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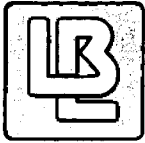
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UNIVERSITY OF CALIFORNIA

EARTH SCIENCES DIVISION

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Final Report: October 1, 1989 to December 30, 1990

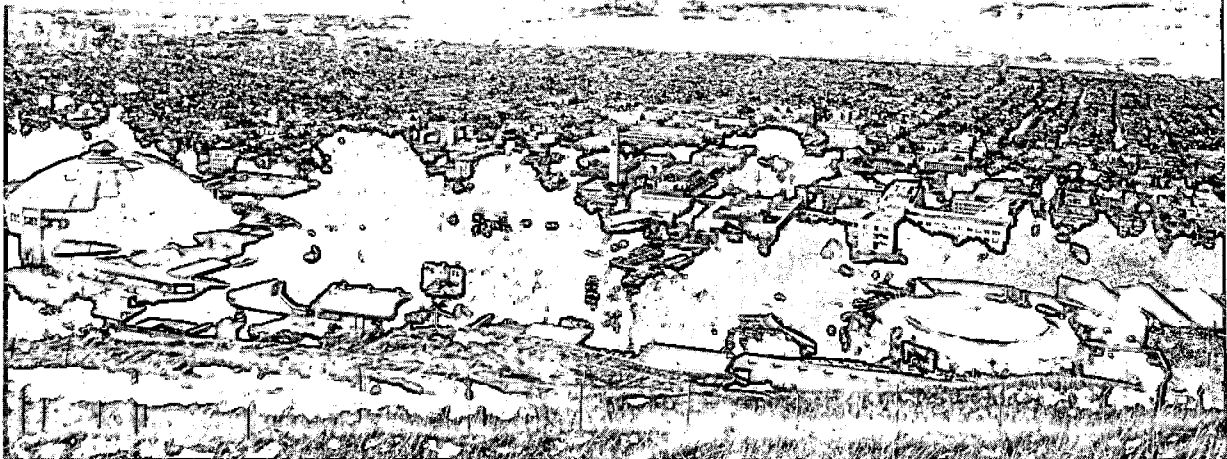
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LBL-30473

**Evaluation of Management Options for Disposal of Salt and Trace Element
Laden Agricultural Drainage Water from the Fallon Indian Reservation,
Fallon, Nevada**

Final Report

October 1, 1989 to December 30, 1990

Tetsu Tokunaga and Sally Benson
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March, 1991

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EXECUTIVE SUMMARY

This is the final report describing work performed on the Fallon Indian Reservation by the Earth Sciences Division at Lawrence Berkeley Laboratory during FY90. The Reservation is located about 75 miles west of Reno, Nevada, just south of the Carson Sink, the terminal drainage of the Carson River. These investigations were initiated at the request of the United States Bureau of Reclamation (USBR) in response to recent concerns regarding disposal of agricultural drainage water from the Reservation. The Reservation consists of 3290 ha (8120 acres) within an area of about 28,400 ha (70,000 acres) of irrigable lands in the Newlands Project. The Newlands Project provides irrigation water from the Truckee and Carson Rivers to the Lahontan Valley and adjacent areas.

The Reservation is transected by numerous irrigation and drainage canals. One of the major features of this system, the TJ Drain, discharges into Lead Lake, a primary wetland within the Stillwater Wildlife Management Area (SWMA). Recent investigations by the U.S. Fish and Wildlife Service (USFWS) have demonstrated that water in the TJ Drain is toxic to several aquatic indicator organisms, including bluegills, fathead minnows and daphnids (Finger *et al.*, 1989). Elevated concentrations of salts and a number of trace elements, including boron (B), arsenic (As), molybdenum (Mo) and lithium (Li) have been cited as potential sources of toxicity. Within the SWMA elevated levels of selenium (Se) and mercury (Hg) in waterfowl have also been cited as a potential concern (Hoffman *et al.*, 1990). This information, coupled with recent die-offs of fish and birds in the Carson Sink and the SWMA, has led to concern about continued discharge of TJ Drain water into local surface waters. In late 1990, plans for closing the TJ Drain and providing for alternative drainage were initiated.

Extensive investigations of the SWMA and surrounding lands are presently being conducted by the U.S. Geological Survey, the USBR and the USFWS. The objective of these detailed investigations is to provide a comprehensive assessment of the hydrogeological, geochemical and ecological consequences of irrigation and drainage of lands in the

Newlands Project. In comparison, the objective of the Lawrence Berkeley Laboratory effort was limited and focused in scope. We aim to provide information for assessing options for disposal of agricultural drainage water from the Reservation. In particular, our studies focused on irrigation and drainage of lands currently serviced by the TJ Drain.

Options for continued irrigation and drainage of the Reservation fall broadly into two categories:

- options that provide an alternative to drain water disposal into the SWMA; and
- options that include continuing the current practice of drain water disposal into the SWMA.

Other options include elements of both of these alternatives. For example, drainage water may be diverted away from the SWMA while concentrations of salts or trace elements exceed pre-set target levels. When levels drop permanently or temporarily below target levels, discharge into SWMA may resume, subject to a consensus of all interested parties. Additional discussion of specific options will follow a brief summary of the technical work supporting our assessment of drainage related issues at the Reservation.

TJ and Paiute Drain Water Quality

In Chapter 2 the following conclusions about water quality in the TJ and Paiute Drains are described.

- First, the water quality analyses of these drain waters performed independently by the USBR, USGS and LBL are generally in agreement with each other. While this agreement in itself is not particularly noteworthy, the credibility of the remaining conclusions is supported by this first point.
- Second, the principal constituents of the TJ Drain waters which impart poor quality to these waters are total dissolved solids, boron and arsenic. High concentrations of selenium have not been measured in the waters entering the SWMA from either the TJ or the Paiute Drains. Other trace elements, in particular molybdenum and

lithium, are also elevated compared to other drain waters, but their effects on wildlife remain to be determined.

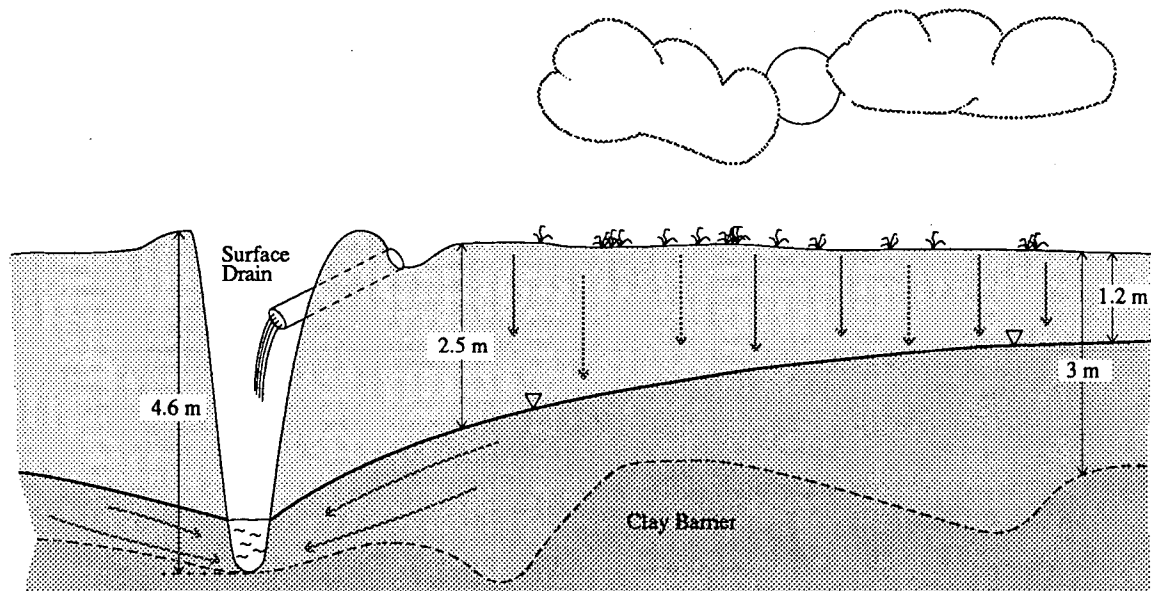
- Third, in reviewing the collective data from October 1985 through August 1990, no significant long-term changes in drain water quality have been detected. Seasonal variations in water quality are repeated annually, with no significant long-term improvement in drain water quality noted over the past 5 years.
- Finally, the TJ Drain water contributes a significant fraction of the salt and trace element load delivered to Lead Lake through the combined flow of the Paiute and TJ Drains. The contribution of the TJ Drain to the total loading of salts and trace elements in the combined flow as it enters Lead Lake is generally greater than 50%, and often is in excess of 80%. This large contribution, despite a generally smaller volumetric contribution arises from the substantially higher concentrations of solutes in the TJ Drain water than in the Paiute Drain water.

Salt and Trace Element Concentrations in Fallon Indian Reservation Soils

Chapter 3 describes the following information relating to the presence and movement of salts and trace elements from irrigated and non-irrigated soils into the TJ Drain.

A schematic showing the relationship between the TJ Drain, irrigated fields and the highly saline groundwater under the Reservation is shown in Fig. E.1. The Drain contains a mixture of tail water run-off from flood irrigation, deep percolation of irrigation water, deep groundwater and operational excess. Determining the relative contribution and water quality from each of these sources is beyond the scope of this investigation. However, our investigations support the following generalizations.

- In non-irrigated and in very recently irrigated farmlands, high concentrations of salts and soluble trace elements are present in the upper 2 m of the soil profile (\approx 6 ft). Water draining into the TJ Drain from these sources will contribute to the poor quality of drainage water.



ESD-9103-0098

Figure E.1. Schematic showing the current configuration of the TJ and A Drain Systems.

- In well-leached fields which have been subjected to longer periods of irrigation (5 to 20 years), soil profiles have become relatively depleted of their initial inventories of salts and soluble trace elements. Salt and trace element loading to a surface drainage system from these fields will be comparatively small.
- Salt concentrations in the upper 1.7 m of soil decrease roughly exponentially with time of irrigation, with a half-life of about 5 years. This provides a rough estimate of the time period over which this contribution to salt and trace element loading into a drainage system will diminish.
- Column leaching experiments demonstrate that in well-drained soil profiles pore waters can have substantially improved water quality following the equivalent of several years of irrigation and drainage.
- Local heterogeneities, in particular zones with higher clay contents exert at least two important influences on irrigation and drainage of the Reservation. First, infiltration rates through soils dominated by such layers are greatly diminished, which in turn greatly diminishes the rate of salt and trace element leaching. Second,

zones of high clay content tend to retain certain trace elements of concern within the soil profile through adsorption. Arsenic and boron have been shown to exhibit such associations in these soils. An important related point concerning local clay-rich soil profiles is the fact that low leaching rates through such low permeability profiles will moderate rates of trace element and salt release from this sites.

- Shallow groundwater provides the majority of salts and trace elements into the TJ Drain.
- The majority of the volume of drainage water in the TJ is derived from tail water run-off and operational excess. This observation is supported by an inverse relationship between water quality and flow rate in the TJ Drain.
- Minimizing the extent to which poor quality groundwater dominates drain water quality (under various drain configurations) even after surface soils have become well leached is crucial in achieving better quality drainage.

Groundwater Under the Fallon Indian Reservation

As part of the LBL effort a detailed investigation of the groundwater system underlying the Reservation was undertaken. Objectives of the investigation were two-fold. First, since shallow groundwater is the major source of salts and trace elements to the TJ Drain, this effort was designed to provide additional information on the extent and chemical characteristics of the saline groundwater discharging into the TJ Drain. Second, the shallow aquifers under the Reservation may be suitable for reinjection of TJ drain waters.

Chapter 4 describes the information obtained from the groundwater investigations. The following is a summary of the information obtained from 15 wells drilled, to a maximum depth of 109 m (358 ft.), by the United States Bureau of Reclamation to support this effort.

- The Fallon Indian Reservation overlies sediments deposited by an alternating sequence rivers and deep lakes in the Lahontan Valley. The first 3 to 5 m consist of

light yellow-brown loam, similar to the surface soils on the Reservation. Although clay, sand and silt fractions vary from site to site and the depth to which this unit extends is variable, it is distinctly different than the massive clay unit underlying it. Under the irrigated lands on the Reservation, this uppermost unit contains an estimated $1.4 \times 10^7 \text{ m}^3$ (11,000 acre-ft) of poor quality groundwater that may eventually discharge to a shallow drainage system.

- A massive clay unit extends from about 5 to 18 m below the ground surface. The light-gray to green-gray high plasticity clay is very similar from site to site. Lack of internal structure and wide-spread uniformity suggest that this was deposited in a deep lake environment. This unit is tentatively correlated with the Seho formation described by Morrison (1964). This unit limits flow of deep groundwater to the shallow drainage system.
- A fine to medium grained sandy unit extends from about 18 to 32 m below the ground surface. This unit is tentatively correlated with the Wymaha Formation of Morrison (1964). This is referred to as the S₁ unit in this report. In the city of Fallon it is an important aquifer for supplying drinking water to local residents. Under the Reservation, this unit may be suitable for reinjection of drainage water.
- The second massive clay unit extends from a depth of about 35 to 50 m. Again, it is remarkably uniform from site to site, suggesting deposition in a deep-lake environment. This unit acts as an aquiclude between the S₁ unit and the S₂ unit described below.
- From a depth of 50 m to about 95 m below the ground surface sediments are characterized by an irregular sequence of interbedded sands, clays and silts. This unit is tentatively correlated with the Paiute formation described by Morrison (1964). In this report this unit is referred to as the S₂ unit. It may be a suitable horizon for disposal of drainage water. The bottom of this unit is marked by a gradual transition to a third massive clay unit.

- Water levels in the S₁ unit are from 1.5 to 2 m below the ground surface. The S₂ unit is artesian, with fluid potentials from 0.1 to 2.5 m above the ground surface.
- The hydraulic conductivities measured in the S₁ and S₂ units are typical of fine sand and mixtures of sand, silt and clay. The average hydraulic conductivity measured for the S₁ unit is 1.5×10^{-5} m/s (4.2 ft/day). The average value measured in the S₂ unit is 8.6×10^{-6} m/s (2.4 ft/day).
- Groundwater under the Reservation is brackish. Salinities are in the range observed in the TJ drain. The salinity alone makes this water unfit for beneficial use as a drinking water supply, for livestock watering or irrigation.
- The ionic composition of groundwaters in the S₁ and S₂ units is dominated by sodium and chloride. Measured chloride concentrations in the S₁ unit range from 2,670 to 8,120 mg/L and the average value is 6,100 mg/L. Within the S₂ unit, Cl concentrations are somewhat lower and range from 1,310 to 2,750 mg/L. The average value is 2,390 mg/L. Cations are dominated by Na in both the S₁ and S₂ units. Concentrations in the S₁ unit range from 1,608 to 4,634 mg/L; the average value is 3583 mg/L. Sodium concentrations in the S₂ unit range from 818 to 1,901 mg/L; the average is 1,413 mg/L.
- Trace element concentrations in groundwater samples from the S₁ and S₂ units are also high, with the exception of selenium. Selenium concentrations in all groundwater samples collected from the S₁ and S₂ units are less than 2 µg/L. Arsenic concentrations range from 3 to 220 µg/L. These are in the range of values reported for the TJ Drain. These concentrations are also typical of groundwaters in the Carson Desert and exceed the drinking water standard of 50 µg/L. The State of Nevada's criterion for protection of aquatic life (40 µg/L) is also exceeded by groundwater from the majority of these wells. Boron concentrations are also high, ranging from 4 to 21 mg/L. Again, these are in the range of values reported for the TJ Drain and shallow groundwater in the Carson Desert .

- The combination of high salinity and elevated As and B concentrations in the groundwater under the Fallon Indian Reservation severely limit, if not preclude, conventional beneficial use of this resource. Although this investigation is not exhaustive, the preponderance of data suggest that good quality water will not be found in the S₁ and S₂ aquifers.

