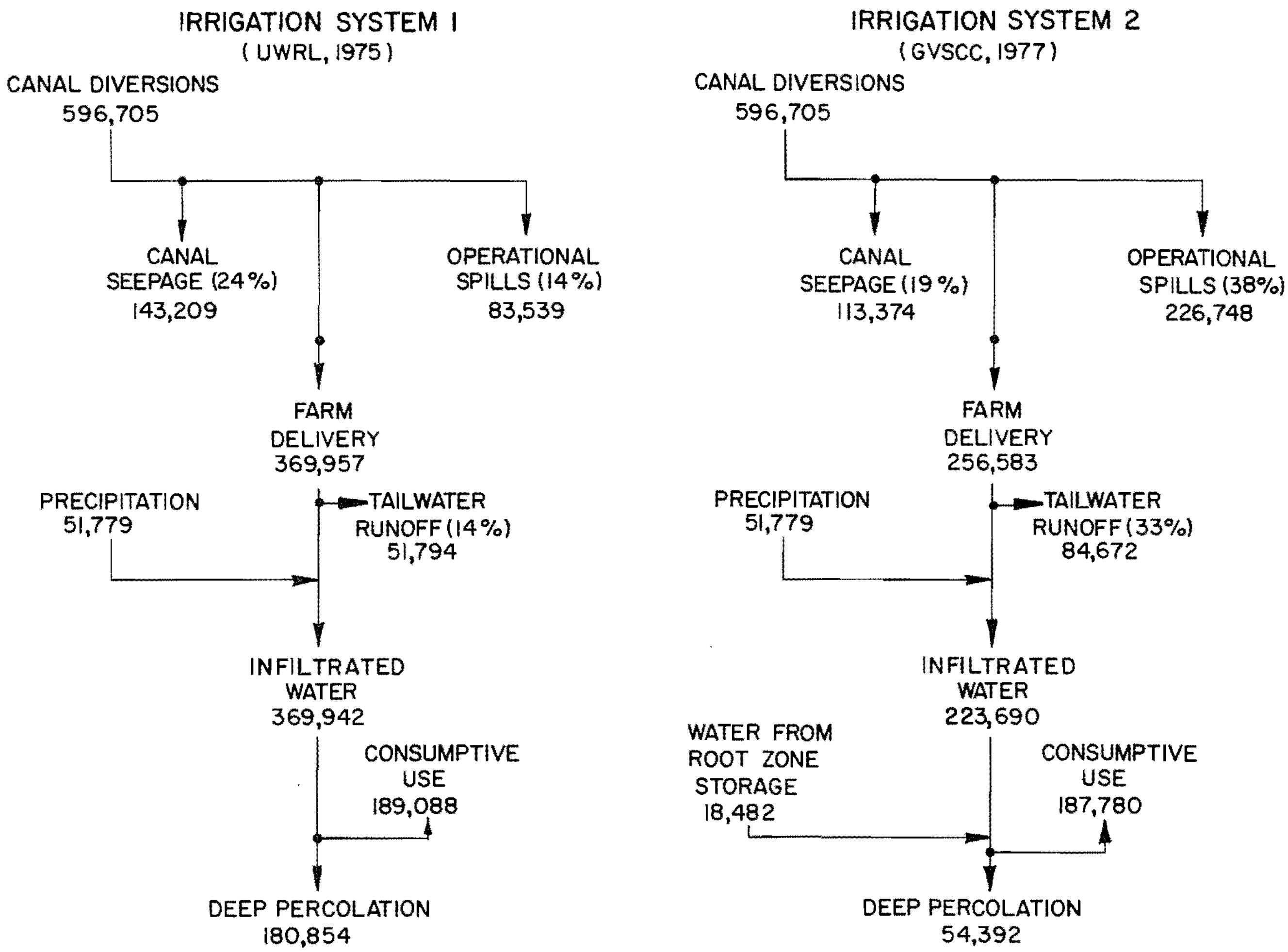


Figure 4.8. Schematic diagrams of two different possible representations of the Grand Valley, Colorado, irrigation system. (All numbers are in acre-feet.)

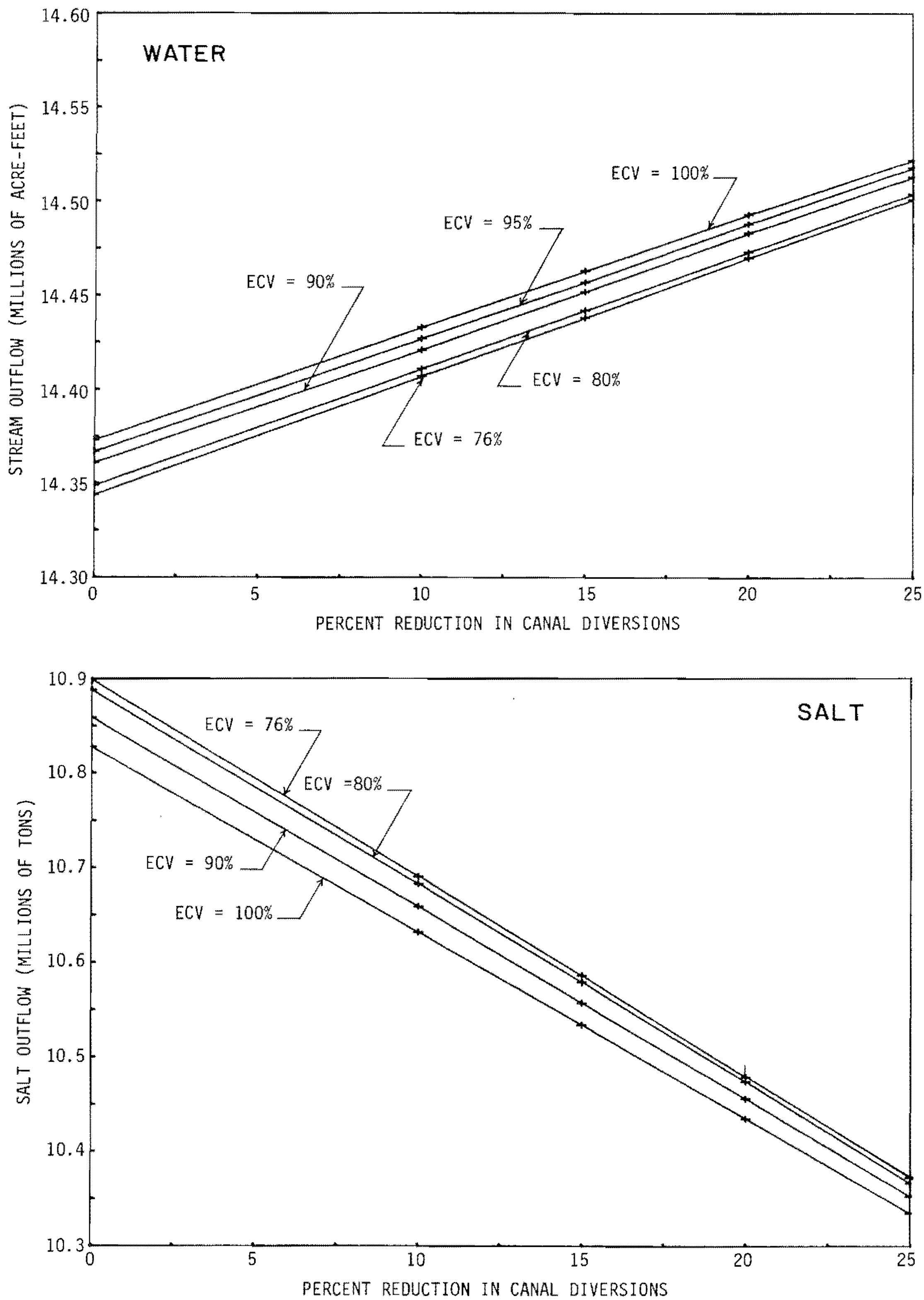
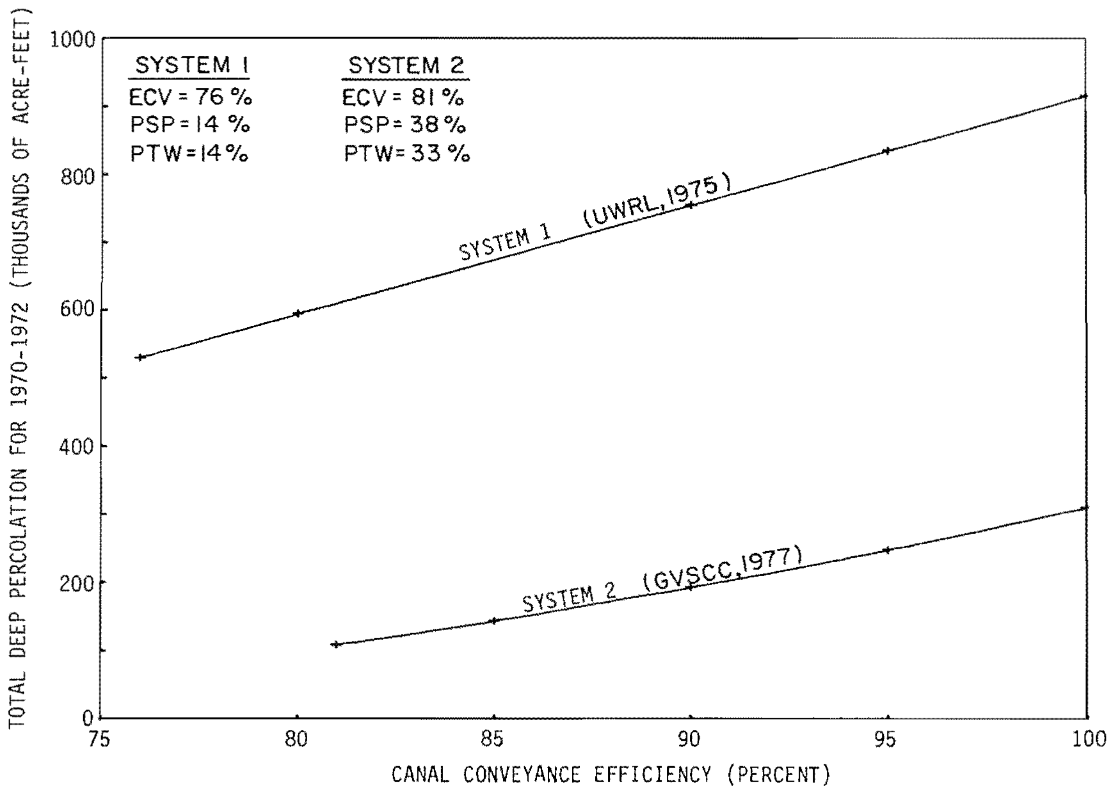


Figure 4.9. Interrelationship between a reduction of historical canal diversions for the 1970-1972 water years and an increase in canal conveyance efficiency for GV system 1. CGV system 1 assumes values of ECV, PSP, and PTW of 76%, 14% and 14%, respectively.)