Salt and Trace Element Concentrations in Surface Waters and Soils in the Vicinity of the Fallon Indian Reservation

In arid, closed-basin environments such as that of the Lahontan Valley and Carson Sink, high salinity and trace element concentrations are not characteristics limited to agricultural drain waters. While the present study is focused on alternatives for management of TJ Drain waters, it is important to develop an appreciation for the general hydrological and geochemical environment in order to select management options appropriate for this particular setting. Therefore, a limited sampling and analysis of soils and surface waters outside the Reservation boundaries was undertaken. Results from this effort are described in Chapter 2 (surface waters) and Chapter 5 (soils) and summarized as follows.

- Very high concentrations of soluble salts and some trace elements are characteristic of the Carson Sink playa soils and SWMA sediments. Water-extracted As concentrations ranged from 235 to 1,650 µg/kg in Stillwater sediments, and from 5,000 to 22,000 µg/kg in Carson Sink soils. Water-extracted B concentrations ranged from 5 to 200 mg/kg in SWMA sediments, and from 120 to 500 mg/kg in Carson Sink soils. Although only a small number of samples were analyzed, these high concentrations are to be expected because these samples were collected from areas near and within the bottom of a very large closed basin.
- Surface water quality goals for protection of aquatic life can be exceeded even when as little as 1% of the soluble inventory of B, As, and Se in Carson Sink surface sediments is dissolved in ponded waters. Laboratory experiments involving ponding of distilled water over some SWMA sediments demonstrated that

high salinities (up to 70,000 mg/L), high B concentrations (5 to 67 mg/L), and high As (90 to 620 µg/L) concentrations in ponded waters can occur simply through recycling from the sediments.

- It is important to recognize that the high salt and trace element inventories in Stillwater sediments and Carson Sink soils are primarily a consequence of the regional geochemical and hydrological processes that have operated since the recession of Lake Lahontan. Although the relatively very recent introduction of agricultural drainage waters from farms throughout the Lahontan Valley into the wetlands has significantly compromised surface water quality, the agricultural contribution is small compared to the naturally occurring inventories of salts and trace elements in these regional soils and sediments. Thus, even with improvements in inflow water quality, careful water management within the wetlands will be required to minimize recycling of salts and trace elements from sediments back into surface waters. While such management is impractical within the Carson Sink (which has no discharge area), it should be recognized that the enormous inventory of salts and trace elements in the sink are largely a product of natural drainage and evaporation over thousands of years rather than a result of recent agricultural activities.

Management Options for Fallon Indian Reservation Drainage Waters

Specific options for drainage water management from the Fallon Indian Reservation fall into two broad categories, options that provide an alternative to drainage water disposal into the SWMA and options that include continued disposal into the SWMA. Other options include elements of both of these alternatives. For example, drain water may be diverted away from the SWMA while concentrations of salts or trace elements exceed pre-set targets. When levels drop permanently or temporarily below these targets, discharge into SWMA may resume, subject to a consensus of all interested parties. Specific options falling into these two categories are listed in Table E.1. An assessment of these options is provided in the paragraphs that follow.

Table E.1. Options for drain water management considered in the LBL study.

| | |
|--|--|
| <p>Management options with continued discharge into the Stillwater Wildlife Management Area.</p> | <ol style="list-style-type: none"> 1. Improve drain water quality and reduce drainage volume by decreasing the depth of the TJ Drain. 2. Improve drainage water quality and reduce drainage volume by replacing the TJ Drain with a shallow, more closely spaced tile drain system. 3. Continue the current disposal practice based on evidence that the incremental loading to the SWMA from the TJ Drain is negligible. |
| <p>Management options providing alternatives to drain water disposal in the SWMA.</p> | <ol style="list-style-type: none"> 1. Dispose of drain water in evaporation basins in the Carson Sink. 2. Accelerate evaporation of drain water by spraying or surface spreading. 3. Accelerate evaporation of drain water with heat from the Stillwater Geothermal Resource. 4. Reinject drain water into shallow aquifers. |

Decrease the Depth of the TJ and A Drain Systems

Decreasing the depth of the TJ and A Drains is expected to improve drain water quality and reduce drainage volumes. These improvements are achieved by reducing the contribution of groundwater inflow into the drainage system. However, these improvements will be accompanied by decreases in the efficiency of the drainage system. To maintain effective control over the water table and achieve effective drainage, the TJ and A Drain system must be augmented by more closely spaced drains. The tile drain system, illustrated in Fig. E.2 and described below, coupled with decreasing the depth of the TJ and A Drain

systems will improve drain water quality and assist in drain water management at the Fallon Indian Reservation.

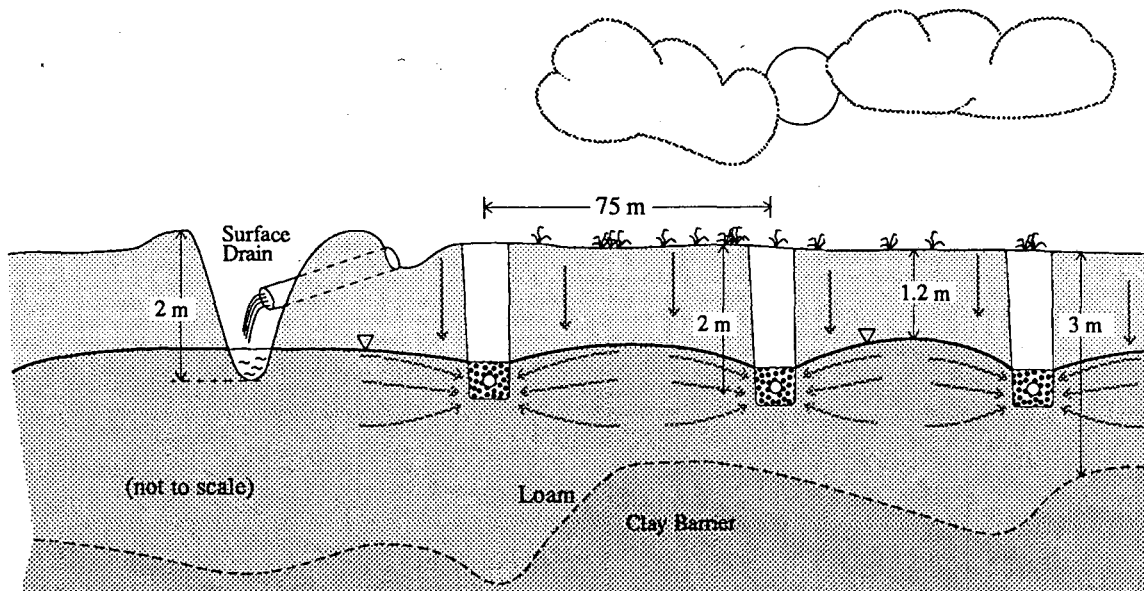
Replace the TJ and A Drains with a Shallow Tile Drain System

Water table control and leaching requirements on the Reservation may be achieved by replacing the TJ and A Drains with a shallow tile drain system as illustrated in Fig. E.2. The tile drain system consists of buried perforated pipes, located about 2 m below the ground surface. These perforated pipes collect subsurface drain water, lower the water table and provide a method for separating subsurface drainage from surface drainage.

In addition to meeting the basic needs for sustaining the agricultural productivity of these lands, this type of drainage system is expected to facilitate drain water management in two ways. First, as illustrated in Fig. E.2, surface run-off and operational excess can be collected in a shallow surface drain system for reuse as irrigation water or to create wetlands on the Reservation. This makes efficient use of this good quality water and reduces the total volume of poor quality drainage requiring disposal. Second, the shallow tile drains may be installed near the top of the water table, thus, limiting inflow of deeper poor quality groundwater. Eventually, drain water collected from a tile drain system should have water quality typical of a well-leached field. The time frame over which such improvements may be realized remains to be tested, because, as described above, the salt and trace element contents of FIR soils is highly heterogeneous. Provisions for pilot-testing shallow tile drains at the Reservation are now underway.

Continue the Current Disposal Practice

Although the overall impact of TJ Drain water on trace element and salt loading in the SWMA remains to be determined, clearly, the TJ was a significant contributor to the salt and trace element loading in South Lead Lake during 1989–90. As a result of this, and legislation directing closure of the TJ, this method of drain water disposal will be terminated shortly.



ESD-9103-0099

Figure E.2. Schematic showing the configuration of a tile drain system and shallow surface drains.

Dispose of Drain Water in Evaporation Basins in the Carson Sink

Disposal of TJ Drain water in the Carson Sink will create large shallow pools that may attract wildlife. The high concentrations of As, B and common salts in TJ Drain water, combined with dissolution of additional salts from the Carson Sink soils may create habitat that is harmful to wildlife. The potential risks of creating such harmful habitat must be carefully weighed before this option is pursued.

Accelerate Evaporation of Drain Water by Spraying, Surface Spreading or Heat from the Geothermal Resource

Drainage water evaporation rates may be accelerated during the hot summer months by a spray system, surface spreading or heat from the nearby geothermal resource. However, at the beginning and end of the irrigation season, cool ambient temperatures will prevent complete evaporation of the drain water. The remaining drain water will form pools that attract wildlife, creating the risk of adverse environmental impacts. Provisions for disposing of the salt and trace element residue from the drain water may also be required. Significant operating and maintenance costs are expected with this option.

Reinject Drain Water into Shallow Aquifers

Two aquifer units (S_1 and S_2) under the Fallon Indian Reservation may be suitable for reinjection of drainage waters. Feasibility of this plan hinges on the following factors:

- Total volume of drainage requiring disposal;
- Injection capacity of each well;
- Compatibility of drainage waters and groundwater in the S_1 or S_2 unit;
- Regulatory requirements regarding protection of beneficial uses of groundwater at the Reservation and in surrounding areas; and
- Costs associated with construction, maintenance and operation of a reinjection facility.

Each of these factors requires careful analysis before this option can be seriously considered. Nevertheless, sufficient information is available for a preliminary feasibility assessment. Each of the first 4 factors listed above is discussed in turn. Cost estimates for constructing and operating a reinjection facility are premature.

Total drainage volume requiring disposal through reinjection can be estimated from past data regarding flows in the TJ Drain. The USBR (1987) estimated annual flows of about $8 \times 10^6 \text{ m}^3$ (6500 acre-ft) in the TJ Drain. Mass balance calculations and visual observations indicate that the majority of flow in the TJ Drain is derived from sources other than the shallow groundwater system, such as tail water run-off and operational excess. If the subsurface drainage component can be separated from these other sources, reinjection of the estimated $9 \times 10^5 \text{ m}^3/\text{year}$ (724 acre-ft/year; USBR, 1987) of surface drainage may be feasible. The tile drainage system described above would provide a means for separating subsurface drainage from tail water run-off. Additional analysis of likely discharge rates from shallow buried drains is required to refine these estimates and evaluate the feasibility of reinjecting discharge from shallow tile drains.

Reinjection rates for wells screened in the S_1 or S_2 unit are estimated to average $2 \times 10^{-3} \text{ m}^3/\text{s}$ (50 acre-ft/year per well). This estimate is contingent on the following factors: wells are completed with a casing diameter of at least 0.20 m (8 inches); the gravel

pack is installed such that no mechanical plugging in the near-well region occurs; wells are screened over the entire thickness of either the S₁ or S₂ unit; and that mixing between drainage water and native groundwaters does not precipitate minerals or create biological plugging of the well-screen. At present, none of the factors present a major obstacle. Injection testing will be required to substantiate the current estimates of the injection capacity of wells screened in the S₁ and S₂ units.

Regulatory requirements for reinjection of agricultural drainage water into the S₁ or S₂ aquifer unit must be developed through discussions with local, State and Federal agencies. Regulations regarding injection of waste waters, in general, are site-specific. Specific concerns that need to be addressed for reinjection of drainage water include the following: degradation of local and regional groundwater resources; impairment of beneficial uses of local and regional groundwater; and potential influences of reinjection on nearby farmlands or wetlands, such as rising water tables or degradation of shallow groundwater or surface water quality. Because of poor water quality in the S₁ and S₂ units under the Reservation, reinjection of drain water is unlikely to impair beneficial uses of the groundwater under the Reservation. However, since the S₁ unit is used for a number of purposes in the City of Fallon and surrounding areas, assessment of potential impacts of reinjection on these users will be required. The basic information is now available to address these concerns. If reinjection is pursued as an option for drainage water disposal, discussions with appropriate regulatory agencies and concerned parties should be initiated soon.

Acknowledgements

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1. INTRODUCTION

This is the final report describing work performed on the Fallon Indian Reservation (FIR) by the Earth Sciences Division at Lawrence Berkeley Laboratory during FY90. As shown in Fig. 1.1, the FIR is located in west central Nevada, south of the Carson Sink and west of the NE-SW trending Stillwater Range. These investigations were initiated at the request of the United States Bureau of Reclamation (USBR) in response to recent concerns regarding disposal of agricultural drainage water from the FIR. The FIR falls within an area of about 28,400 ha (70,000 acres) of irrigable lands in the Newlands Project. The Newlands Project, some features of which were completed by the USBR in 1915, provides irrigation water from the Truckee and Carson Rivers to the Lahontan Valley and adjacent areas. Eligible lands within the Newlands Project, including the FIR, may receive annually a maximum of 1.07 ha-m/ha year (3.5 acre-ft/acre) for bottom land and 1.37 ha-m/ha year (4.5 acre-ft/acre) for bench land.

As shown in Fig. 1.2, the FIR consists of the adjacent units Fallon I and Fallon II, that together cover an area of 3290 ha (8120 acres). Approximately one-half of Fallon I is developed for agriculture. Until 1989, Fallon II remained undeveloped. Now, 34 ha (85 acres) is under development in Fallon II. The FIR is transected by numerous irrigation and drainage canals, major features of which are shown in Fig. 1.2. The TJ Drain, which was constructed in 1982, transports subsurface and surface drainage waters from irrigated and non-irrigated lands of the FIR. After merging with the Paiute Diversion Drain, it ultimately discharges into Lead Lake, a primary wetland within the Stillwater Wildlife Management Area (SWMA). Recent investigations by the U.S. Fish and Wildlife Service (USFWS) have demonstrated that water in the TJ Drain is toxic to several aquatic indicator organisms, including bluegills, fathead minnows and daphnids (Finger *et al.*, 1989). Elevated concentrations of a number of trace elements, including boron (B), arsenic (As), molybdenum (Mo) and lithium (Li) have been cited as potential sources of toxicity. Within the SWMA elevated levels of selenium (Se) and mercury (Hg) in waterfowl have also been cited as a

potential concern (Hoffman *et al.*, 1990). This information, coupled with recent die-offs of fish and birds in the Carson Sink and the SWMA, has led to concern about continued discharge of TJ Drain water into local surface waters. In late 1990, plans for closing the TJ Drain were made.

Extensive investigations of the SWMA and surrounding lands are presently being conducted by the U.S. Geological Survey (USGS), the USBR and the USFWS. The objective of these detailed investigations is to provide a comprehensive assessment of the hydrogeological, geochemical and ecological consequences of irrigation and drainage of lands in the Newlands Project. A reconnaissance investigation has recently been completed by the Department of the Interior outlining potential environmental concerns regarding the effects of elevated concentrations of trace elements on the ecological health of the SWMA (Hoffman *et al.*, 1990). The USBR (Denver) has also recently installed a network of water quality monitoring wells and soil sampling sites on the FIR. Analyses from this effort are currently underway and a final report will be available in the future. These data are expected to provide an areally extensive assessment of shallow groundwater quality and distribution of trace elements within FIR soils. In addition to current efforts, extensive investigations of the soils and groundwater in this area have been conducted by various state and federal agencies over the past 80 years. Although the majority of studies did not focus on the presence of trace elements in soils or groundwater, they do provide a wealth of data on groundwater quality, soil classification and the hydrologic setting. All of these sources of information have been drawn upon in our analysis of drainage options at FIR.

In light of the extensive investigations currently underway at Stillwater, the objective of the Lawrence Berkeley Laboratory effort was limited and focused in scope. We aimed to provide information for assessing options for disposal of agricultural drainage water from the FIR. In particular, our studies focused on irrigation and drainage of lands currently serviced by the TJ Drain.

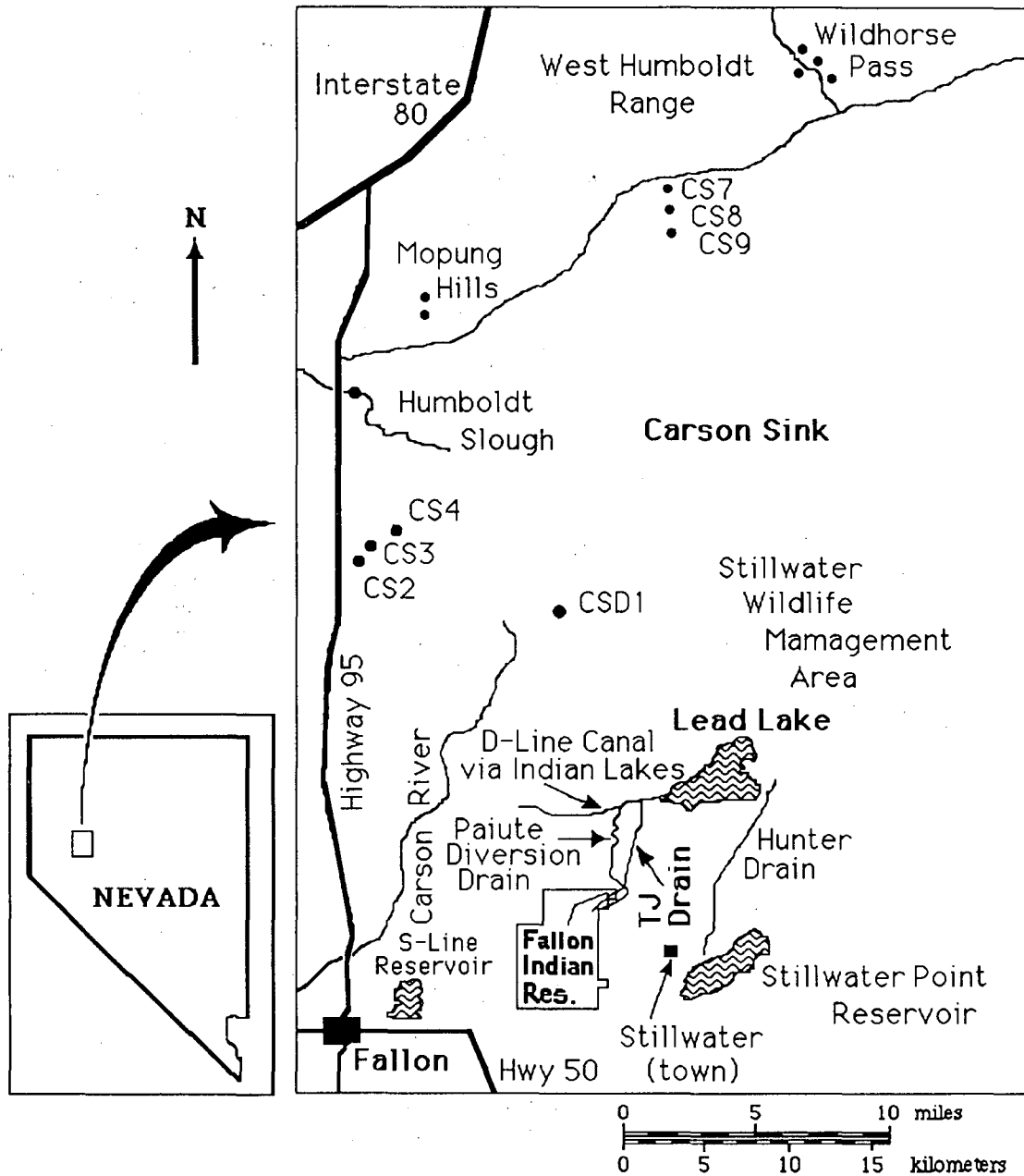


Figure 1.1. General location map of the Carson Sink Area, including the Fallon Indian Reservation.

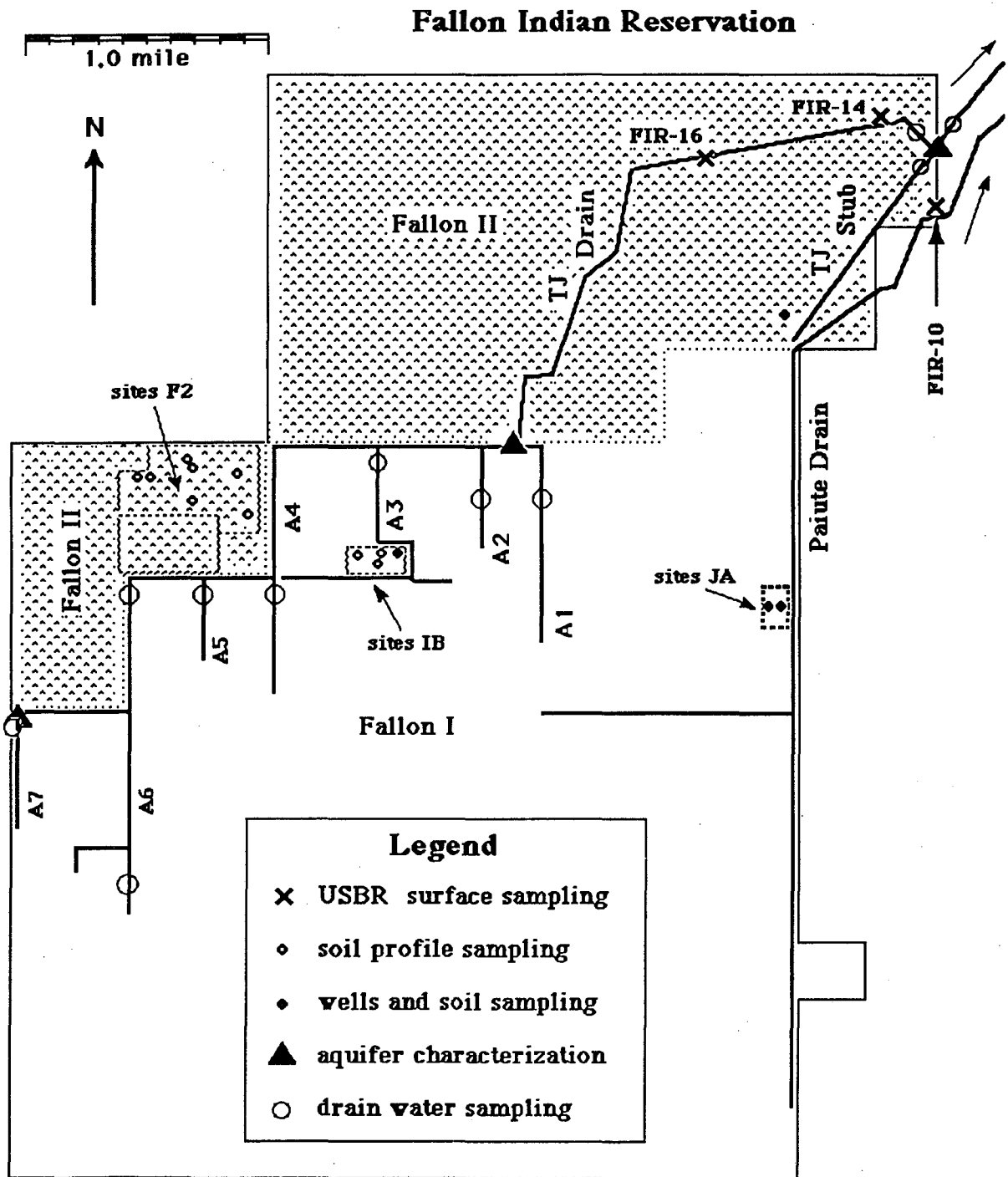


Figure 1.2. Location map showing sampling sites within the Fallon Indian Reservation used for the LBL study.

Options for continued irrigation and drainage of those lands that we considered fall broadly fall into two categories:

- options that provide an alternative to disposal into the SWMA; and
- options that include continuing the current practice of disposal into the SWMA.

Other options include elements of both of these alternatives. For example, drainage water could be diverted away from the SWMA while concentrations of salts or trace elements exceed pre-set target levels. When levels drop permanently or temporarily below the target levels, discharge into SWMA may resume, subject to a consensus of all interested parties.

Specific options that fall into each of these categories are outlined below. These were developed through conversations with USBR and USGS personnel who are familiar with the Stillwater area. Alternatives to the current practice of disposal into the Paiute Diversion Drain and Lead Lake include but are not limited to:

- surface discharge with or without accelerated evaporation; possible disposal areas include FIR lands or the Carson Sink;
- inject TJ Drain water into a confined aquifer with poor water quality under the FIR or Carson Sink; several local horizons including the geothermal aquifer may be suitable targets; and
- accelerate evaporation of the drain water with heat from the local geothermal resource.

Alternatives for continued discharge following the current practice of drain-water disposal into the Paiute Diversion Drain and Lead Lake are listed below. Options include:

- improving the water quality and/or reducing the drainage volume by decreasing the depth of the TJ Drain;

- improving the water quality and/or reducing the drainage volume by replacing part of the TJ Drain with buried pipe;
- replacing the current deep open-channel drainage system with a shallower, more closely spaced tile drainage system; and
- continuing the current discharge practice, contingent upon demonstration that the TJ Drain contributes an insignificant incremental contribution to salt and trace element loading of Lead Lake.

All of these options have been reviewed. In the executive summary and the following chapters, recommendations for the most feasible options are provided.

The LBL effort was divided into the five tasks summarized in Table 1.1. This report is organized to roughly parallel these tasks. Chapter 2 summarizes the surface water investigations in the TJ Drain and a limited set of off-site locations. Chapter 3 discusses soils in the Fallon Indian Reservation, Chapter 4 describes the groundwater investigations and Chapter 5 summarizes off-site soil investigations in the Stillwater Wildlife Management Area and Carson Sink.

Table 1.1. Summary of the FY90 tasks for TJ Drain investigations.

| | |
|--------------|---|
| Task TJ-I. | Literature Review and Data Synthesis |
| Task TJ-II. | Field Investigations of the Interactions of TJ Drain with Soil Water and Groundwater A. Detailed soil water, groundwater and drain-water chemistry monitoring B. Detailed hydraulic monitoring of soil water, groundwater and drain water |
| Task TJ-III. | Laboratory Investigations of the Mobility of Salts and Trace Elements A. Salt and trace element leachate from FIR soil columns B. Salts and trace element loading from wetting and drying cycles in SWMA and Carson Sink sediments |
| Task TJ-IV. | Groundwater Investigations A. Identification of aquifer system structure in the vicinity of TJ Drain B. Depth profiles of water quality at potential injection sites |
| Task TJ-V. | Analytical Support and QA/QC A. Analytical support B. Quality Assurance and Quality Control |

2. SURFACE WATERS

In this chapter, a brief review of some of the available earlier water quality data pertinent to the TJ and Paiute Drains is presented in Section 2.1.1. This review is followed with presentation of some of the water quality information from these drains obtained during 1989–90 as part of the present study (Section 2.1.2).

Other surface water samples in areas surrounding the FIR have also been collected as part of the present study. While the present study is focused on alternatives for management of TJ Drain waters, it is important to develop an appreciation for the general hydrological and geochemical environment in order to select management options appropriate for this particular setting. Data of this nature from some previous investigations was not accessible at the beginning of the present study, hence a limited sampling of surrounding surface water features was conducted. Some of these data are reviewed in Section 2.2.

Throughout this report various water quality criteria, goals, effect levels and beneficial use criteria are provided for reference purposes. A discussion of water quality goals for the Lahontan Valley is beyond the scope of this report. Furthermore, our use of these criteria does not necessarily imply that we endorse these water quality goals or criteria in this particular environment.

2.1. TJ and Paiute Drain Waters

In this section, data supporting the following important conclusions about water quality in the TJ and Paiute Drains are described.

- First, the water quality analyses of these drain waters performed independently by the USBR, USGS and LBL are generally in agreement with each other. While this agreement in itself is not particularly noteworthy, the credibility of the remaining conclusions is supported by this first point.

- Second, the constituents of the TJ Drain waters which impart poor quality to these waters are total dissolved solids (TDS), boron and arsenic. High concentrations of selenium have not been measured in the waters entering the SWMA from either the TJ or the Paiute Drains. (Other trace elements, in particular molybdenum and lithium, are also elevated compared to other drain waters, but their effects on wildlife remain to be determined.)
- Third, in reviewing the collective data from October 1985 through August 1990, no significant long-term changes in drain water quality have been detected. Seasonal variations in water quality are repeated annually, with no significant long-term improvement in drain water quality noted over the past 5 years.
- Finally, the TJ Drain water contributes a significant fraction of the salt and trace element load delivered to the SWMA through the the combined flow of the Paiute and TJ Drains. The contribution of the TJ to the total loading of salts and trace elements in the combined flow as it enters Lead Lake is generally greater than 50%, and often is in excess of 80%. This large contribution, despite a generally smaller volumetric contribution arises from the substantially higher concentrations of solutes in the TJ Drain waters than in the Paiute Drain water.

2.1.1. Review of Previous TJ and Paiute Drain Water Quality Data

The chemical composition of TJ Drain waters has been characterized by USBR (1987), Finger *et al.* (1989), and in Hoffman *et al.* (1990). The USBR (1987) report provides monthly drain water quality data for the period of October 1985 through September 1986. To date, it represents the public document with the most comprehensive data set on TJ Drain water quality. Samples from the TJ Drain were collected from within the boundaries of the FIR. Analyses were included for As, Ba, Be, Cd, Cu, Fe, Hg, Li, Mn, Mo, Ni, Pb, Se, Zn, Ca, Mg, Na, K, CO₃, HCO₃, Cl, SO₄, NO₃, F, SiO₂, B, TDS, EC and pH. Analyses for the pesticides ethyl parathion, Dursban, Sencor, 2,4-D, Furdan, Banvel, Tordan and Naled all tested below detection limits. The study suggested that the principal

trace elements of concern are As, Se and possibly Mo. More recently, concerns were raised with respect to drain water B concentrations and TDS (e.g. Finger *et al.*, 1989; and Hoffman *et al.*, 1990). Therefore drain water quality data from the most northern (downstream) USBR monitoring sites (FIR 14 and FIR-16, see Fig. 1.2 for the location of these two monitoring sites) will be summarized with respect to these elements of concern.

All waters of the TJ Drain which originate within the FIR, excluding the TJ Stub section, pass through the USBR FIR-14 and FIR-16 monitoring sites before entering the SWMA. Since a very small fraction of the total drain flow is associated with the TJ Stub, it appears reasonable to consider the FIR-14,16 water quality as representative of flow-volume averaged properties of all FIR inputs. Only the FIR-16 data are used after April 1986 since runoff from an adjacent pond mixed with FIR-14 waters, resulting in significant dilutions of drain waters (USBR, 1987). These data will be compared with those from Hoffman *et al.* (1990), collected at sites further downstream within the SWMA from June 1986 to July 1987. In addition, both the USBR (1987) and Hoffman *et al.* (1990) studies include data from the Paiute Drain. From the USBR report, the Paiute Drain data collected at site FIR-10, at the northern end of the FIR will be referenced. In the work of Hoffman *et al.* (1990), Paiute Drain water samples were collected at both upstream and downstream locations with respect to the point where the TJ Drain discharges into the Paiute Drain. These data will also be compared. In a later section, similar data collected recently (July 1989 to August 1990) by LBL will be presented.