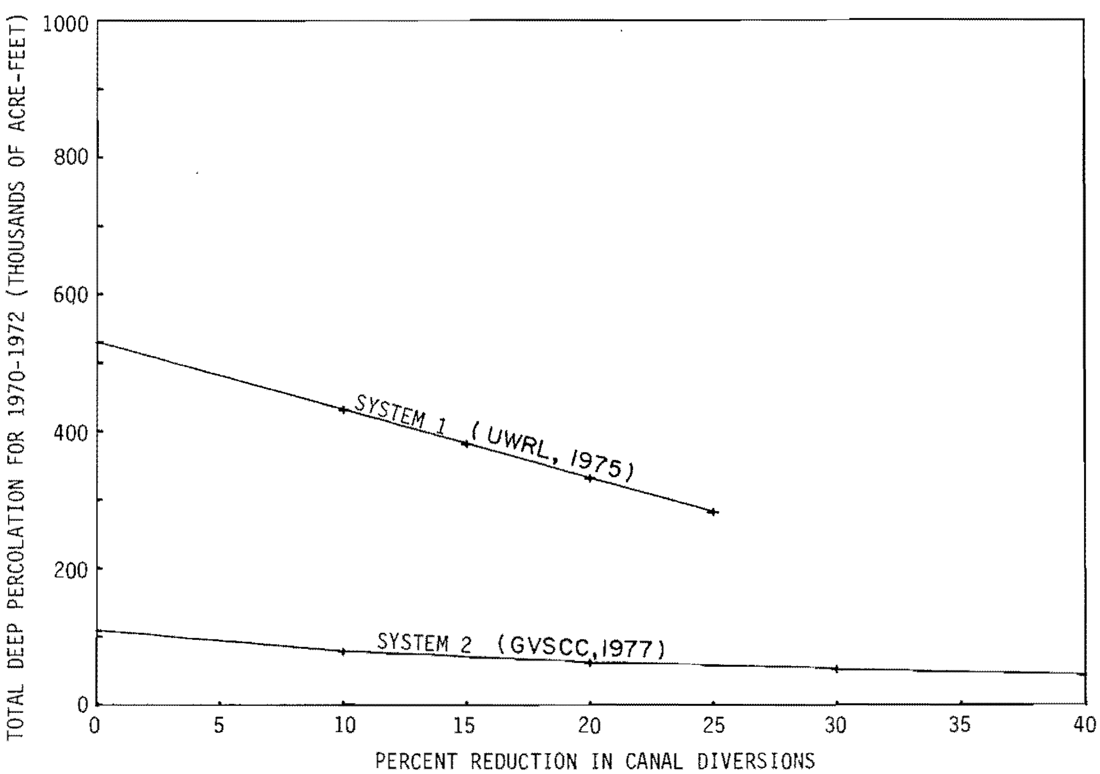


Figure 4.10. Relative sensitivity of system 1 and system 2, Grand Valley, Colorado, deep percolation to changes in canal conveyance efficiency and reductions in canal diversions.

CHAPTER 5.
MANAGEMENT STUDIES
WITH BSAM-SALT

Objective

The next goal of this study was to test the capability of the BSAM-SALT model to evaluate the possible impacts of irrigation water management alternatives on the quantity and quality of the receiving stream. A secondary objective was to demonstrate use of the model to simulate different irrigation management practices.

Procedure

The management studies were organized as outlined in Figure 5.1. First, a base was established from which the changes relating to salinity achieved by an alternative under study could be measured. The base consisted of the results generated by the hydrosalinity model using the parameter values obtained in calibrating the model to the irrigation management system described by the Grand Valley Salinity Coordinating Committee. The next step was to impose the selected management alternative on the model by changing the appropriate model parameters and then to evaluate the results. The specific objective was to evaluate the effect of improvements in canal lining as well as the on-farm application efficiency singly or in combination on the salinity loading in the Colorado River.

The formulation of the management runs requires a comprehensive understanding of the model both conceptually and as programmed, as well as of the irrigation system and practices to be simulated. For example, there is nothing in the model that may be set to explicitly perform the runs necessary to predict the change in salinity outflow from the area resulting from a change from flood to sprinkler irrigation methods. In order to do this, one must know which parameters are associated with the method of irrigation application. Very few of the parameters have a one-to-one correspondence with management options. More often, replication of a management option requires the setting of several parameters which interact together to produce the desired management condition. The salinity outflow response is not guaranteed to be unique; that is, the same salinity response may be obtained by several different sets of parameter values. Therefore, the correct interpretation of the results requires some judgment on the part of the user and often entails an examination of the internal salt concentrations generated by the run rather than just noting the bottom line showing the computed salt outflow amounts and concentrations. The following section derives some of the relationships that may be useful in performing the management studies.

Parametric Management Formulation

The major potential management application is to irrigated agriculture. The applicable model parameters include the canal diversions, CNL, operational spills, PSP, conveyance efficiency, ECV, tail water runoff, PTW, tail water evaporation, ECUTW and canal diversion adjustment coefficient, CNA. Figure 5.2 shows how these factors interact by tracing the flow of one unit of canal diversion through the conveyance system. A management alternative is simulated by modifying the values of the appropriate combination of these parameters. The following relationships have been derived in order to aid in making realistic and meaningful management simulations.

Conveyance System Management: Unchanged Farm Delivery

One approach to irrigation system management is to modify operation of the delivery system while staying with current on-farm water management practices. The practice preserves water rights intact and does not assume any control of the irrigation company over individual farmers. The farm delivery remains unchanged if:

$$CNL_1 \ (ECV_1 - PSP_1) = CNL_0 \ (ECV_0 - PSP_0) \quad (5.1)$$

where the variables are as previously defined, the subscripts 0 and 1 denote consecutive time periods. Equation 5.1 may be rewritten to define a canal diversion adjustment coefficient (CNA) as follows:

$$CNL_1 = CNA * CNL_0$$

where

$$CNA = \frac{ECV_0 - PSP_0}{ECV_1 - PSP_1} \quad . \ . \ . \ . \ . \ . \ . \ . \ . (5.2)$$

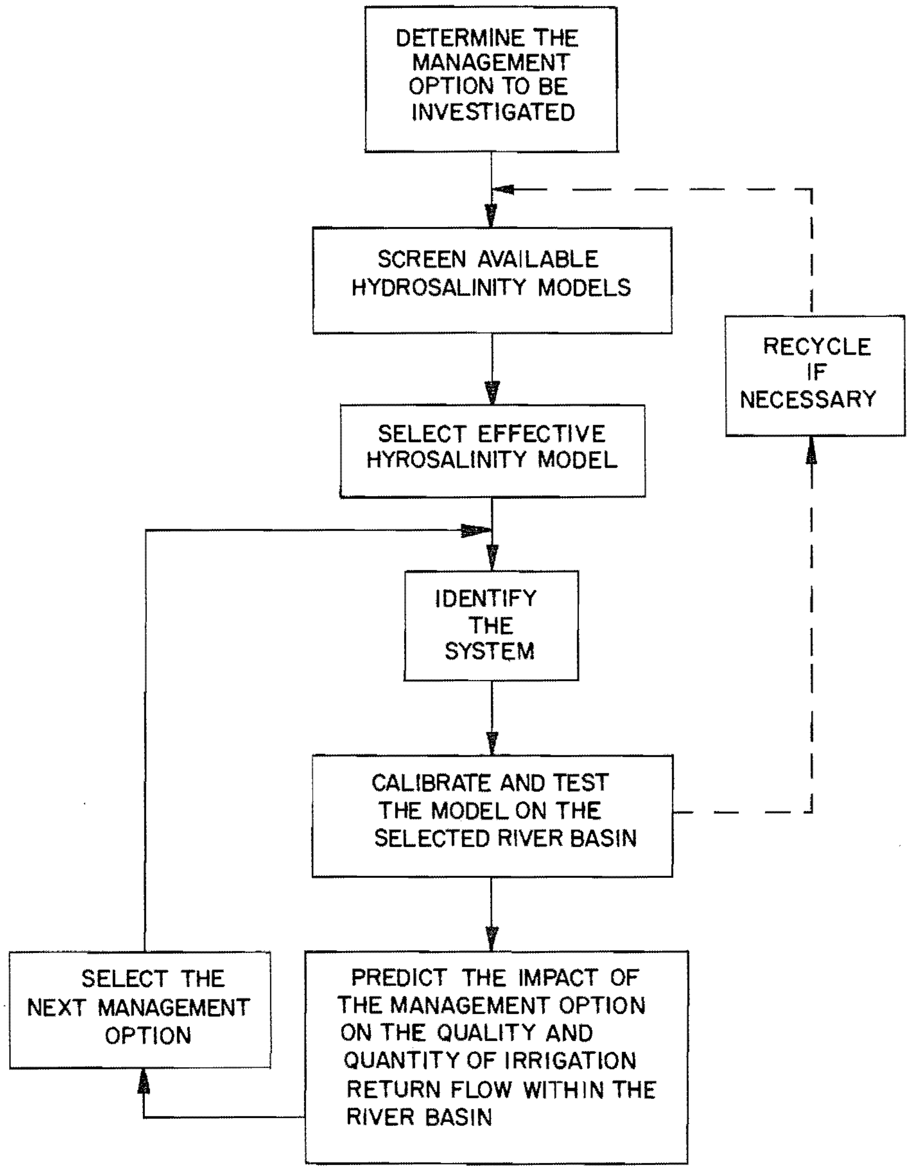


Figure 5.1. Simplified flow diagram for studying the effects of management options on the quality and quantity of irrigation return flows.

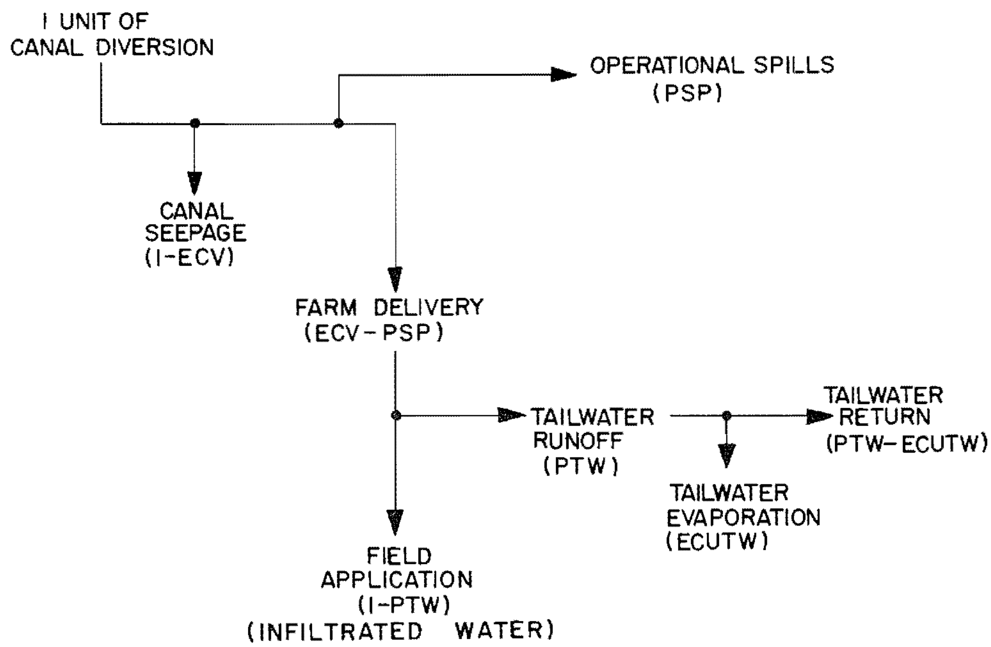


Figure 5.2. Flow schematic for one unit of irrigation canal water from its diversion to point of field application.