Drain water salinity, as indicated by data on total dissolved solids (TDS) or electrical conductivity (EC), has been documented in both the USBR (1987) and Hoffman *et al.* (1990) reports. In the USBR (1987) report, an arithmetic average TDS of 16,150 mg/L was reported for the FIR-16 monitoring site during the April 1986 to September 1986 period (USBR, 1987; Fig. 5 and also from Table B2, page B16). At the FIR-14 site, the average TDS from October 1985 to February 1986 was 20,900 mg/L (USBR, 1987; Table B2, page B15). It should be noted that the period during which the FIR-14 data were collected corresponds to the months during which irrigation waters were not applied. The

sample with the lowest TDS (10,100 mg/L), was collected in August 1986. The maximum TDS from site FIR-14, (26,000 mg/L) was collected in February 1986. The minimum and maximum TDS from the TJ Drain waters reported in Hoffman *et al.* (1990, Table 12, page 93) are 2,740 and 29,800 mg/L respectively. The low and high TDS drain water samples were collected on July 23, 1987 and March 12, 1987 respectively.

Both of the studies referenced above have noted that drain water quality and flow rates follow a general seasonal pattern which responds primarily to local irrigation practices. Irrigation water is supplied to the region by the beginning of April and continues through the summer months. A large fraction of the applied irrigation water is estimated to enter the drain ditches via surface runoff of tail waters. Thus, higher drain flow rates and lower drain water TDS are associated with the irrigation season. Drain water quality generally improves as the irrigation season progresses. In contrast, the fall and winter months are generally associated with lower drain flow rates. The higher concentrations of dissolved salts and trace elements that are measured during these months are generally associated with the slow drainage from the saline shallow aquifer. The relative contribution of shallow groundwater from regions below the crop root zone (deeper than about 2 m, or 6 ft) into the TJ Drain is believed to be relatively large as a result of the great depth to which the drain ditches were excavated. Thus an appreciation of the influence of the shallow groundwater on drain water quality is needed in order to understand both the physical (flow rates) and chemical (salinity and trace element concentrations) characteristics of the TJ Drain water.

The early portion of each irrigation season generally also provides drain waters of high salinity and high trace element concentrations since this period represents the beginning of the annual phase of active flushing. It is emphasized that the temporal pattern in drain water quality described above represents the *average* response of the system to the annual irrigation schedule. Large variations in TJ Drain flow rates and water quality can be observed over short time interval, reflecting responses to local irrigation and drainage practices.

The Paiute Drain was reported to have had a flow-weighted TDS of 803 mg/L at the northern boundary of the FIR during the period from October 1985 to September 6 (USBR, 1987; Fig. 5 and Table B2, page B13). The minimum TDS of 400 mg/L was obtained in the August 1986 sample, while the maximum TDS sample, with 3,630 mg/L was collected in February 1986. In the study by Hoffman *et al.* (1990), the Paiute Drain was sampled both upstream and downstream of the TJ Drain discharge point. The upstream site had minimum and maximum TDS values of 282 and 1660 mg/L respectively, while the corresponding values for the downstream site were 907 and 5030 mg/L (Hoffman *et al.*, 1990; Table 12, p. 93). The low and high TDS values at both of these sites were obtained on August 30, 1987 and May 12, 1986 respectively. Comparisons of the upstream and downstream TDS values cited above indicate that discharges from the TJ Drain have contributed the majority of the salt loading in the Paiute Drain. Estimates of relative loadings of salts and several trace elements will be discussed in Section 2.1.2.

Boron concentrations in the TJ Drain (FIR-14,16) ranged from 9.3 to 25 mg/L in the USBR (1987) study. Hoffman *et al.* (1990) report boron (B) concentrations ranging from 3.5 to 28.0 mg/L for the TJ Drain in the SWMA. The two reports provide Paiute Drain B concentrations ranging from 0.35 to 4.10 mg/L (USBR, 1987), and 0.19 to 2.00 mg/L (Hoffman *et al.*, 1990). Again, maximum concentrations are generally associated with pre-irrigation drain waters. The B effect level for vertebrate embryos is 0.20 mg/L (Hoffman *et al.*, 1990, p. 34). The Nevada State criterion for B in irrigation water is 1.00 mg/L.

In the USBR (1987) study, As concentrations at the FIR-14,16 sites ranged between 120 and 300 µg/L (ppb). It should be noted that some sites further upstream within the A-Drain system exhibited higher As concentrations over the period of October 1985 through May 1986. In the USBR (1987) report, the highest As concentration, 560 µg/L, was obtained on March 16, 1986 at a site between the A5 and A6 branch drains. In general, winter, pre-irrigation season drain water quality is typically the poorest within drains of the region. In some cases drain water levels are too low to sustain flow and are therefore subject to evaporative concentration. Drain water quality generally improves as the irrigation sea-

son progresses due primarily to dilution with irrigation tail waters. The concentrations of As in drain water reported in Hoffman *et al.* (1990) are generally consistent with those of USBR (1987), with some possible downstream dilution. Arsenic concentrations in the TJ Drain ranged from 39 to 190 $\mu\text{g/L}$ in the work of Hoffman *et al.*, (1990), with the maximum concentration measured on March 12, 1987 (pre-irrigation). For comparison, the Paiute Drain provided As concentrations in the range of 13 to 32 $\mu\text{g/L}$ in the USBR (1987) study (site FIR-10, near the TJ Stub), while Hoffman *et al.* (1990) report concentrations ranging from 23 to 43 $\mu\text{g/L}$ at a site further downstream within the SWMA (yet still upstream of the confluence of the TJ and Paiute Drains). The State of Nevada As criterion for aquatic life is 40 $\mu\text{g/L}$.

Selenium concentrations in TJ Drain water ranged from 2 to 7 $\mu\text{g/L}$ at site FIR-14,16 (USBR, 1987). It should be noted that one TJ Drain site included in the USBR study yielded higher Se concentrations. This was the TJ Stub (16 to 26 $\mu\text{g/L}$), which is not believed to contribute significantly to the TJ Drain. Selenium concentrations at the downstream TJ Drain site were all $\leq 1 \mu\text{g/L}$ (Hoffman *et al.*, 1990). All Paiute Drain water Se concentrations are reported as $< 1 \mu\text{g/L}$ in both USBR (1987) and Hoffman *et al.* (1990). In view of the generally low Se concentrations in these and other water sources for the SWMA, the origin of elevated Se concentrations in wildlife of the area remains to be determined.

Molybdenum concentrations in TJ Drain water ranging from 0.55 to 1.90 mg/L (site FIR-14,16, USBR, 1987) and from 0.086 to 0.86 mg/L (Hoffman *et al.*, 1990) have been reported. The same sources report Paiute Drain Mo concentration ranges of < 0.01 to 0.16 mg/L and 0.010 to 0.039 respectively. Presently there are no water quality standards for Mo in surface waters.

2.1.2. Recent (1989–90) Drain Water Quality Information

Since July 1989, TJ and Paiute Drain waters at several locations within the FIR and SWMA have been collected at nominally 3 month intervals. It is recognized that wide

fluctuations in drain water quality and flow rates do occur on much shorter time scales. The drain water samples were collected primarily to obtain an independent and up-to-date set of water quality data for comparison with results from the previous studies. It is particularly important to determine whether or not TJ Drain water quality has changed significantly during the years of its operation. Drain water sampling sites located within the FIR are depicted in Fig. 1.2. Only the water quality data from the TJ and Paiute drains will be addressed in this section. Water quality data collected in the Hunter Drain and other surface waters in the SWMA and surrounding areas will be presented and discussed in Section 2.2.

The water quality data to be described from these drain samples include salinity (as indicated by EC data), B, As, Se and Mo concentrations. The data presented are comparable to the drain water data of USBR (1987) and Hoffman *et al.* (1990) reviewed in the previous Section (2.1.1). These data are collected from the TJ and Paiute Drains where they mix before entering South Lead Lake. In addition, data on salinities and trace element concentrations in various sections of the A Drain system at various times of the year are also presented. Finally, a compilation of drain water quality data obtained at the northern boundary of the FIR during the past year is presented in tabular form.

For each drain water sampling event, a minimum of two separate water samples were collected and analyzed. Surface water samples are collected in a 60 ml syringe, pre-rinsed twice with the water to be sampled and filtered (0.45 μm) into replicate 60 ml polyethylene bottles. One bottle of each replicate is acidified. In a number of samples, unfiltered samples have also been collected for further digestion at the laboratory (alkaline peroxide, Weres *et al.*, 1989) prior to filtration. Details of the sampling procedure and the quality assurance and quality control program are provided in the LBL QA/QC document (October, 1990).

Electrical conductivities of TJ and Paiute Diversion (PD) drain waters are shown in Fig. 2.1a. (Figures for this section are presented at the end of the section.) The 1986–87 data presented in this and similar bar graphs on the TJ and PD drains were obtained from the study by Hoffman *et al.* (1990, Tables 11 and 14). The PD and TJ Drain samples ob-

tained by LBL during 1989–90 were collected at a point 50 m upstream of their confluence. The “PD+TJ” samples were collected 100 m downstream of the confluence. From this figure, it is clear that the TJ Drain water salinities are consistently and substantially higher than those of the Paiute Drain. The ranges of salinities in the 1989–90 waters, as indicated through the EC data, are generally similar to those reported in the earlier works. The increased EC values in the mixture of the two drain waters (PD+TJ) relative to the incoming PD waters is an approximate indicator of the relative contribution of TJ Drain water volume and TDS. It is apparent from the comparisons between the PD and the PD+TJ data in Fig. 2.1a that the relative contribution of the TJ Drain to the salt loading of the combined flow is important. In most cases where comparisons are available, the EC of the combined PD+TJ waters is measurably greater than the EC of the PD water prior to mixing.

An estimate of the relative contribution of drain water volume and salt loading provided by the TJ Drain in the combined PD+TJ waters can be obtained from the combination of the TJ, PD and TJ+PD water EC data. These estimates rely on the relatively linear relation between the EC and TDS of waters and the fact that (in the absence of losses) the TDS of a mixture is equal to the volume-weighted average of individual source TDS values. It is recognized that the use of EC or TDS data for such estimates ignores potential sources of error associated with solute losses (for example, by precipitation or decomposition), and in the case of reliance on EC data, nonuniqueness and nonlinearity in the EC-TDS relation. In general, it would be preferable to use TDS data instead of EC data. However, for the system under consideration, such sources of error are minor, when the goal is limited to the estimation of relative volumetric contributions and TDS loadings. The principle source of error arises from assuming a linear EC-TDS relation. Over the EC ranges of interest, such errors introduce as much as a 16% relative uncertainty in the final calculation.

With the aforementioned qualifications in mind, an estimate of the fraction of the combined TJ+PD drain water volume that originates from the TJ during a particular sampling is obtained by

$$f(\text{TJ}) \approx \frac{\text{EC}(\text{PD} + \text{TJ}) - \text{EC}(\text{PD})}{\text{EC}(\text{TJ}) - \text{EC}(\text{PD})} \quad \text{Eq. [2.1]}$$

where $f(\text{TJ})$ is an estimate of the fraction of the total flow that originates from the TJ Drain. The fractional contribution of the TJ Drain to the TDS concentration can then be estimated by

$$F^*(\text{TJ}, \text{TDS}) \approx \frac{\text{EC}(\text{TJ})}{\text{EC}(\text{PD} + \text{TJ})} \times f(\text{TJ}) \quad \text{Eq. [2.2]}$$

where $F^*(\text{TJ}, \text{TDS})$ is an estimate of the fraction of salt in the total flow that comes from the TJ Drain.

Estimates of the relative volumetric contribution and relative salt load contribution from the TJ Drain into South Lead Lake are presented in Fig. 2.1b. It is apparent that despite an often relatively low drain water volume contribution, the relative contribution of salts from the TJ Drain is substantial. On a number of occasions, the TJ Drain contributes over 80% of the salt load in the combined PD+TJ flow. The data in Fig. 2.1a also demonstrate that relative drain water volume and salt loading patterns appear not to have changed in any significant manner during the past 4 years (Fig. 2.1b). Inclusion of the USBR (1987) information extends this conclusion back to the past 5 years.

EC data for samples collected in the A, TJ and Paiute Drains at various times within the FIR are shown in Fig. 2.2. While more samples were collected, the data in this and similar figures to follow come from the few extensive samplings of the drain systems within the FIR. The letters a, b and c alongside the EC data at particular locations indicate the time of year in which the samples were obtained. The a, b and c sampling times are from mid-summer (07-20-89), winter (Feb., Mar. 1990), and the beginning of the 1990 irrigation season (04-07-90). As expected, the local drain water EC trends roughly follow the seasonal pattern of the collective drain water. Lowest salinities occur during the summer when irrigation tail waters dilute the drain water. High local salinities generally coincide with the late winter and very early stages of the irrigation season.

In studying this EC map and other similar maps for B, As, Mo and Se presented later in this section, it should be noted that wider ranges in salinity and trace element concentrations will generally be observed in the collective data from very localized sampling in comparison with the ranges of values obtained in the main drain lines. This is to be expected for two interrelated reasons. First, the main drain systems provide flow-weighted average concentrations of the various solutes being transported from the upstream tributaries. Thus, extreme values are averaged out in the mixing of individual sources. On the other hand, extremes in local branch drain water quality are expected since these relatively isolated local waters more strongly reflect local differences in soil salinity, shallow groundwater composition, local tail water spills and (in the case of stagnant waters) evaporative concentration.

Boron concentrations in the TJ, Paiute and PD+TJ drains from various samplings are shown in Fig. 2.3a. Note that a semi-logarithmic scale has been used on the concentration axis so that the widely differing B concentrations in the TJ and PD Drains could be clearly illustrated. Included on this graph is a horizontal line set at 0.2 mg/L (0.2 ppm), denoting the effect level for B on vertebrate embryos (Hoffman *et al.*, 1990). Both the TJ and PD Drain waters regularly exceed this concentration. It should also be noted that this effect level B concentration is periodically exceeded in some “background” waters such as the Carson River below the Lahontan Reservoir (Hoffman *et al.*, 1990). Various reservoir and canal waters sampled in the present study also indicated that B concentrations in the local surface water supply commonly exceed the 0.20 mg/L effect level (Section 2.2.2). From Fig. 2.3a, it is clear that the TJ Drain contributes high concentrations of B into the SWMA relative to the Paiute Drain.

The relative contribution of B in the PD+TJ flow into the SWMA can be estimated in a manner analogous to that previously described with respect to estimating the relative TDS contribution. The proportion of the B load in the combined PD+TJ flow delivered by the TJ Drain can be estimated by multiplying the volume fraction $f(\text{TJ})$ obtained in Equation [2.1] by the ratio of B concentrations in the TJ versus PD+TJ waters, i.e.,

$$F^*(TJ, B) \approx \frac{B(TJ)}{B(PD + TJ)} \times f(TJ) \quad \text{Eq. [2.3]}$$

where $F^*(TJ, B)$ is the fraction of the B load originating from the TJ Drain. These estimates of the fractional contribution of B from the TJ have been performed and are summarized in Fig. 2.3b. Estimates of volumetric and TDS fractional contributions are also included in this figure for comparison. As shown in Fig. 2.3b, the TJ drain often has contributed between 80% and nearly 100% of the B load in the combined flow in the TJ and Paiute Drains. It is also apparent that the fractional contribution of B is generally very similar to that for TDS. This is consistent with the high correlation between B and TDS observed by Hoffman *et al.* (1990, page 35).

Boron concentrations in the drains within the FIR are presented in Fig. 2.4. As with the previously discussed distribution of EC data, B data exhibit the expected seasonal patterns. Also note that, as with the EC map data, wider ranges in B concentrations are observed in these local drain water samples.

Arsenic concentrations at the TJ, PD and PD+TJ Drains are presented in Fig. 2.5a. Included in this figure is a horizontal line indicating the State of Nevada freshwater aquatic life beneficial use criterion of 40 $\mu\text{g/L}$. While the 1986–87 As data from Hoffman *et al.* (1990) and the 1989–90 LBL data are generally similar, the latter data set has, on average, higher As concentrations. The significance of these differences remains to be determined. As noted previously, large fluctuations in water quality from all drains have been observed, even on relatively short time scales so that the small sample populations being compared in Fig. 2.5a are inadequate for determining the significance of the differences. Nevertheless, the data all indicate that the beneficial use criterion with respect to As is regularly exceeded in the TJ Drain waters and is at least close to being exceeded in the PD waters. Consequently, the mixture of the two waters in the PD+TJ samples will also commonly have As concentrations near or in excess of the State of Nevada water freshwater quality criterion.