By changing the parameter CNA in accordance with Equation 5.2, the effects of changes in canal lining (conveyance efficiency) and system operation (operational spills) may be simulated quite easily.

Combination Conveyance and Irrigation System Management: Unchanged Field Application

A second approach to irrigation system management would be to hold field application rather than farm delivery constant. Again referring to Figure 5.2:

$$CNL_1\ (ECV_1 - PSP_1)\ (1 - PTW_1)$$

$$= CNL_0\ (ECV_0 - PSP_0)\ (1 - PTW_0) \quad \ldots\ (5.3)$$

solving for CNL_1, yields:

$$CNL_1 = CNB * CNL_0$$

or

$$CNB = \left(\frac{ECV_0 - PSP_0}{ECV_1 - PSP_1}\right)\left(\frac{1 - PTW_0}{1 - PTW_1}\right) \quad \ldots\ (5.4)$$

and all other variables are as previously defined. The effect of changing from one method of irrigation to another may be assessed by changing CNB in accordance with Equation 5.4 and by changing the proper model parameters to reflect the different irrigation practice. In general, sprinkler irrigation methods are characterized by little tail water runoff, with some evaporative losses which depend on temperature and wind. The model parameters which simulate these conditions are PTW and ECUTW, respectively.

Computed Canal Diversions for Management

A final management mode programmed into the model calculates the monthly canal diversion required to satisfy potential crop evapotranspiration (PET), the various operating system losses (ECV and PSP), a root zone soil moisture level to be maintained (CMS), and an overall composite field application efficiency (EAP). The equation for month i is as follows:

$$CNL_i = QSL_i + \frac{PET_i + (CMS_i - SMI_i) - RPSM_i\ (1 - CDP)}{EAP\ (ECV - PSP)} \quad \ldots\ (5.5)$$

where:

CNL_i is the computed canal diversion required

QSL_i is a specified diversion required for salt leaching

PET_i is the potential crop evapotranspiration

CMS_i is the soil moisture level specified to be maintained

SMI_i is the initial soil moisture level

$RPSM_i$ is the rain plus snow melt

CDP is the proportion of soil moisture storage above a critical level that deep percolates

EAP is a pseudo irrigation field application efficiency

ECV is the canal conveyance efficiency

PSP is the proportion of canal diversion that result in operational spills

i is the month

The parameter, EAP, can be considered a pseudo efficiency. The actual field application efficiency is defined as the proportion of irrigation water delivered to the field that enters the root zone and remains available for crop consumptive use. The resulting interaction between soil properties, climatic conditions, and farm management practice causes the short term field application efficiency to fluctuate quite widely throughout the year. Therefore, it is not treated as a constant, but is evaluated by the model and becomes part of the simulation output. When using the parameter EAP to estimate the canal diversion, the computed application efficiency will fluctuate much less than otherwise and will approach the value specified as EAP.

The canal diversion determined by this operating option may not correspond to the actual diversion record because it does not consider any restrictions on diversions that may be imposed by water rights constraints or the common practice of operating managers to change the diversion according to some perceived or forecast climatic conditions. However, it is a very useful option where actual diversion records are not available.

Management Studies

Eight alternative management strategies were used to test the model. Each alternative was designed to test a particular aspect of irrigation management. Over 50 simulations were run and the results are summarized below along with the descriptions of the various alternatives considered. The first five management alternatives maintained fixed levels of field application while the last three maintained a specified level of soil moisture.

Management Alternative 1 (MA-1)

The first management strategy was designed to assess the response of the system to canal lining while delivering the same amount of water to the farms. The operational parameters characterizing operational spills (PSP) and tail water runoff (PTW) were kept at the same levels as in the base calibration run. The diversion adjustment value, CNA, was calculated by Equation 5.2. The effect of alternative degrees of lining of the canals was simulated by adjusting the conveyance efficiency from the calibrated value of 81 percent to 100 percent. The results from Management Alternative 1 are given in Table 5.1 and are shown in Figure 5.3. This set of simulations indicates that the canal lining may be very effective in reducing the salt load of the Colorado River. Simulated salt concentrations decreased from 566 to 520 mg/l, a total of 46 mg/l, the water outflow increased 10,000 acre-feet, and the salt outflow decreased 897,000 tons during the 3-year period 1970-1972.

Management Alternative 2 (MA-2)

The second management strategy was also designed to assess the effect of increasing conveyance efficiency while delivering the

Table 5.1. Parameter values and 1970-1972 cumulative response summary for Management Alternative 1: line canals and maintain farm delivery at base line level.

	Parameter settings					Outflow responses - 1970-1972			
Run	PSP	ECV	CNA	PTW	ECUIW	Actual EAP	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1 (Base)	0.38	0.81	1.000	0.33	0	56	14,370	[illegible] 055	566.1
13	"	0.85	0.915	"	"	56	14,372	10,799	552.9
14	"	0.90	0.827	"	"	56	14,375	10,535	539.2
15	"	0.95	0.754	"	"	56	14,378	10,318	528.0
16	"	1.00	0.694	"	"	56	14,380	10,158	519.8

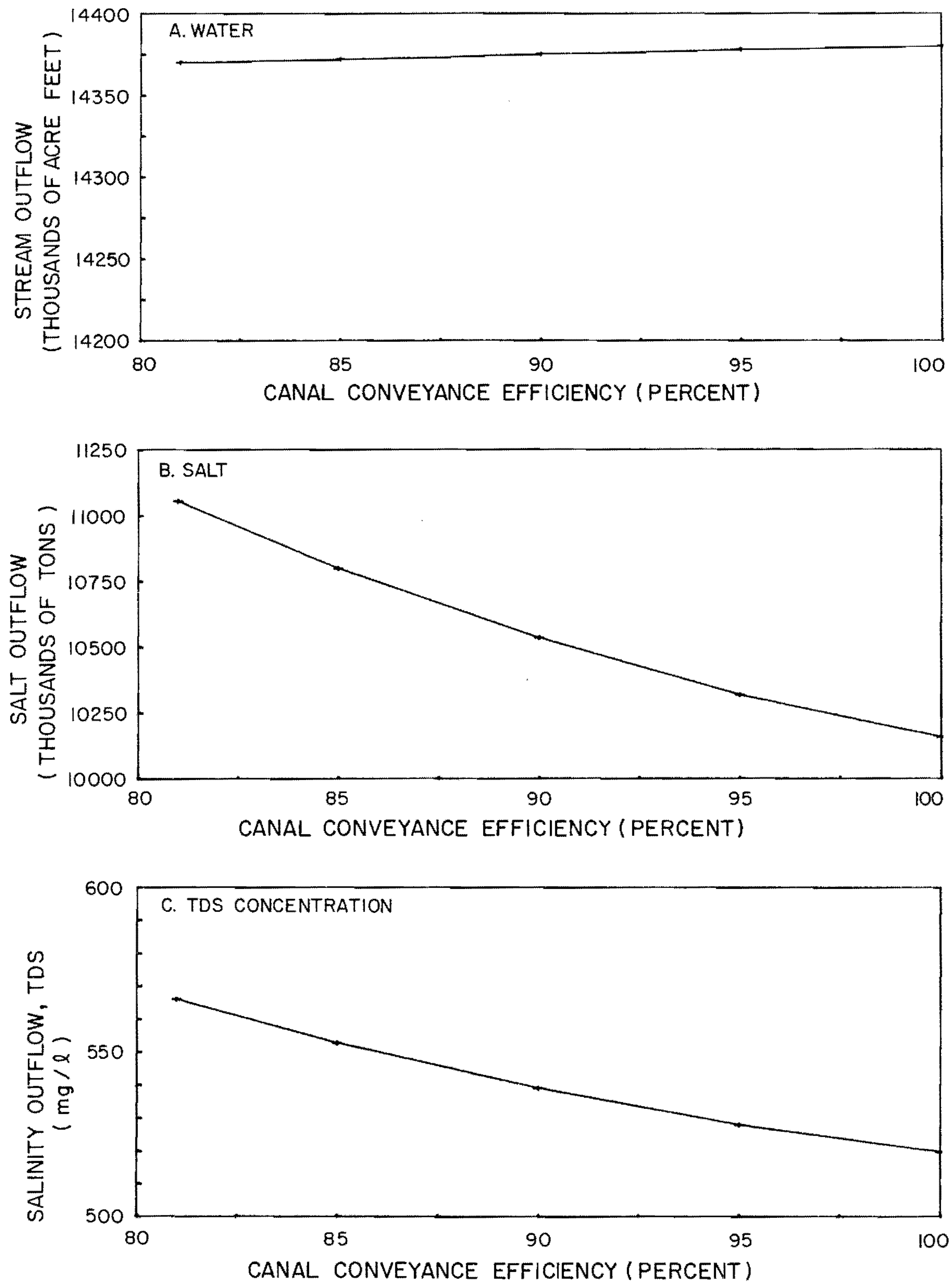


Figure 5.3. Predicted outflow response for period 1970-1972 for Management Alternatives 1 and 2 of a) water, b) salt, and c) salinity concentration (TDS) to lining of canals. Farm delivery unchanged from base level.

same amount of water to the farms. In addition, it projected a reduction in the operational spill percentage from 38 percent to 20 percent, to reflect improved conveyance system management. The diversion adjustment value, CNA, was calculated by Equation 5.2. The results of simulating this strategy are essentially identical to those resulting from the first alternative tested and are summarized in Table 5.2. Over the 3-year period, 1970-1972, the water outflow increased about 10,000 acre-feet, the salt outflow decreased 896,000 tons, and the TDS concentration decreased 46 mg/l.

Management Alternative 3 (MA-3)

Management Alternative 3 was designed to assess the response to the system to changes in the method of field application from flood to sprinkler irrigation. The diversions were maintained at the base level but operational spills were increased and tail water runoff was decreased in order to maintain the field application rate at the baseline level. The increase in operational spills required to maintain the field application rate at the base level was calculated by solving Equation 5.4 for PSP_1 as follows:

$$PSP_1 = ECV_1 - \frac{(ECV_0 - PSP_0)(1 - PTW_0)}{CNA\ (1-PTW_1)} \quad \ldots\ldots (5.6)$$

The move to sprinklers was simulated by changing the tail water runoff (PTW) and tail water evaporation (ECUTW) coefficients from 0.33 to 0 and 0 to 0.50 respectively. The results are given in Table 5.3 and Figure 5.4 and were somewhat unexpected. It was anticipated that an increase in application efficiency would decrease salinity. Instead, there was a slight (1 mg/l) increase caused by the concentrating effect of the evaporative losses from the sprinkler irrigation method. The sprinklers brought an increase in application efficiency by making it possible to decrease farm delivery even though the amount of water applied to the fields was held constant. Since the bulk of the salt pickup comes from the percolating water which remained constant because the field application was held constant, the volume of salt outflow also remained constant. Thus, an increased application efficiency does not necessarily reduce the salinity loading of the stream caused by irrigation return flows.

Table 5.2. Parameter values and 1970-1972 cumulative response summary for Management Alternative 2: line canals, reduce operational spills, and maintain farm delivery at base line level.

	Parameter settings					Outflow responses - 1970-1972			
Run	PSP	ECV	CNA	PTW	ECUTW	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1 (Base)	0.38	0.81	1.000	0.33	0	56	14,370	11,055	566.1
8	0.20	0.81	0.705	"	"	56	14,373	10,784	552.1
9	"	0.85	0.662	"	"	56	14,374	10,615	543.4
10	"	0.90	0.614	"	"	56	14,377	10,430	533.8
11	"	0.95	0.573	"	"	56	14,379	10,276	525.8
12	"	1.00	0.538	"	"	56	14,380	10,159	519.8

Table 5.3. Parameter values and 1970-1972 cumulative response summary for Management Alternative 3.

	Parameter settings					Outflow responses - 1970-1972			
Run	PSP	ECV	CNA	PTW	ECUTW	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1 (Base)	0.380	0.81	1.00	0.33	0	56	14,370	11,055	566.1
17	0.398	"	"	0.30	0	58	14,369	11,056	566.1
18	0.426	"	"	0.25	0.10	64	14,353	11,055	566.7
19	0.450	"	"	0.20	0.20	70	14,344	11,055	567.1
20	0.471	"	"	0.15	0.30	75	14,342	11,056	567.2
21	0.490	"	"	0.10	0.40	79	14,347	11,055	567.0
22	0.507	"	"	0.05	0.50	81	14,356	11,055	566.6

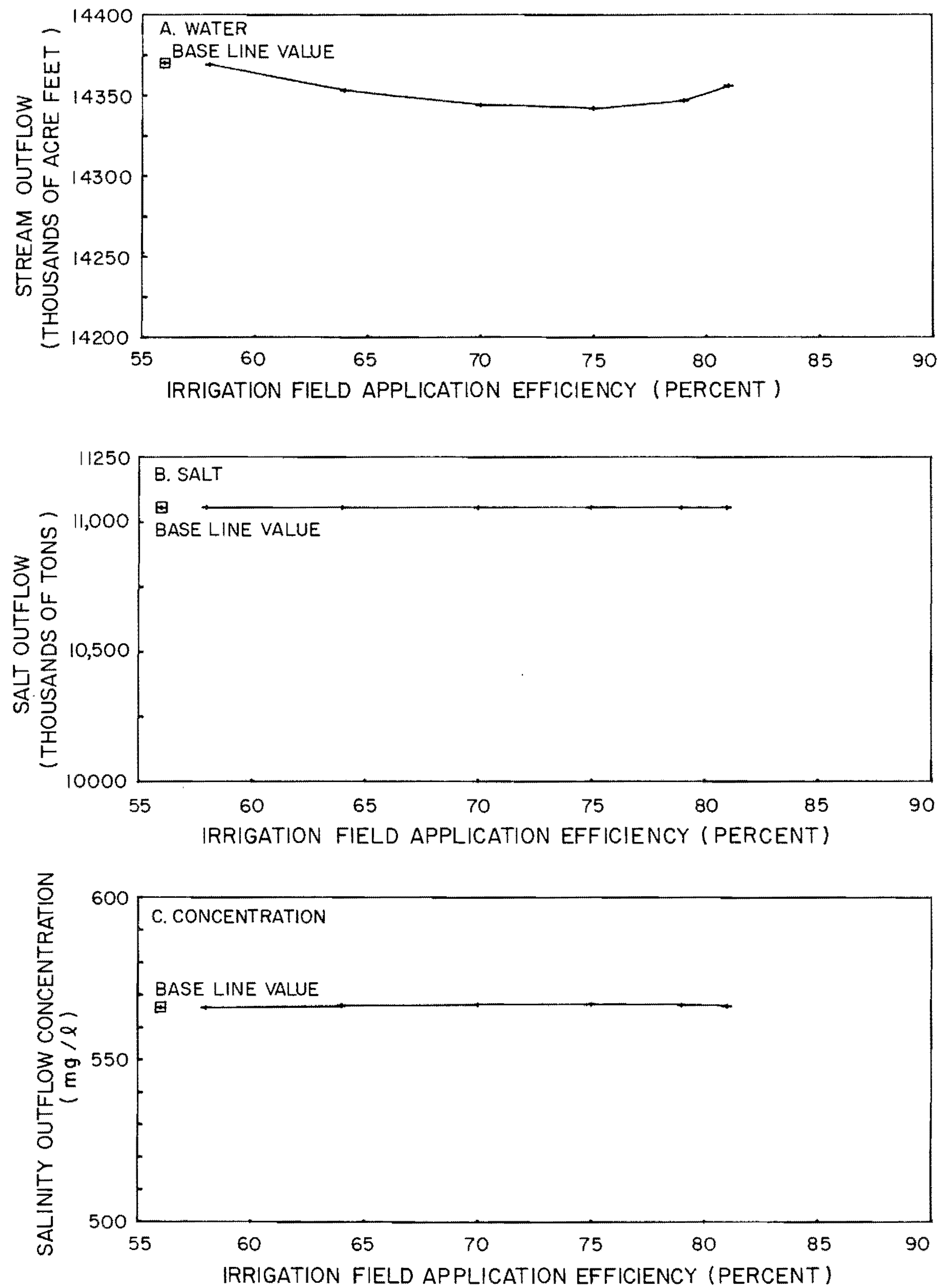


Figure 5.4. Predicted outflow response for period 1970-1972 for Management Alternative 3 of a) water, b) salt, and c) salinity concentration (TDS) to changing from flood to sprinkler irrigation. Field application rate unchanged from base level.

Management Alternative 4 (MA-4)

The fourth strategy was designed to assess system response to a policy of reducing the diversions through more efficient delivery and application while keeping the field application amount unchanged. This was simulated by changing PSP according to Equation 5.6. The results are given in Table 5.4 and Figure 5.5. The predicted reduction in salt loading is 491,000 tons of salt and a TDS reduction of about 24 mg/l. The water outflow is also reduced by about 23,000 acre-feet.

Management Alternative 5 (MA-5)

This alternative was designed to assess the effect of lining the canals, reducing operating spills, and reducing the diversions while maintaining a constant field application rate. The simulation was accomplished by increasing the conveyance efficiency and calculating the diversion adjustment, CNA, required to maintain the field application rate by Equation 5.4. The results of applying Management Alternative 5 are given in Table 5.5 and Figure 5.6. All three 1970-1972 response outflows were reduced. The water outflow was reduced 18,000 acre-feet, the salt outflow by 897,000 tons and the TDS concentration by 45 mg/l.

Management Alternative 6 (MA-6)

This alternative was designed to identify an operating strategy that could reduce the salinity impact on the river while maintaining the proportion of operating spills and tail water runoff at their calibrated values. This was accomplished by setting the minimum leaching water requirement (QSL) to zero and using Equation 5.5 to compute the required diversion to maintain soil moisture at field capacity and also satisfy potential crop use. The results are summarized in Table 5.6 and Figure 5.7. In order to achieve the reduction in salinity predicted by this set of runs, the canal diversions would have to be reduced as shown

Table 5.4. Parameter values and 1970-1972 cumulative response summary for Management Alternative 4.

Run	Parameter settings					Outflow responses - 1970-1972			
	PSP	ECV	CNA	PTW	ECUTW	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1 (Base)	0.380	0.81	1.00	0.33	0	56	14,370	11,055	566.1
20	0.471	"	1.00	0.15	0.30	75	14,342	11,056	567.2
24	0.433	"	0.90	"	"	74	14,343	10,964	562.5
25	0.386	"	0.80	"	"	74	14,344	10,871	557.6
26	0.326	"	0.70	"	"	75	14,345	10,779	552.9
27	0.245	"	0.60	"	"	75	14,346	10,686	548.1
28	0.132	"	0.50	"	"	75	14,347	10,564	541.8

Table 5.5 Parameter values and 1970-1972 cumulative response summary for Management Alternative 5.

Run	Parameter settings					Outflow responses - 1970-1972			
	PSP	ECV	CNA	PTW	ECUTW	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1 (Base)	0.38	0.81	1.000	0.33	0	56	14,370	11,055	566.1
51	0.15	0.81	0.516	0.15	0.30	75	14,346	10,614	544.4
29	"	0.85	0.484	"	"	75	14,348	10,485	537.7
30	"	0.90	0.452	"	"	75	14,350	10,354	530.9
31	"	0.95	0.424	"	"	74	14,351	10,244	525.2
32	"	1.00	0.399	"	"	74	14,352	10,158	520.8