The relative As loading from the TJ Drain has been estimated in a manner analogous to that previously described for relative B loading. Results are plotted in Fig. 2.5b. In

Fig. 2.5b, relative contributions of the TJ Drain to the volumetric flow and the TDS loading have again been included for comparison. Similar to the patterns in TDS and B loading by the TJ Drain, the relative As loading from the TJ Drain is in excess of its volumetric contribution. This can actually be deduced from the fact that As concentrations in the TJ Drain are systematically higher than As concentrations in the Pauite Drain.

Concentrations of drain water As in various portions of drainage systems within the FIR are mapped in Fig. 2.6. The temporal trends and wide ranges of concentrations are similar to the previously noted patterns with respect to EC and B.

None of the main drain water samples collected at the exits from the FIR have contained Se concentrations exceeding either the Nevada 24-hour average criterion of 35 $\mu\text{g/L}$, or the lower, more commonly used surface water goal of 5 $\mu\text{g/L}$. The observation that Se concentrations are less than 5 $\mu\text{g/L}$ in the drain waters exiting the FIR is consistent with the results from previous studies of the USBR (1987) and Hoffman *et al.* (1990). Because of the very low Se concentrations in the TJ and PD Drains, estimates of fractional Se loading were not performed. Selenium concentrations in drain waters within the FIR are mapped in Fig. 2.7. Note that in this map several locations can be found where Se concentrations significantly in excess of 5 $\mu\text{g/L}$ have been observed. It is equally important to note that the flow-averaged Se concentrations in waters exiting the FIR are less than 5 $\mu\text{g/L}$ (upper-right corner of Fig. 2.7), consistent with the previous discussion. The higher local Se concentrations may arise from several processes. In stagnant branches of the system, evaporative concentration of solutes including Se will occur. In branches carrying soil water leached from a newly developed field, an initial pulse of seleniferous drain waters may result. Finally, in branch drains where water levels are rising such that salt crusts along the drain banks become redissolved, Se enrichment may occur. This latter process is a potentially more general mechanism for trace element recycling and will be discussed further in Section 2.2 and in Chapter 5.

Molybdenum concentrations in drains of the FIR are shown in Fig. 2.8. Presently no surface water quality standards or goals have been established for Mo. All Mo concentra-

tions in the drains are less than 1.4 mg/L. The TJ Drain Mo concentrations are systematically higher than the Mo concentrations measured in the Paiute Drain. The seasonal pattern in drain water quality is less apparent in the case of Mo.

In summarizing the drain water data collected for this study, two general trends can be discerned for EC (TDS), B, As, Se and Mo. First, TJ Drain waters are generally more concentrated with respect to these trace elements and TDS than waters of the Paiute Drain. Second, seasonal fluctuations in drain water quality are generally observed. In particular, drain water concentrations of most constituents tend to reach maxima during the winter and at the beginning of the irrigation season. These observations were first presented in the USBR (1987) report and subsequently in Hoffman *et al.* (1990). It is important to note that these periods of highest trace element and TDS loading often coincide with periods of minimum drain flow rates. Thus fluctuations in concentrations alone are not reliable indicators of changes in transported inventories of solutes. Accurate measurements of flow rates through the various drain systems are needed to determine actual loading rates. It is also recognized that water quality and flows fluctuate on much shorter time scales, not only from day to day, but within hours also. Concerning Se, neither TJ nor Paiute drain waters exhibit elevated concentrations at the exit points of the FIR or at the entry point into the SWMA. However, some locally high drain water Se concentrations were observed (27 to 46 µg/L), generally in low flow or stagnant portions of the TJ Drain. A tabulation of TJ Drain water quality data at its exit point in the FIR is presented in Table 2.1 to provide an overview of the points discussed in this section.

Table 2.1. Summary of electrical conductivity (EC), boron, arsenic and selenium data from TJ Drain waters sampled at the exit point at the northern boundary of the FIR.

| Date | Time | E.C. dS/m | B mg/L | As µg/L | Se µg/L |
|----------|-----------|--------------|-----------|------------|------------|
| 7/20/89 | ~12:00 pm | 10.0 | 10.0 | 200 | 0.8 |
| 7/21/89 | 8:30 pm | 19.8 | 12.0 | 201 | 1.4 |
| 7/21/89 | 8:40 pm | 0.7 | 0.6 | 20 | 0.2 |
| 7/21/89 | 8:50 pm | 0.8 | 0.6 | 29 | 0.4 |
| 7/23/89 | ~1:00 pm | 1.5 | 1.1 | 35 | 0.3 |
| 10/31/89 | 10:00 am | 16.6 | 13.3 | 240 | 1.0 |
| 3/21/90 | 8:40 am | 26.8 | 19.0 | 361 | 1.8 |
| 4/7/90 | 6:50 pm | 30.3 | 23.2 | 470 | 0.3 |
| 4/7/90 | 6:52 pm | 30.9 | 23.3 | 299 | 0.8 |
| 5/10/90 | 1:09 pm | 10.3 | 7.7 | 156 | 1.6 |
| 5/10/90 | 1:23 pm | 10.8 | 8.3 | 250 | 4.2 |
| 5/12/90 | 10:40 am | 3.0 | 2.7 | 54 | 1.1 |
| 6/13/90 | 9:05 pm | 20.4 | 15.0 | 128 | 1.6 |
| 8/23/90 | 5:01 pm | 2.5 | 2.3 | 73 | 0.6 |
| 8/23/90 | 5:31 pm | 2.5 | 2.1 | 75 | 0.4 |

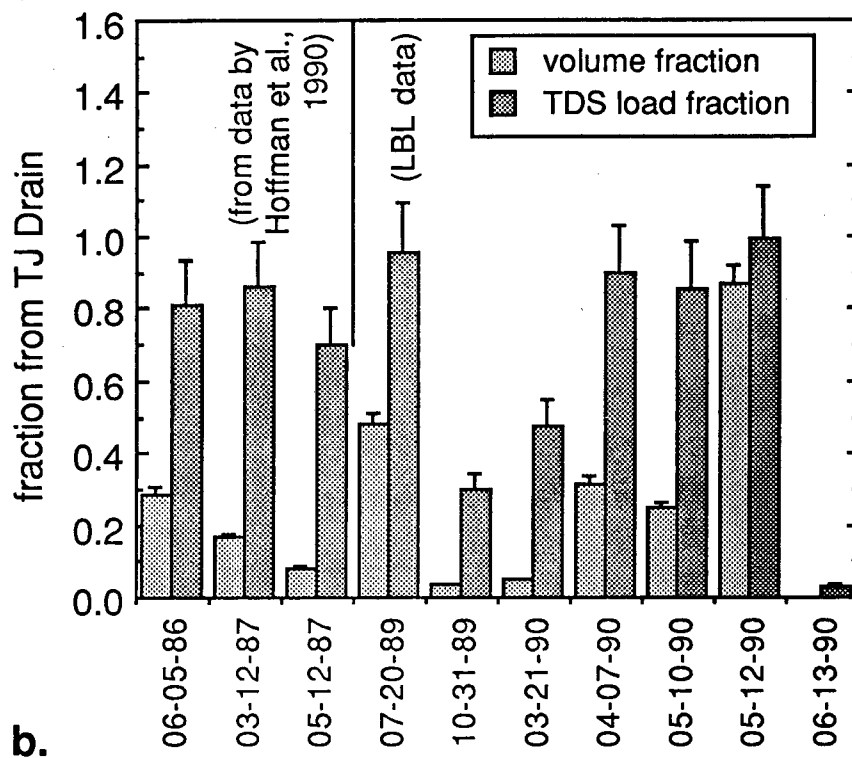
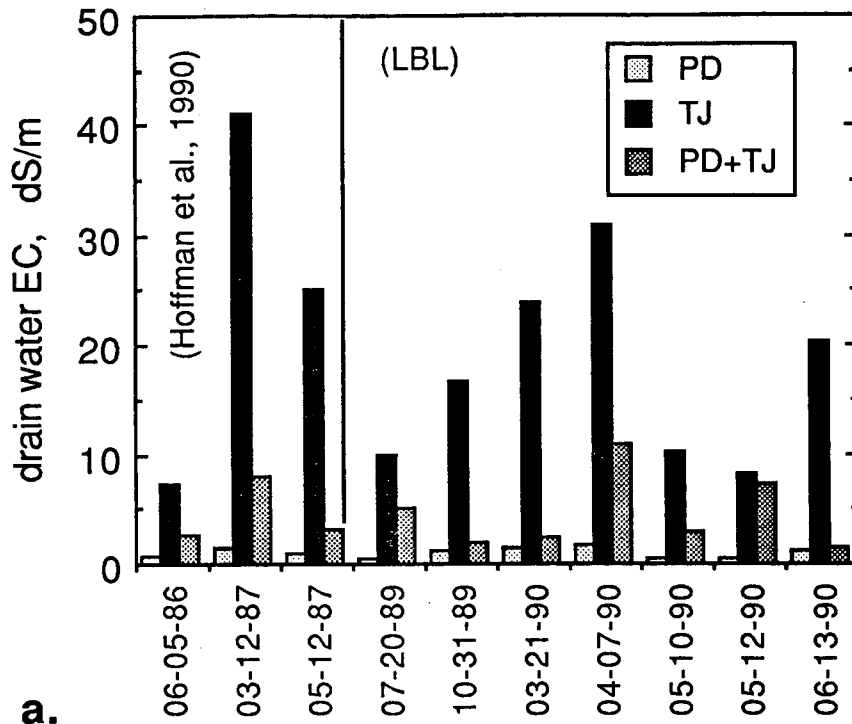


Figure 2.1. (a) Electrical conductivity (EC) of TJ, Paiute Diversion (PD) and mixed (PD+TJ) waters at the point of discharge of the TJ Drain into the Paiute Diversion Drain in the SWMA. (b) Estimates of volumetric and TDS loading contributions of the TJ Drain into the combined PD+TJ Drain flows into the SWMA.

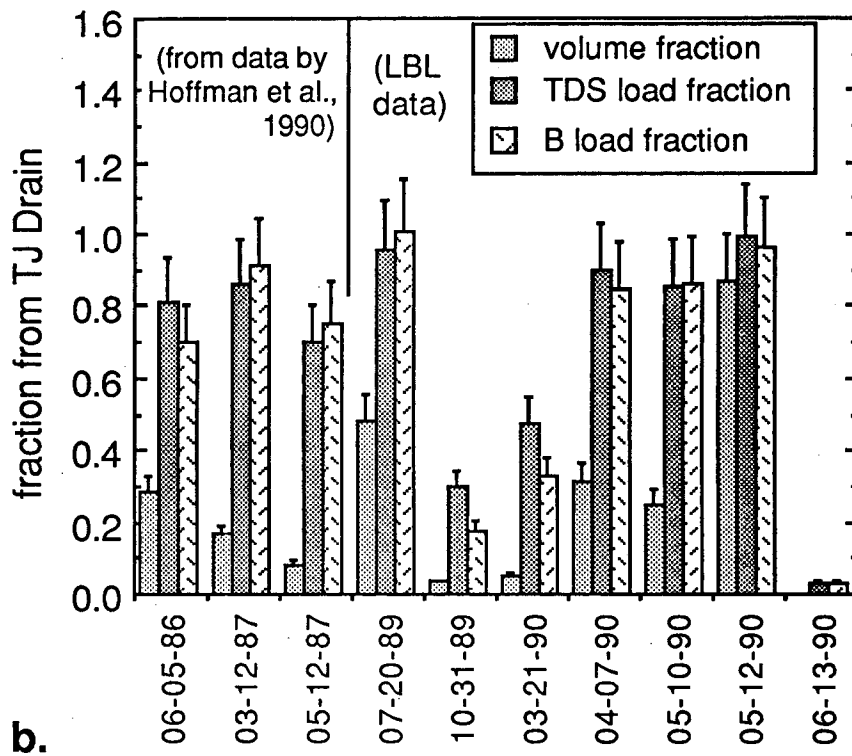
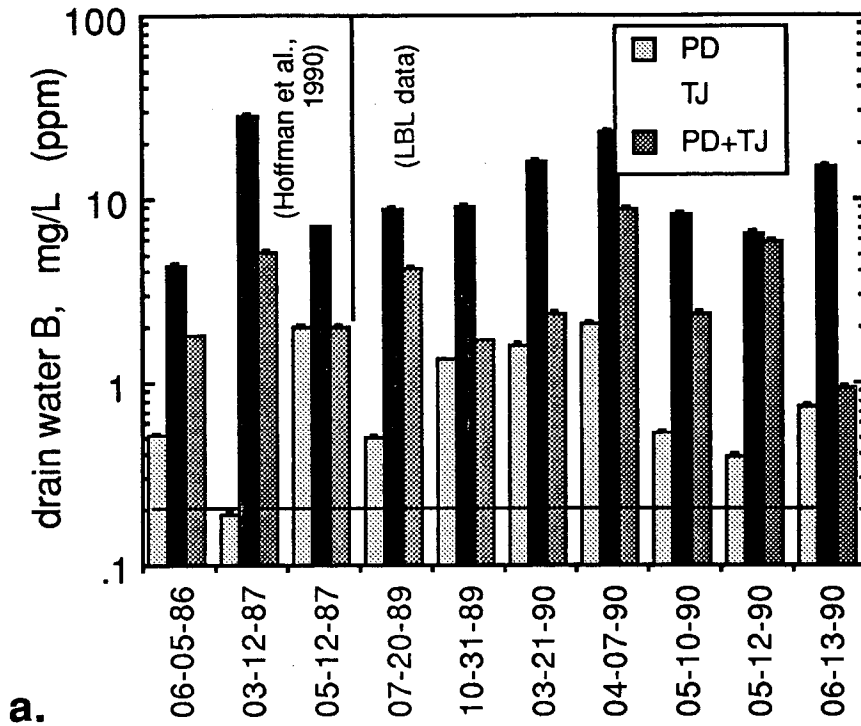


Figure 2.3. (a) Boron concentrations in the TJ, Paiute Diversion (PD) and mixed (PD+TJ) waters at the point of discharge of the TJ Drain into the Paiute Diversion Drain in the SWMA. (b) Estimates of volumetric, TDS and B loading contributions of the TJ Drain into the combined PD+TJ Drain flows into the SWMA.