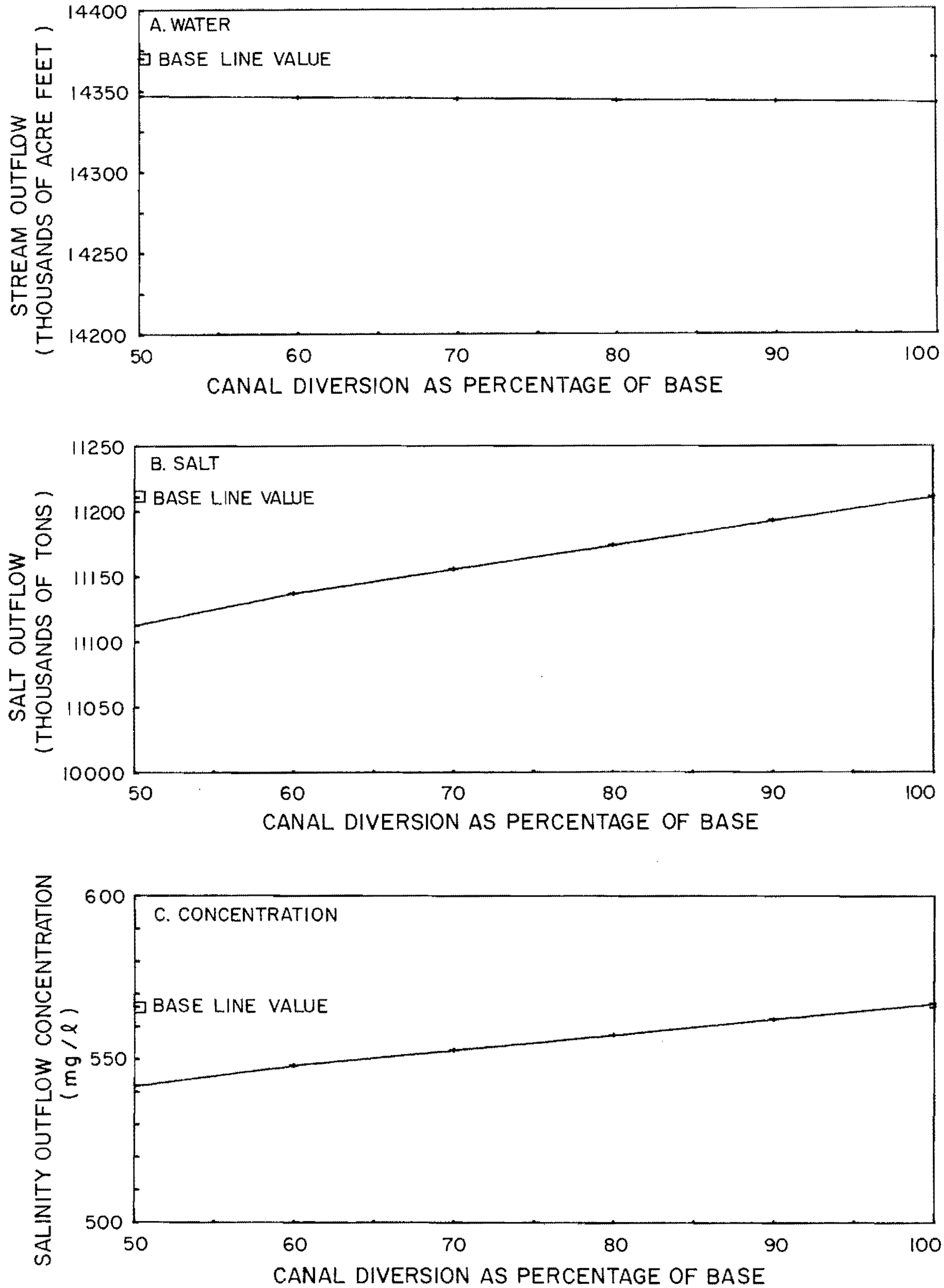


Figure 5.5. Predicted outflow response for period 1970-1972 for Management Alternative 4 of a) water, b) salt, and c) salinity concentration (TDS) to changes in canal diversions while maintaining the field application rate constant.

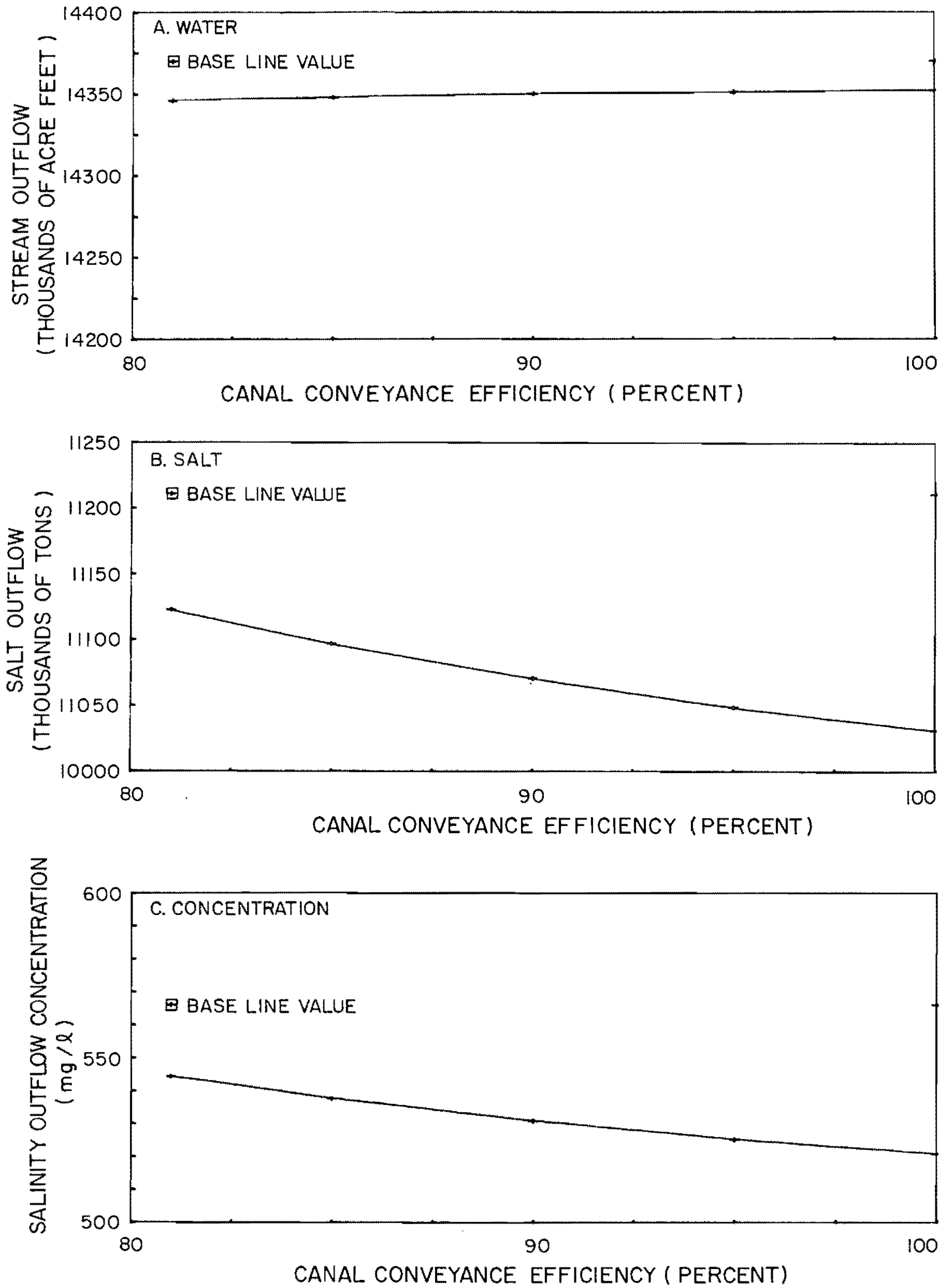


Figure 5.6. Predicted outflow response for period 1970-1972 for Management Alternative 5 of a) water, b) salt, and c) salinity concentration (TDS) to changes in canal diversions and lining of canals while maintaining the field application rate constant.

Table 5.6. Parameter values and cumulative response summary for Management Alternative 6.

Run	PSP	ECV	PTW	ECUIW	EAP	Diversion as percent of base	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
	Parameters					Outflow response - 1970-1972				
1	0.38	0.81	0.33	0	0.56	100.0	56	14,370	11,055	566.1
33	"	"	"	"	0.30	188.5	33	14,315	13,058	671.2
34	"	"	"	"	0.40	148.8	42	14,322	12,143	622.9
35	"	"	"	"	0.50	121.7	52	14,328	11,502	590.7
36	"	"	"	"	0.60	102.8	61	14,332	11,011	565.3
37	"	"	"	"	0.70	99.4	63	14,336	10,920	560.5
38	"	"	"	"	0.80	97.9	63	14,339	10,885	558.6
39	"	"	"	"	0.90	96.5	63	14,343	10,835	555.8

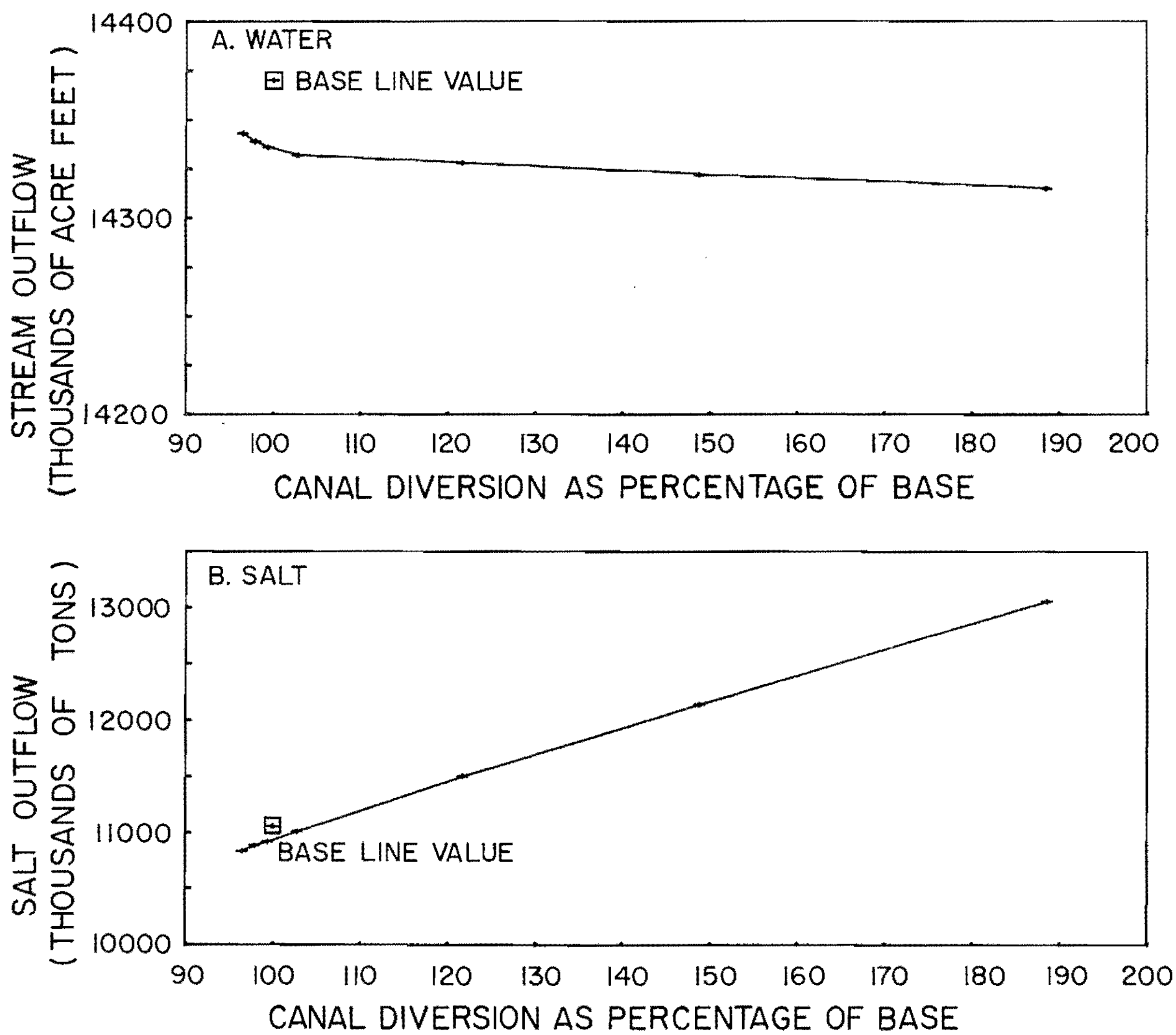


Figure 5.7. Predicted outflow response for period 1970-1972 for Management Alternative 6 of a) water, b) salt, c) salinity concentration (TDS), and d) application efficiency to changes in canal diversions.

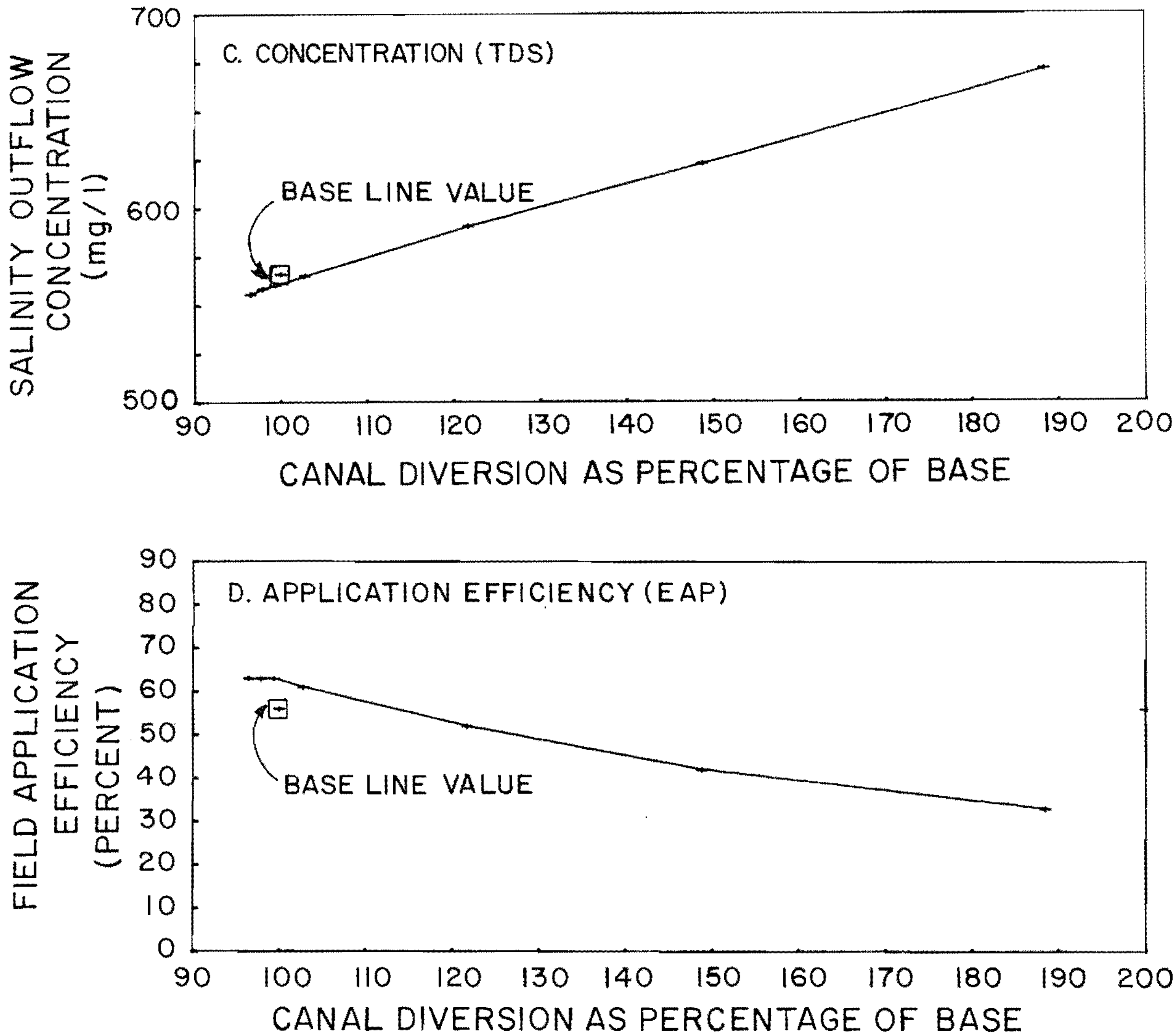


Figure 5.7. Continued.

in Figure 5.8. The salt load could then be reduced 220,000 tons or about 10 mg/l over the 3-year period without changing any irrigation practices. The total diversions were actually about 3.5 percent less than the historical diversions, but water rights constraints may not allow the diversion pattern to be altered.

Management Alternative 7 (MA-7)

This alternative strategy was similar to MA-6 except that the operational spills were reduced from 38 percent to 15 percent and the tail water runoff was reduced from 33 percent to 15 percent. This would simulate the effect of a partial conversion to sprinkler irrigation methods and more careful management of the water delivery system. The results are summarized in Table 5.7 and Figures 5.9 and 5.10. They illustrate that with a total reduction of almost 50 percent in diversions, a net reduction in salt loading of 567,000 tons or 27 mg/l could be achieved by modifying the pattern of diversions to correspond to Figure 5.10. This would also require improving the on-farm application efficiency to 83 percent by reducing the tail water runoff from 33 percent to 15 percent. This should not involve any water rights problems because the diversions are less than those made historically.

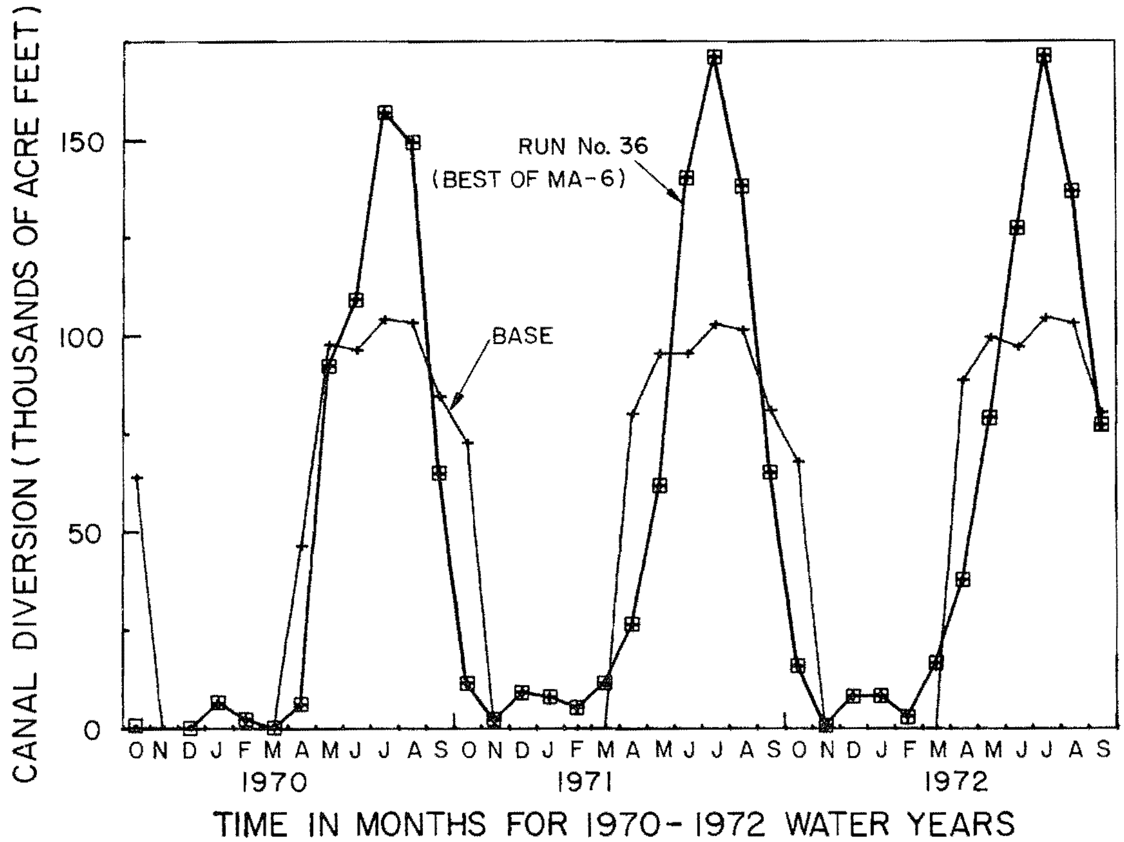


Figure 5.8. Comparison of canal diversions for the period 1970-1972 for the Grand Valley, Colorado, area between the base calibration and the best run (# 36) from Management Alternative 6.

Table 5.7. Parameter values and cumulative response summary for Management Alternative 7.