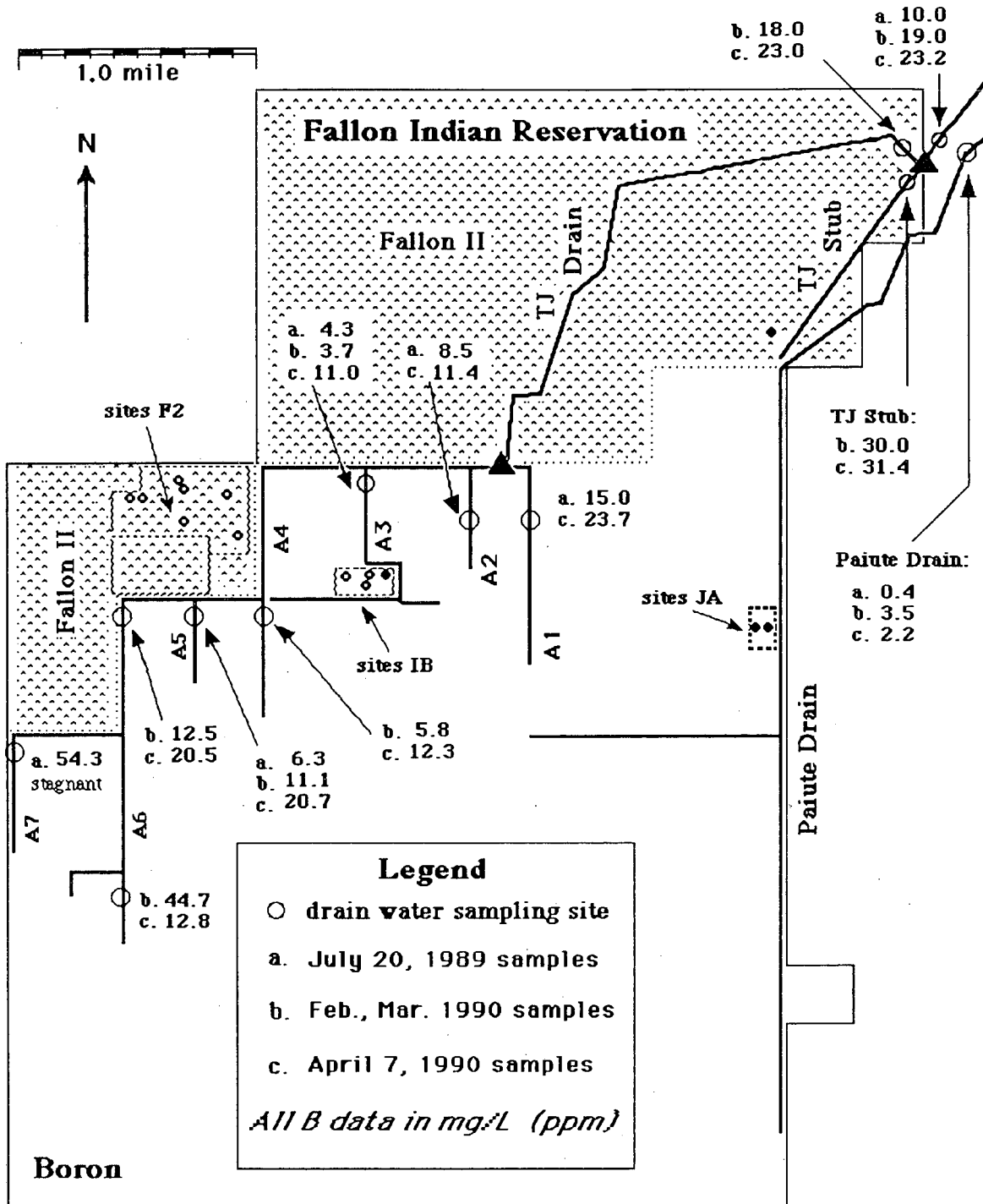


Figure 2.4. Map of FIR drain water boron (B) concentrations at several sampling times during 1989-90.

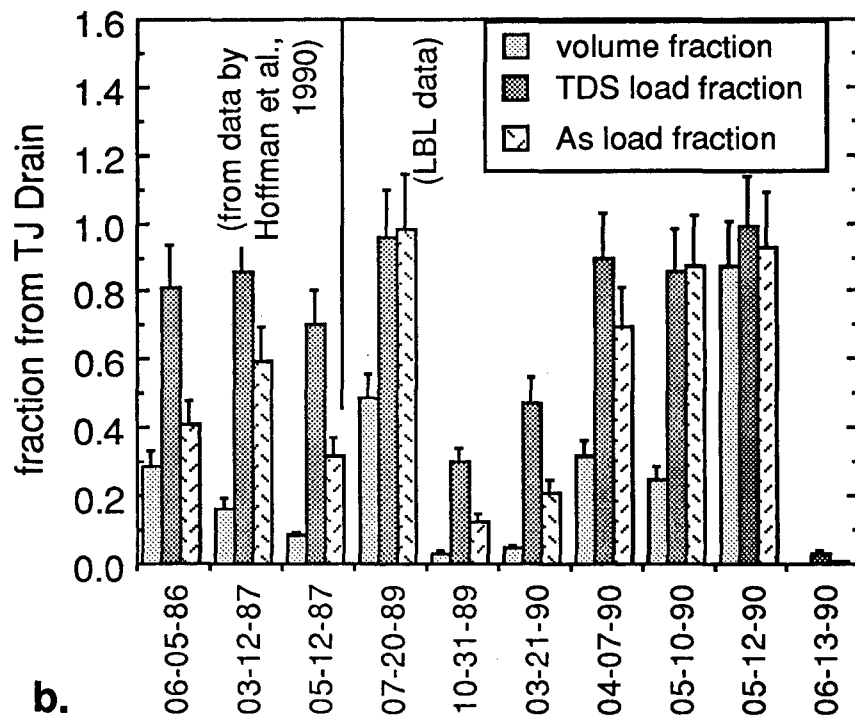
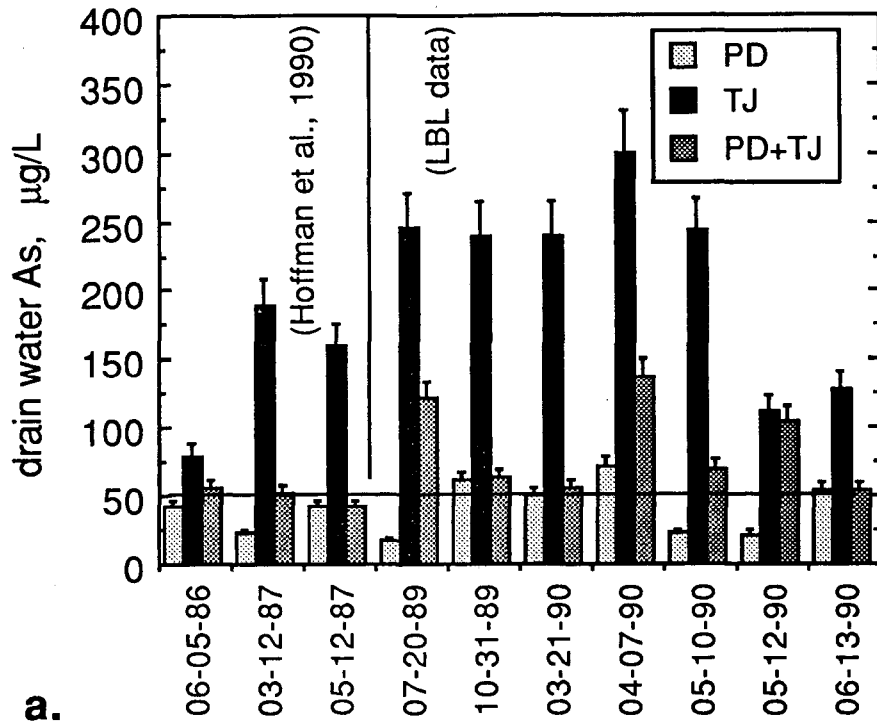


Figure 2.5. (a) Arsenic (As) concentrations in the TJ, Paiute Diversion (PD) and mixed (PD+TJ) waters at the point of discharge of the TJ Drain into the Paiute Diversion Drain in the SWMA. (b) Estimates of volumetric, TDS and As loading contributions of the TJ Drain into the combined PD+TJ Drain flows into the SWMA.

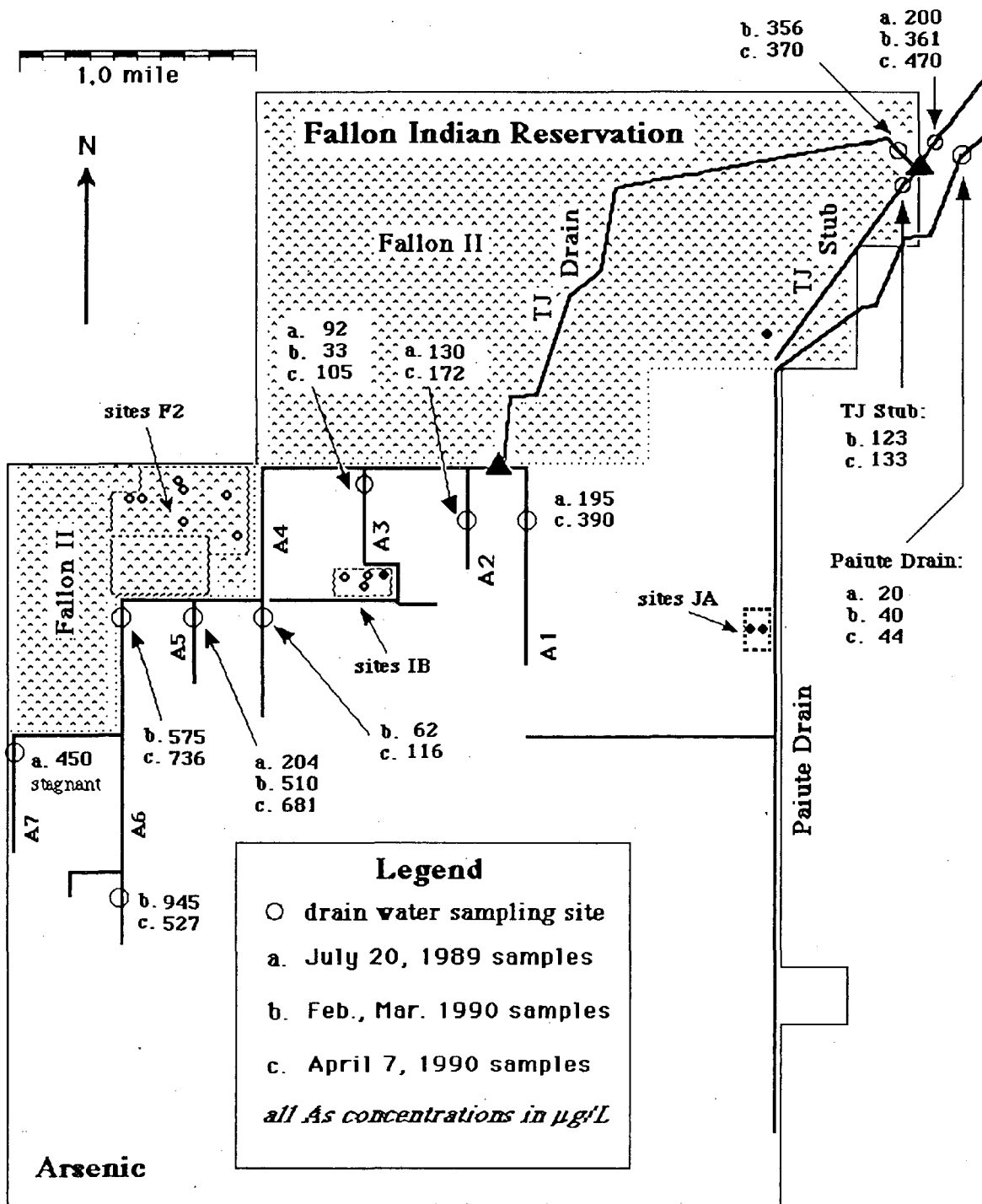


Figure 2.6. Map of FIR drain water arsenic (As) concentrations at several sampling times during 1989-90.

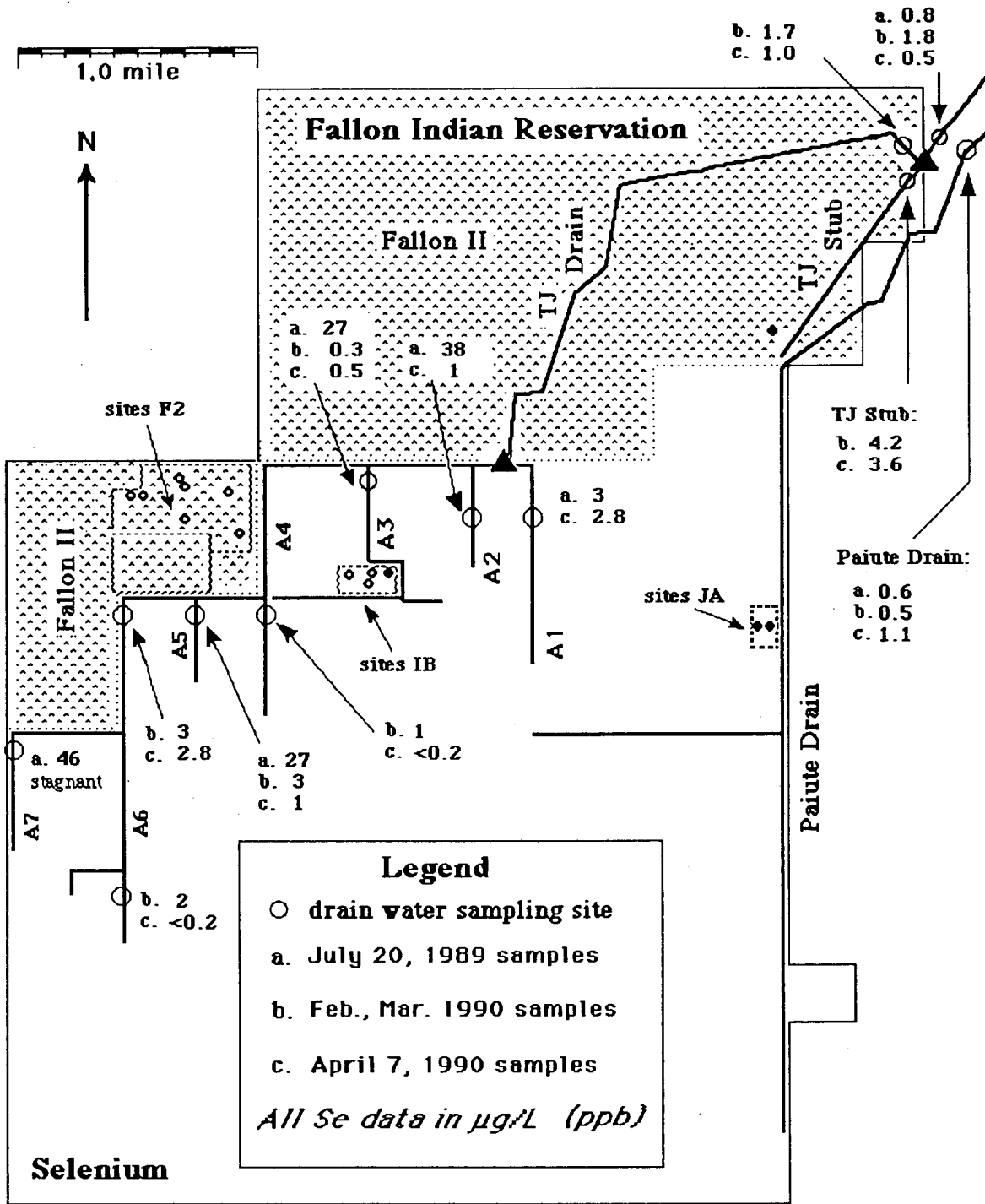


Figure 2.7. Map of FIR drain water selenium (Se) concentrations at several sampling times during 1989-90.

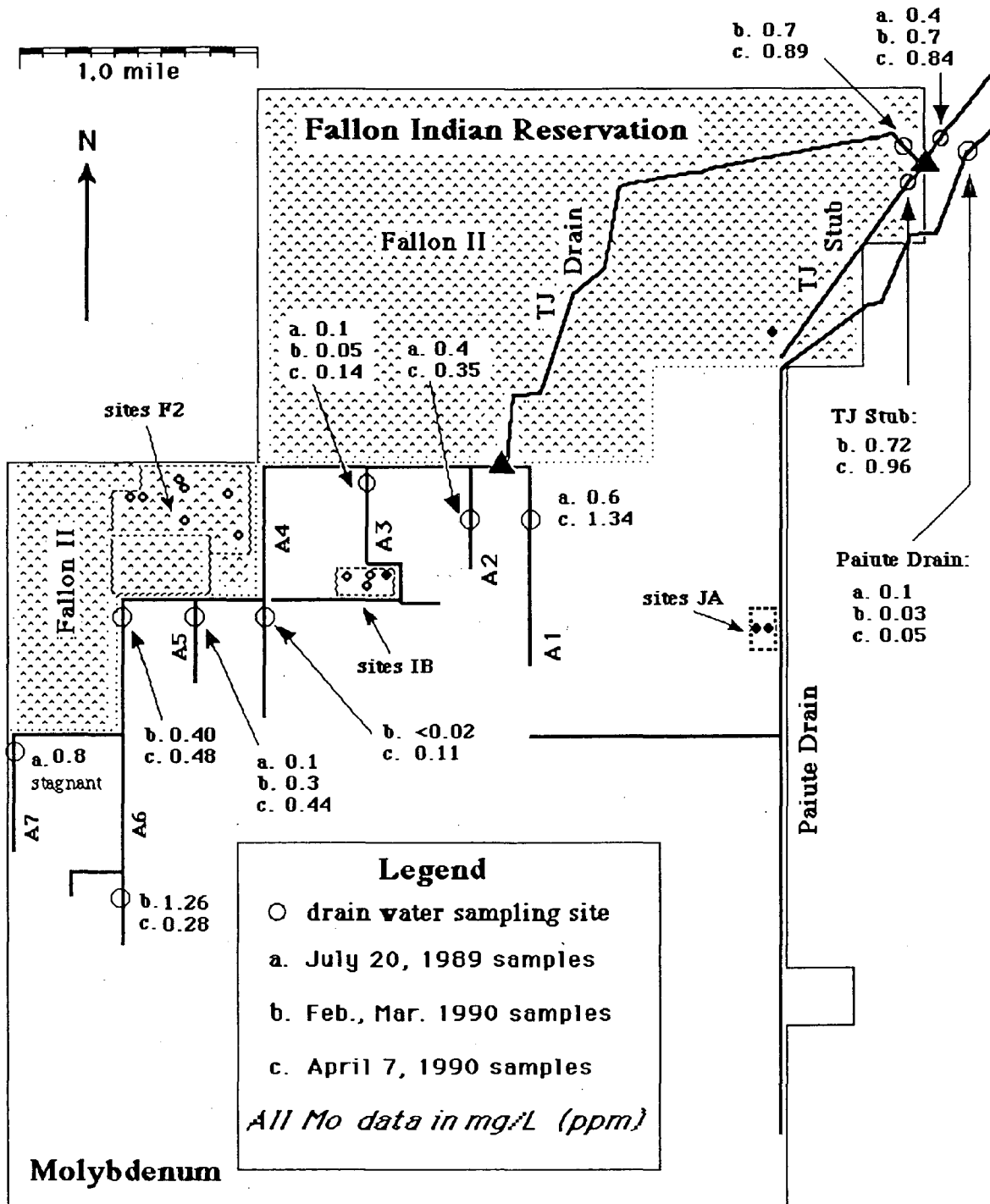


Figure 2.8. Map of FIR drain water molybdenum (Mo) concentrations at several sampling times during 1989-90.

2.2. Other Surface Waters

In order to gain an understanding of patterns in surface water quality in the areas surrounding FIR, a number of surface water samples have been collected within the Lahontan Valley, Carson Desert and surrounding mountain ranges. In particular, water samples have been collected in various canals, ditches, sloughs, pools and ponds within the surrounding area, including the SWMA and Carson Sink and from several seeps in the Stillwater Range. Water samples have also been collected at various locations along the Hunter Drain and Hunter Drain Sump on several occasions. In addition, a one time sampling of some surface waters located in and along the west side of the Carson Sink was performed. In a preliminary survey of the literature, no data on water quality in small pools within the Carson Sink was found. The Hunter Drain water samples and their analyses obtained as part of the present study, while limited, appear to be more comprehensive than any other currently available data set. The information obtained from the combination of these surface water quality data (to be described in this section) and information on the salinity and trace element composition in associated soils and salt crusts (to be discussed in Chapter 5) is useful in gaining a more complete understanding of the problem of trace element cycling in the region.

In the following subsections, three separate sets of surface water data will be addressed. These include the freshwater supply for the region distributed from the Lahontan Reservoir (Section 2.2.1), small pools within the Carson Sink (Section 2.2.2), and the Hunter Drain and Hunter Drain Sump (Section 2.2.3). Many of the other surface water features which have been sampled during the course of this work have been well characterized in previous studies. While the freshwater supply from the Lahontan Reservoir is also one of the systems previously characterized, the focus of subsection 2.2.1 will be limited to discussions concerning B and Se.

Several important features of certain surface waters in the Lahontan Valley and Carson Sink are discussed in the sections to follow. First, boron concentrations in even

the best available water supplies in the area will periodically exceed the 0.2 mg/L “effect” level for vertebrate embryos. Second, small pools within the Carson Sink, which are not only highly saline but highly concentrated with respect to trace elements of concern, may be indicative of naturally occurring periodic extremes in aquatic environments which contribute to wildlife die-offs. Application of drain waters over Carson Sink playas for evaporative disposal would introduce risks of generating such environments. Third, the Hunter Drain and Hunter Drain Sump are systems in which water quality is about as poor as that of the TJ Drain. Contrary to previous surveys of drain waters in the area, water samples collected from the Hunter Drain and Hunter Drain Sump in the present work have been shown to contain Se at concentrations which exceed suggested surface water quality goals.

2.2.1. Lahontan Reservoir Supply Water Boron and Selenium Concentrations

In the various recent studies concerning water quality problems in the SWMA, B and Se have been included in the list of trace elements which warrant attention. In Section 2.1, it was noted that high concentrations of B are characteristic of TJ Drain waters and that even the Paiute Drain waters have B concentrations which commonly exceed the effect level for vertebrate embryos. It was also noted that selenium concentrations in these drain waters have not been detected at elevated levels in the main drains. In view of attention directed at trace element concentrations in drain waters and their wildlife impacts, it is equally important to evaluate alternative water sources with respect to these same criteria. Such an investigation is not part of the LBL study. However, the water quality of the Lahontan Reservoir as it is distributed into the Fallon-Stillwater area is of direct interest in this study for two reasons. First, the extent to which a direct supply of Lahontan Reservoir water into the SWMA would meet water quality goals was of interest since the failure of TJ Drain waters in meeting these goals is central to this work. A second motivation for analyzing the irrigation supply water arose from the possibility that its chemical composition may enhance the leaching of certain trace elements of concern from local soils, including those of the FIR.

Regarding the first concern, it is clear that Lahontan Reservoir water quality is far superior to drain waters in most respects. This is especially apparent considering the low TDS (generally less than 300 mg/L). In the case of B however, waters originating from the Lahontan Reservoir appear to at least periodically exceed effect-levels or goals, as shown in Table 2.2. Although data in this table are from a very limited sampling, B concentrations in excess of 200 µg/L in "background" waters have also been reported in previous studies (USBR, 1987; and Hoffman *et al.*, 1990). It should also be noted that the Sagouspe Dam water is a mix of Lahontan Reservoir water and agricultural drain water. The implications of B concentrations in the water supply to the SWMA that are in the range of the cited effect level remain to be determined.

It is also worth noting that with respect to selenium concentrations, Lahontan Reservoir supply waters are similar to Se concentrations measured in TJ Drain waters (Table 2.2).

The second motivation for characterizing the Lahontan Reservoir water is related to the possibility that its composition may enhance extraction of certain trace elements from the soils of the FIR and surrounding areas. In particular, the relatively high fraction of the anionic composition of these waters attributed to bicarbonate and silica gave rise to the possibility that these oxyanions may enhance the displacement of certain trace elements residing in the area soils. In particular, these oxyanions are known to displace As, B, Se and Mo from soils. The commonly suggested mechanism for this displacement is ligand exchange, a particular example of anion exchange. Reviews of numerous studies addressing this process are provided by Parfitt (1978), Hingston (1981), and Sposito (1984).

In a preliminary set of experiments designed to address this second issue, various FIR soils were batch-extracted with Lahontan Reservoir waters in a 1:5 (soil:water) ratio for 1 hour. Replicates of the same soils were also batch-extracted with distilled water and with a simulated S-Line Canal water in the same manner. No significant differences were measured between any combination of water types. The lack of differences may have resulted from the low concentrations of HCO₃ and SiO₂, or the high water:soil ratio.

Column leaching experiments using irrigation waters with differing ionic compositions are presently being planned for testing the influence of other variables on mobilizing As and B for FIR soils.

Table 2.2. Arsenic, boron and selenium concentrations in Lahontan Reservoir-supplied surface waters.

| Source | Date | As μg/L | B μg/L | Se μg/L |
|--|---------|-----------------|-------------------|---------------------|
| Sagouspe Dam | 4/7/90 | 73 | 870 | 1.3 |
| S-Line Reservoir | 4/7/90 | 14 | 370 | 0.1 |
| S-Line Reservoir spillway | 4/7/90 | 14 | 370 | 2.0 |
| S-Line Canal before FIR | 4/7/90 | 13 | 330 | 0.1 |
| S-1 Canal at Hwy 50/Stillwater turnoff | 4/30/90 | 11 | 290 | 2.8 |
| Lahontan Reservoir near spillway | 11/1/90 | 24 | 550 | 0.3 |
| effect concentration | | | 200 ¹ | 2 to 5 ² |
| Nevada beneficial use criterion | | 40 ³ | 1000 ⁴ | 35 ³ |

¹Effects level for vertebrate embryos

²Surface water quality goal for Se

³State of Nevada criterion for aquatic life

⁴Beneficial use criterion for irrigation water

2.2.2. Trace Elements and Salinity in Small Pools in the Carson Sink

One of the possible options for drain water disposal which received some initial consideration was that of spreading the drain waters over a portion of the Carson Sink to permit rapid evaporation. The Sink, being an extensive saline playa, is clearly an environment which would not become significantly salinized by the introduction of drainwaters from a relatively small area. However, the risk of ponding substantial quantities of water in the Sink over periods of time long enough to attract wildlife appeared too great, given the composition of the TJ Drain waters even prior to evaporative concentration. In order to gain an appreciation for possible surface water composition under such conditions, samples of surface waters, salt crust and surface soils within the Sink were collected and analyzed.

The massive fish and bird die-offs which occurred in the Carson Sink in 1987 have increased general interest in water quality problems of the area. The extent to which such large-scale die-offs have occurred in the past, prior to the introduction of irrigated agriculture to the area does not appear to have been addressed in any substantial manner. It appears possible that such events may have occurred occasionally as wildlife population numbers and distributions responded to naturally occurring extremes in surface water availability (Rowe and Hoffman, 1987). During the drying cycle following an extremely wet season, wildlife attracted to such playa environments is exposed to surface waters of diminished quantity and quality. Thus, an interest in gaining a more comprehensive perspective on the possible trace element compositions of such waters provided additional motivation for the sampling and analyses of surface waters, surface soils and salt crusts in the Carson Sink. Surface water quality data will be described in this section. Analyses of salt crusts and surface soil samples from the Carson Sink will be included as part of Chapter 5.

As a first step towards gaining information on soluble trace element inventories within evaporatively concentrated surface waters of the Carson Sink, a limited sampling of this region was conducted. Surface waters of the Sink, which were sampled on October 29, 1989, are all located on the west side. The general locations of these sites are depicted in Fig. 1.1. Site CS3 is in the south end of a ditch, located 1 km east of Highway 95 and 6 km south of Parran Station. Both the ditch water and seep water flowing into the ditch were sampled. Site CS4 is a small pool located 2 km east of Hwy 95 and 5 km south of Parran Station. The pool had a nominal surface area of 30 m² and an average depth of about 0.5 m. The CS4 pool is situated in a depression which appears to be a result of past excavation. The CS3 and CS4 samples were collected because they were the only surface waters encountered within the main body of the playa at considerable distances from the SWMA and the terminus of the Carson River. A final surface water sample was collected in the Humboldt Slough (flowing into the Sink) at the Hwy 95 crossing. None of these sampling sites are located within the influence of agricultural drain waters.

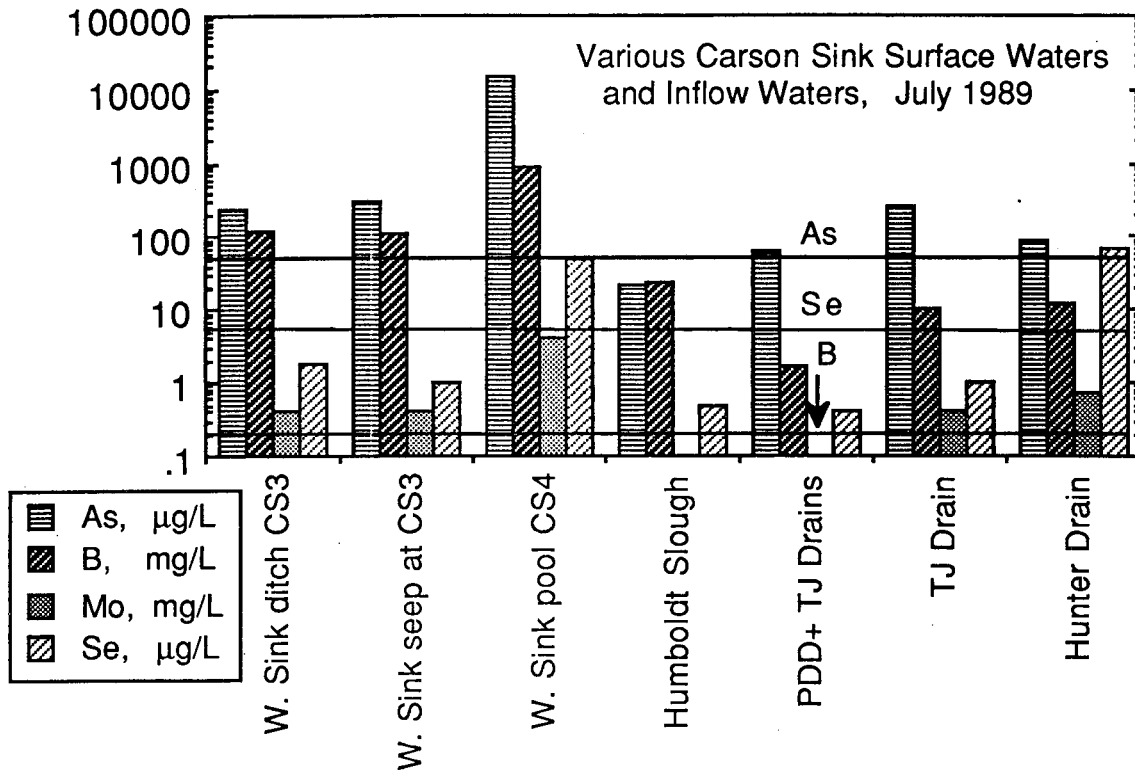


Figure 2.9. Arsenic, boron, molybdenum and selenium concentrations in various Carson Sink surface waters sampled in July 1989. Data from waters collected in the TJ, Paiute and Hunter Drains during the same period are included for comparison.

Results of analyses of these Sink waters for As, B, Mo and Se are summarized in Fig. 2.9, along with reference lines for water quality with respect to As, B and Se. For comparison, concentrations of these trace elements measured in July 1989 samples collected from the TJ, Paiute and Hunter Drains are also included. (The Hunter Drain and Hunter Drain Sump waters are described in the following Section 2.2.3.) The waters associated with sites CS3 and CS4 are well in excess of water quality goals for As and B. The highly concentrated pool CS4 contained 15,100 $\mu\text{g/L}$ As and was the only sample of this set with elevated Se concentrations (50 $\mu\text{g/L}$). Again, it is emphasized that the CS3 and CS4 sites may not be typical of the Carson Sink. However, the water quality data

from such sites may provide indications of possible trace element concentrations in waters during final stages of surface water evaporation following naturally occurring cycles of periodic ponding in the Sink. In such cases it would be important to recognize that the resulting water quality is not a consequence of agricultural activities.