	Parameters					Predicted response - 1970-1972				
Run	PSP	ECV	PTW	ECUTW	EAP	Diversion as percent of base	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1	0.38	0.81	0.33	0	0.56	100.0	56	14,370	11,055	566.1
40	0.15	0.81	0.15	0.30	0.30	130.7	35	14,239	13,443	694.7
41	"	"	"	"	0.40	98.3	46	14,268	12,288	633.7
42	"	"	"	"	0.50	78.8	56	14,286	11,585	596.7
43	"	"	"	"	0.60	65.9	66	14,298	11,109	571.7
44	"	"	"	"	0.70	56.9	76	14,307	10,752	553.0
45	"	"	"	"	0.80	51.5	83	14,312	10,505	540.1
46	"	"	"	"	0.90	51.0	83	14,315	10,488	539.1

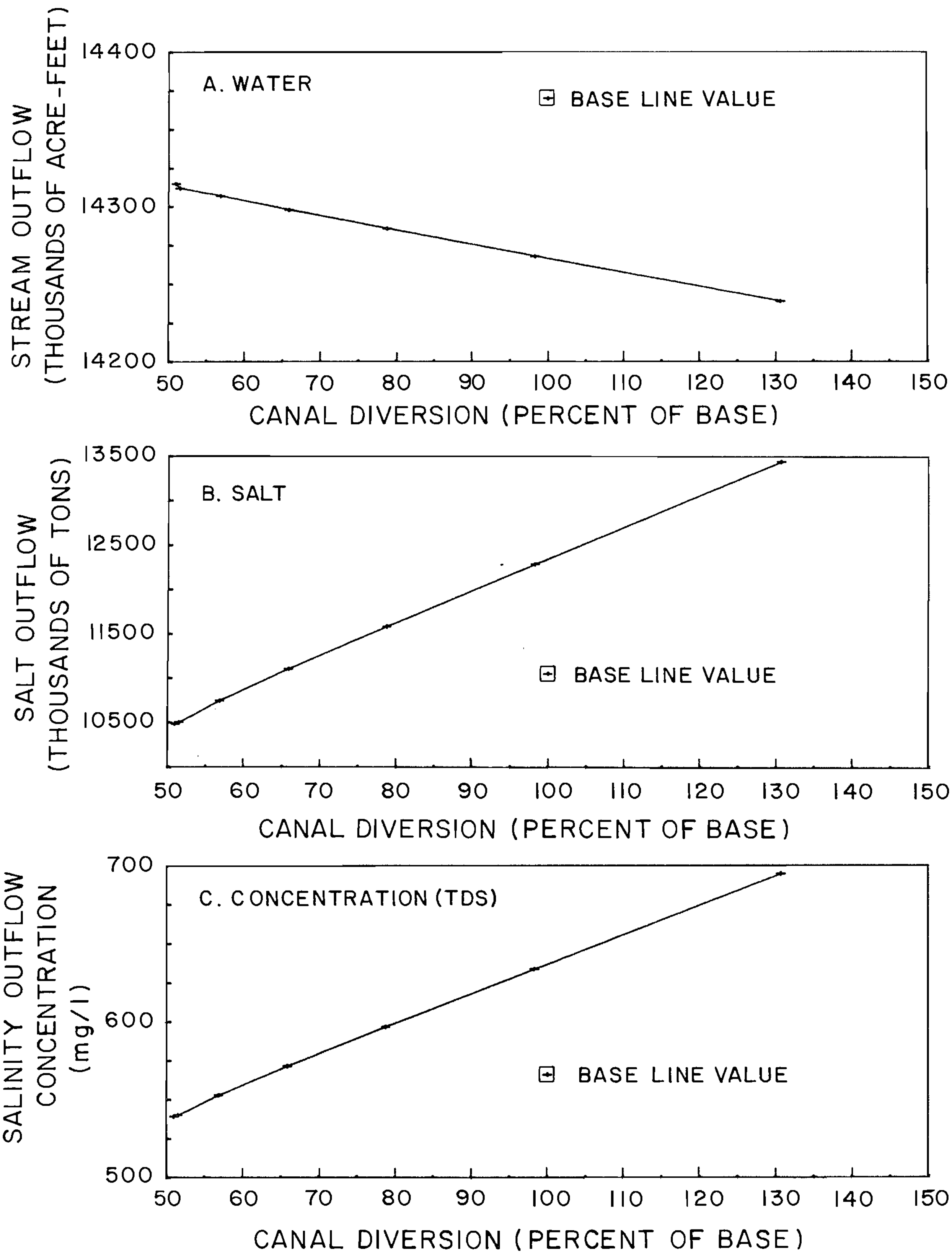


Figure 5.9. Predicted outflow response for period 1970-1972 for Management Alternative 7 of a) water, b) salt, c) salinity (TDS) and d) application efficiency to changes in canal diversions and EAP.

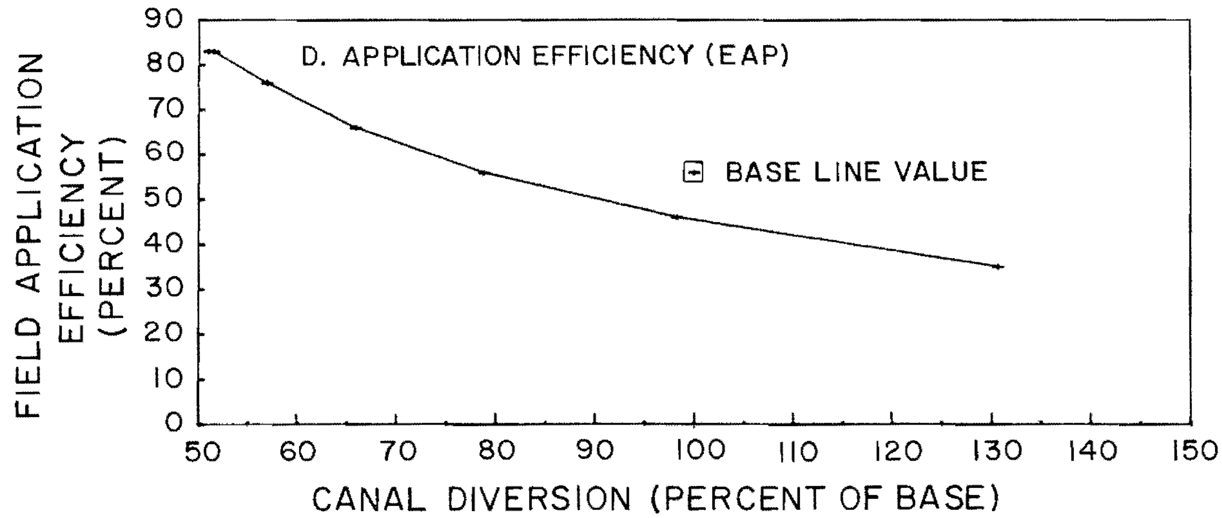


Figure 5.9. Continued.

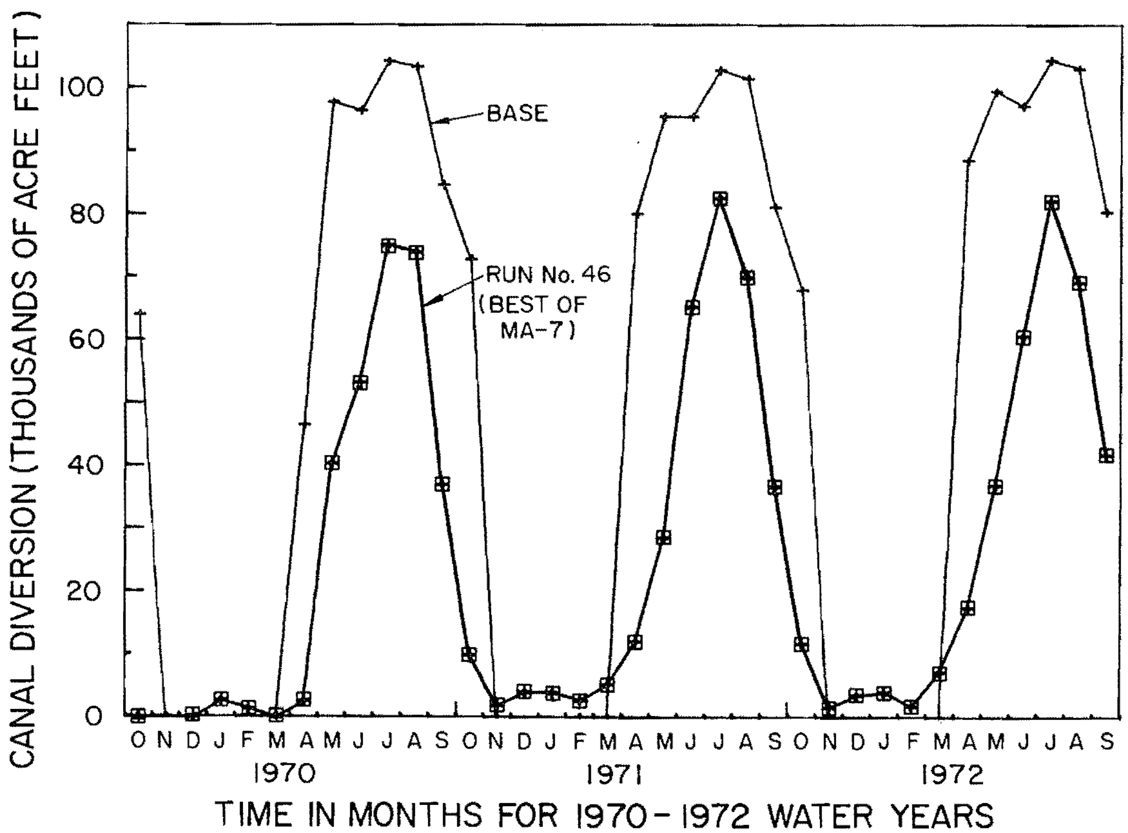


Figure 5.10. Comparison of canal diversions for the period 1970-1972 for the Grand Valley, Colorado, area between the base calibration and the best run (# 46) from Management Alternative 7.

Management Alternative 8 (MA-8)

The strategy for this alternative was similar to MA-7 except that canal lining was simulated by increasing the conveyance efficiency and the application efficiency was held constant at 66 percent. The results are summarized in Table 5.8 and Figures 5.11 and 5.12. By lining the canals and keeping the field application efficiency constant, the amount delivered was held constant throughout all five of the simulation runs. An overall reduction of 49 percent in the amount of canal diversions decreased salt loading by 531,000 tons or about 25 mg/l over the 3-year period.

Many more management strategies could be devised and run; however the preceding eight were deemed sufficient to test the model, demonstrate how the model can be used, and provide general information on probable trends. The results should not be taken as absolute because of the unresolved issue as to correct characterization of the actual system existing in Grand Valley.

Table 5.8. Parameter values and cumulative response summary for Management Alternative 8.

	Parameters					Predicted response - 1970-1972				
Run	PSP	ECV	PTW	ECUTW	EAP	Diversion as percent of base	Application efficiency (percent)	Water (thousands of acre-feet)	Salt (thousands of tons)	TDS (mg/l)
1	0.38	0.81	0.33	0	0.56	100.0	56	14,370	11,055	566.1
43	0.15	0.81	0.15	0.30	0.60	65.9	66	14,298	11,109	571.7
47	"	0.85	"	"	"	62.2	66	14,300	10,959	563.9
48	"	0.90	"	"	"	58.0	66	14,302	10,795	555.4
49	"	0.95	"	"	"	54.4	66	14,304	10,651	547.9
50	"	1.00	"	"	"	51.2	66	14,305	10,524	541.3

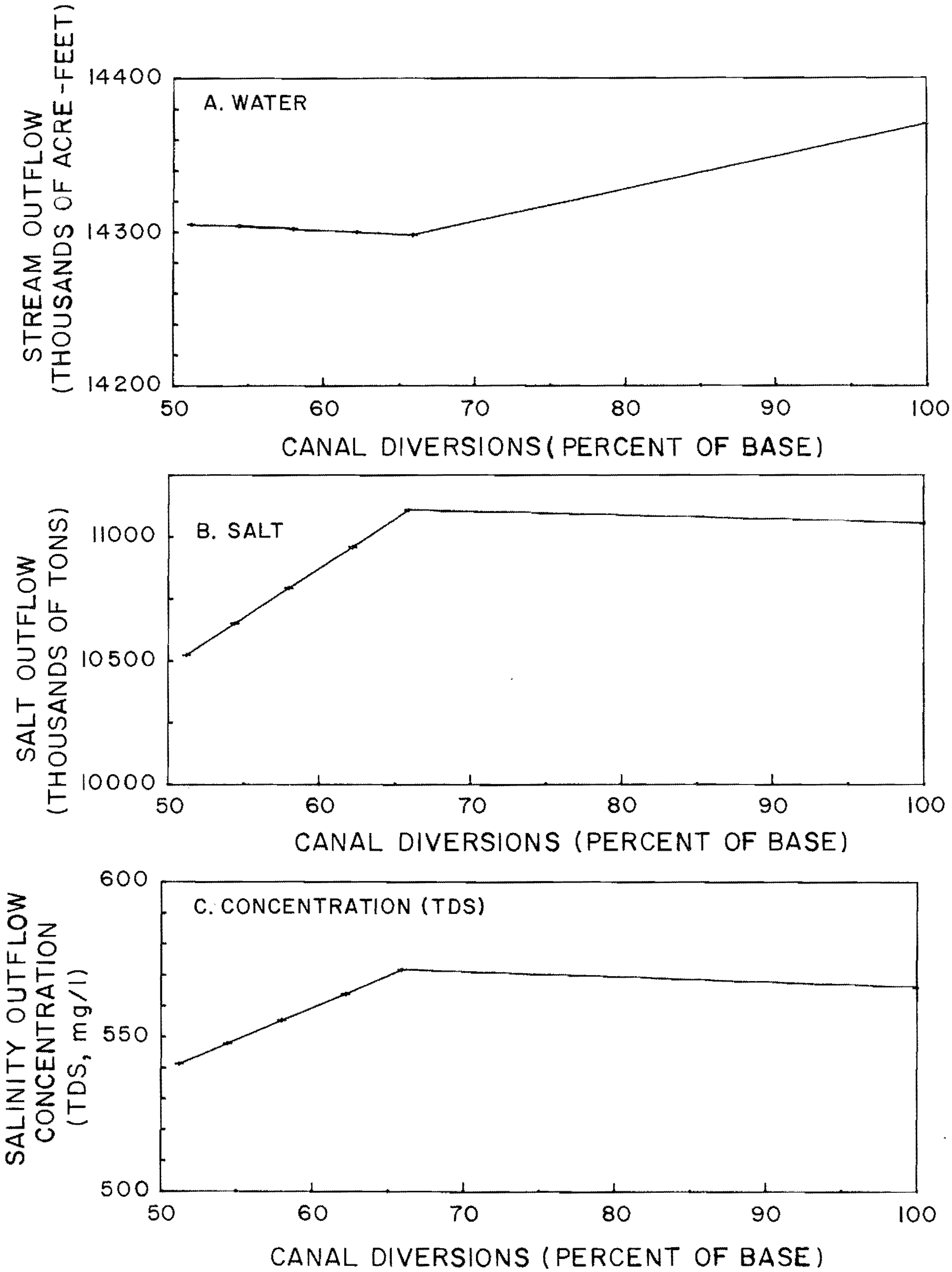


Figure 5.11. Predicted outflow response for the period 1970-1972 for Management Alternative 8 of a) water, b) salt, and c) salinity concentration (TDS) to changes in canal conveyance efficiency at a constant field application efficiency of 66 percent.

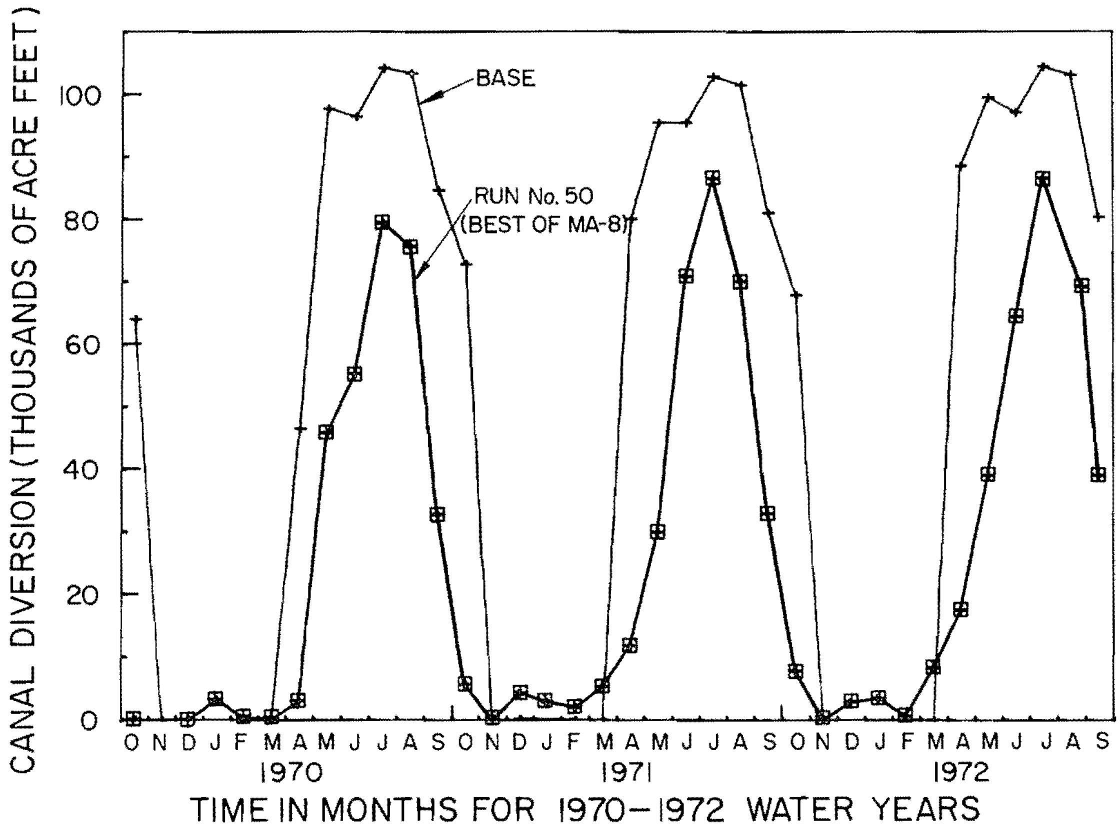


Figure 5.12. Comparison of canal diversions for the period 1970-1972 for the Grand Valley, Colorado, area between the base calibration and the best run (#50) from Management Alternative 8.

CHAPTER 6.
CONCLUSIONS AND RECOMMENDATIONS

A review of the state-of-the-art of hydrosalinity models identified one of the major gaps in modeling as inadequate understanding and representation of the quantity and quality interrelationships between surface water, drainage water, and groundwater. Most models predict relatively constant levels of salinity over time in surface drains during the irrigation season and an increase in concentration in similar drains at other locations during the nonirrigation season.

This study further revealed that current hydrosalinity models vary widely in their representation of 1) chemical processes, 2) interrelationships between surface water, irrigation drainage water, and groundwater, and 3) the salt pickup phenomena. The more complex hydrosalinity models assume equilibrium among constituent ions in the root zone soil water system and use the kinetic approach to simulate the nitrogen transformations in the root zone. Such models require extensive data that are available only for experimental plots under controlled conditions. Models at the other extreme utilize a simple conservation of mass approach to simulate salinity (TDS) movement, without accounting for salt precipitation and mineral dissolution in the deeper groundwater zone. The groundwater quality models generally assume an absence of chemical reactions between the soil water and the aquifer which might affect the dissolved solids concentrations.

In order to obtain reasonable results despite the absence of the data required by the more complex models, a new concept was introduced. This idea is that a site specific 'Threshold Concentration' (TC) of dissolved solids within a soil-water system can be identified, adequately estimated, and represented in a simple hydrosalinity model. Salt concentrations above the TC result in precipitation of salts within the soil profile. These higher values of TC, however, exist in the deeper layers of the unsaturated zone, depending upon the movement of salts through the soil layers. It was possible, based on these concepts, to represent salinity (TDS) in the return flows as a composite of individual component TDS outflows from the unsaturated zones and the saturated groundwater zone, thus retaining simplicity in the model by allowing the interrelationships among the surface water, drainage water, and groundwater to be represented in a lumped parametric manner. It is still necessary to have a detailed hydrologic model identifying all of the flow components, although the quality component is still the TDS.

These concepts were formulated into a hydrosalinity model designed to predict the effectiveness of alternative proposals to reduce salinity (TDS) discharges by irrigation water management. The model resulting from this effort parametrically represents the salinity processes that were identified as being important and still retains relative simplicity and can be calibrated and run from generally available data. The model termed BSAM-SALT was tested using field data from irrigated areas in Grand Valley, Colorado, and the Circleville subbasin of the Sevier River Basin in Utah. A set of management runs was made to demonstrate the utility of the model in predicting the salt loading caused by irrigated agriculture in the Grand Valley, Colorado, area.

The process of calibrating the model to the Grand Valley area revealed the importance of having an accurate identification of the irrigation conveyance and application system. The model proved to be very sensitive to the irrigation system definition. If the flow through the system really follows one pattern, salinity control is very sensitive to canal lining whereas, for another equally plausible system definition, salinity control was relatively insensitive to canal lining but highly sensitive to the field application efficiency. This points out the importance of accurate system definition to establishing irrigation water management practices that can be effective in salinity control. One can also see that the best management practice in one situation is not necessarily best in another.

Because of the problem of the uncertainty over present flow patterns within the Grand Valley system, the results from the management runs may not be directly applicable to that system. However, they do demonstrate the utility of the model as a tool in the evaluation process. In addition, the means by which a computer model can be used to simulate management alternatives was explained and explicit relationships existing between model parameters in the irrigated agriculture part of the BSAM-SALT model were derived to aid in making the management runs.

Recommendations for further research and study arising from this effort include:

1. Increased emphasis on identifying the salt pickup processes from natural sources as opposed to the agricultural sources so that component salt loading from these processes can be properly and accurately predicted.

2. Development of a technique to measure in situ soil salinity to facilitate estimating the threshold concentration of dissolved solids within the soil profile.

3. Collection and analysis of field measurements to verify the assumption that salt pickup is proportional to the amount of percolating water. Although results based on column experiments support this assumption, a comprehensive analysis of pertinent field data is necessary to really verify it.

4. Application of the model over a wide variety of irrigated agriculture sites to aid in refining the representation of the interrelationship among the surface water, drainage water, and groundwater and the salinity processes linked to them.

5. Identify the importance of sediment pickup and movement to the salinity problem, and if significant, develop a means for including it in the model.

6. Develop relationships between the management strategies that are possible to impose on an irrigation system and the corresponding parameter interrelationship that must be imposed on the model to accurately predict the effect of applying a specified strategy.

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