2.2.3. Trace Elements and Salinity in the Hunter Drain and Hunter Drain Sump

Among the other drains which enter the SWMA, the Hunter Drain has been mentioned as having relatively poor water quality. This drain flows northward from near Stillwater Point Reservoir into Hunter Drain Sump, covering a distance of approximately 12 km (\approx 7 miles). The Hunter Drain Sump, located to the south of Tule Lake and to the east of South Lead Lake is reportedly only intermittently ponded to various shallow depths. Surface water samples were collected in the Sump during April and May 1990. Although flow rates through the Hunter Drain are generally very low in comparison with other local drains, including the TJ, its salinity and trace element concentrations were reported to be very high. This latter characteristic of the Hunter Drain, being similar to that of the TJ Drain, provided motivation for examining the Hunter Drain. The information obtained from these two systems should enhance our understanding of salt and trace element transport into the SWMA.

Surface water quality data collected in the Hunter Drain and Hunter Drain Sump are summarized in Table 2.3. The Hunter Drain data are also graphed in Figs. 2.10a and b. Salinities and concentrations of As, B, Mo and Se are considerably elevated and in some cases exceeded concentrations of the various constituents in the TJ Drain.

While these results are again in general agreement with previous works, the detection of significant concentrations of Se in these waters has not been reported by others. Because of this discrepancy, the Se analyses on many of these samples have been subjected to reanalyses. The reanalyses have supported the Se concentrations reported here. To further provide a somewhat independent check on these Se data, unfiltered samples were also collected alongside filtered samples on several occasions. The unfiltered samples were

collected to test for the possible presence of Se associated with suspended colloidal materials which would have been excluded from analyses by filtration of the samples in the field. The unfiltered samples were oxidized with hydrogen peroxide and treated according to the method outlined in Weres *et al.* (1989), and analyzed for Se along with distilled water controls which were subjected to the same hydrogen peroxide digestion. The unfiltered samples yielded between 1 and 2 µg/L higher Se concentrations than the filtered samples, indicating that a relatively small fraction of the Se is being transported in the suspended load of the drain.

Table 2.3. Hunter Drain and Hunter Drain Sump water quality information. "D" and "S" denote samples collected in various locations in the Drain and Sump respectively.

| Date | Site | E. C. dS/m | pH | As µg/L | B mg/L | Li mg/L | Mo mg/L | Se µg/L |
|---------|------|---------------|-----|------------|-----------|------------|------------|------------|
| 7/22/89 | D | 8.1 | 7.8 | 68 | 9.6 | na | 0.2 | 45.0 |
| 7/22/89 | D | 16.0 | 9.0 | 113 | 13.1 | na | 0.7 | 97.0 |
| 4/7/90 | D | 23.3 | 8.3 | 92 | 27.3 | 1.1 | 1.3 | 8.7 |
| 4/7/90 | D | 26.6 | 8.2 | 140 | 31.7 | 1.3 | 1.3 | 7.4 |
| 4/7/90 | D | 29.7 | 8.3 | 142 | 35.0 | 1.5 | 1.5 | 9.2 |
| 4/7/90 | D | 15.9 | 8.3 | 106 | 18.1 | 0.8 | 0.8 | 5.6 |
| 4/8/90 | D | 87.3 | 8.6 | 251 | 102.0 | 5.3 | 1.9 | 11.7 |
| 4/8/90 | D | 27.5 | 7.9 | 173 | 18.5 | 1.1 | 1.0 | 6.6 |
| 4/8/90 | D | 27.4 | 8.0 | 174 | 18.7 | 1.1 | 1.0 | 4.9 |
| 4/8/90 | S | 64.4 | 8.2 | 438 | 45.9 | 2.1 | 1.0 | 4.3 |
| 4/8/90 | S | 63.2 | 8.2 | 412 | 44.3 | 2.1 | 1.0 | 3.5 |
| 5/10/90 | D | 86.8 | 8.3 | 507 | 61.1 | 3.0 | 1.4 | 7.8 |
| 5/10/90 | D | 87.2 | 8.3 | 502 | 61.2 | 3.0 | 1.4 | 7.7 |
| 5/12/90 | D | 73.5 | 8.3 | 377 | 52.8 | 2.8 | 1.3 | 8.2 |
| 5/12/90 | S | 86.3 | 8.4 | 445 | 64.6 | 3.0 | 1.3 | 6.0 |
| 6/13/90 | D | 101.0 | 8.0 | 333 | 190.0 | 9.7 | 3.2 | 16.2 |
| 6/13/90 | D | 109.0 | 8.0 | 376 | 184.0 | 10.4 | 2.8 | 17.0 |
| 6/13/90 | D | 52.1 | 8.9 | 261 | 45.0 | 2.0 | 0.0 | 7.1 |
| 6/13/90 | D | 51.9 | 8.9 | 260 | 49.0 | 2.0 | 1.6 | 6.1 |

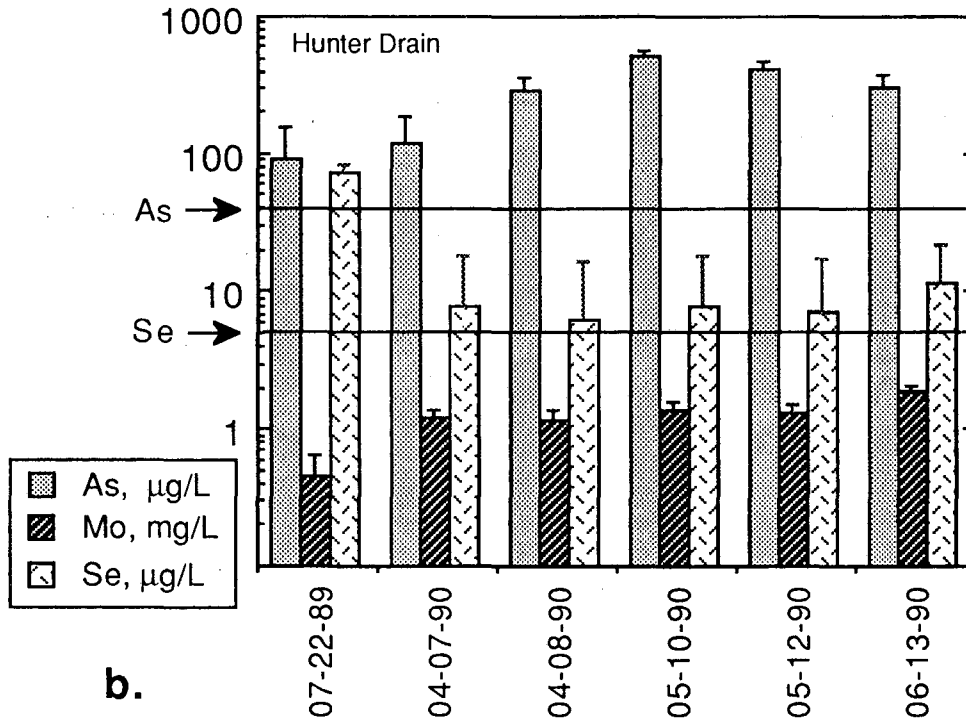
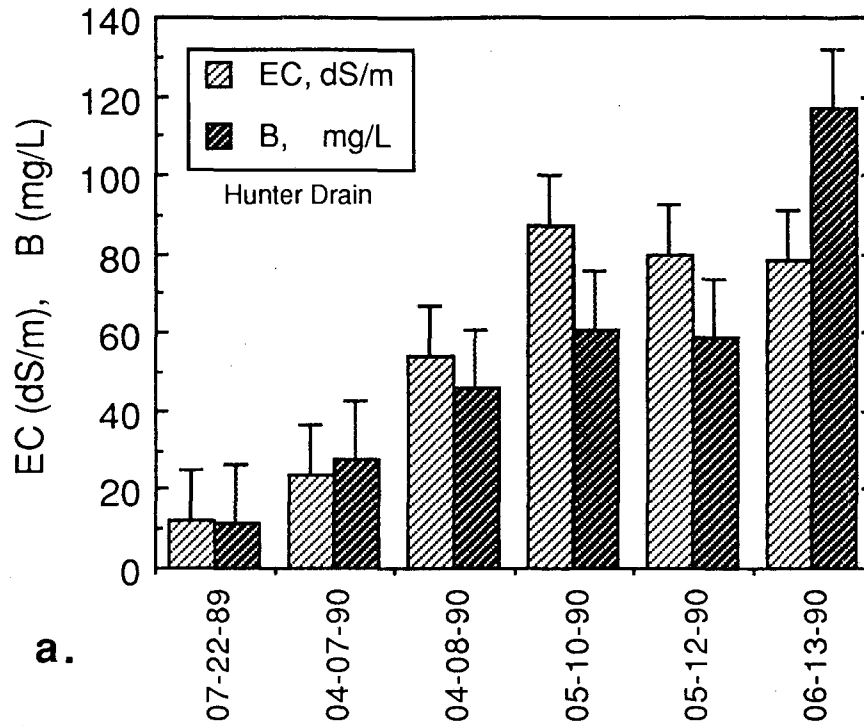


Figure 2.10. Hunter Drain water quality information. (a) Electrical conductivity (EC) and boron concentrations. (b) Arsenic, molybdenum and selenium concentrations. Error bars indicate standard errors of values obtained in 2 to 4 samples collected on a particular day.

3. TRACE ELEMENTS AND SALTS IN SOILS OF THE FALLON INDIAN RESERVATION

3.1. Introduction

The central task in this work is that of selecting a set of reasonable alternatives to managing drain waters presently being discharged via the TJ Drain into the SWMA. This selection process requires an understanding of all aspects of the system, rather than a narrow focus on the TJ Drain by itself. Such a narrow focus may have contributed to its original design and construction. In an effort to address the problem of drain water management in a proper context, characteristics of drain water quality as well as water quality in several other relevant surface water systems were considered in Chapter 2. Furthermore, the ponding of waters of any initial quality can under certain circumstances create the potentially unfavorable environments. This is demonstrated in Chapter 5 and has further limited the set of possible options for drain water disposal. Understanding the water quality and hydrogeologic characteristics of the groundwater discharging into the TJ Drain is of equal importance to resolving the problem of drain water management as is understanding characteristics of the soils in these newly developed lands. The pertinent information on the groundwater system at the FIR is presented in Chapter 4.

In Section 3.2, previous information on FIR soils is briefly reviewed. This is followed in Section 3.3 with the recent characterization of soil profiles selected in three different fields within the FIR, which have been under irrigation for different lengths of time. Soil profiles have been analyzed from each of these fields to estimate rates at which salts and relatively mobile trace elements are leached out of the local irrigated lands. Such information is useful in predicting future drain water composition under various alternatives. In Section 3.4, results from laboratory soil column leaching experiments are presented. The column leaching experiments provide a relatively quick assessment of rates of salt and

trace element leaching from a previously unirrigated site and supplement field characterization efforts directed at estimating future drain water quality.

The major points relating to the presence and movement of salts and trace elements from the FIR soils described in this chapter are as follows:

- In non-irrigated and in very recently irrigated farmlands, a considerable inventory of salts and soluble trace elements resides within the upper 2 m (\approx 6 ft).
- In well-leached fields which have been subjected to longer periods of irrigation (5 to 20 years), soil profiles have become relatively depleted of their initial inventories of salts and soluble trace elements.
- The column leaching experiments demonstrate that in well-drained soil profiles pore waters can have substantially improved water quality following the equivalent of several years of irrigation and drainage.
- Salt concentrations in the upper 1.7 m of soil decrease roughly exponentially with time of irrigation, with a half-life of about 5 years.
- Local heterogeneities, in particular zones with higher clay contents exert at least two important influences. First, infiltration rates through soils dominated by such layers are greatly diminished, which in turn greatly diminishes the rate of salt and trace element leaching. Second, zones of high clay content tend to retain certain trace elements of concern within the soil profile through adsorption. Arsenic and boron have been shown to exhibit such associations in these soils. An important related point concerning local clay-rich soil profiles is the fact that low leaching rates through such low permeability profiles will moderate rates of trace element and salt release from this sites.
- Minimizing the extent to which poor quality groundwater dominates drain water quality (under various drain configurations) even after surface soils have become well leached is crucial in achieving better quality drainage.

3.2. Review of Previous Data on FIR Soils

The principle sources of available, recent descriptions of soils within the Fallon Indian Reservation (FIR) include the Soil Survey of the Fallon-Fernley Area, Nevada (Dollarhide, U.S.D.A., 1975), the Fallon Indian Reservation Water Quality Report (U.S. Dept. of Interior, Bureau of Reclamation, Sept. 1987), the Fallon Indian Reservation Land Classification Delivery Order (CH2M Hill, U.S. Bureau of Reclamation, 1989 draft report), and a preliminary report on TJ Drain studies (S. Wilson and R. Tidball, USGS, Denver). The work of Dollarhide (1975), while being a general soil survey of the Lahontan Valley, does provide a detailed mapping of soil series within the FIR. This work provides detailed information on characteristic soil profiles within the FIR. Although general patterns of soil drainage and salinity are described adequately, no information concerning trace element distributions is available in the soil survey. The chemical composition of soil extracts (saturation paste and other extract procedures) and total elemental analyses from soil samples collected at the FIR are described in CH2M Hill (1989, draft). That report also includes information on infiltration rates, permeabilities, depths to barriers and depths to the water table. The chemical composition (including trace elements and pesticides) of drain waters in the FIR has been described in the works of U.S.B.R (1987), and Hoffman *et al.*, (1990). Additional information on the chemical composition of FIR soils, groundwaters, drainwaters and TJ drain sediments will soon be available in the work of S. Wilson and R. Tidball (USGS, Denver). Information provided in summary form in the following paragraphs was obtained from the above mentioned documents.

Soils of the FIR, being located in the Lahontan Valley, south of the Carson Sink, have developed from lacustrine (Pleistocene Lake Lahontan), alluvial (Carson River), and aeolian deposition of mixed parent materials. Soils within the FIR are physiographically situated on a mix of flood plains, lacustrine deposits, lake terraces and dunes. All mapping units are included within the entisol, aridisol and mollisol soil orders. Most undisturbed soils of the area are level to gently sloping. Where irrigation has been practiced, commonly using the border-strip method, land-levelling is often employed. Area farming is domi-

nated by alfalfa, with various grains and silage corn making up most of the remainder of irrigated crops (Dollarhide, 1975).

Irrigation water in the Lahontan Valley is supplied almost exclusively through the Truckee-Carson Irrigation District. Seepage losses from canals and deep percolation from irrigated fields in the valley have resulted in the rise of the shallow water table. Information on the extent of irrigation-induced shallow water table rise throughout the FIR was not located. It is noted in the CH2M Hill (1989 draft) report that local water table mounds appear to be associated with seepage from irrigation supply ditches and ponded irrigation water. Presently, the water table is from 1.0 to 4.0 m below the ground surface in most areas within the FIR.

The combined influences of soil development at the margins of a basin, low precipitation (0.13 m mean annual precipitation), high potential evaporation (1.19 m average May–October pan evaporation; Dollarhide, 1975), and generally shallow depth to the water table, have resulted in soils of the area that are typically saline and often alkaline. Soils exhibiting white, precipitated salts characteristic of saline conditions are commonly observed within the FIR. Black crusts of dispersed organic matter often associated with sodic conditions are much less common.

Native vegetation consists primarily of species adapted to arid, saline conditions of the area. The commonly occurring genera include *Sarcobatus* (greasewood), *Chrysothamnus* (rabbitbrush), *Tamarix* (salt cedar), *Atriplex* (shadscale, saltbrush), *Distichlis* (saltgrass) and *Suaeda* (desert blight). The *Tamarix* is actually introduced from Europe (Mozingo, 1987), as is *Kochia* which also is found in disturbed sites within the area.

The chemical composition of soils from an extensive sampling program within the FIR is provided in the CH2M Hill (1989 draft) report. Data from that study generally exhibit features characteristic of saline and saline-sodic soils of the western U.S. Saturation extract electrical conductivities generally ranged between 15 and 40 dS/m, with numerous sites exhibiting high sodium adsorption ratios ($SAR > 15$) as well. Saturation paste pH values commonly ranged between 7.5 and 8.5. Also included in the CH2M Hill study

were analyses for total As, hot-water extractable B and total Se. Total As concentrations ranged from 3.6 to 65 mg/(kg soil) (ppm). Hot-water extracted B ranged from 0.9 up to 85.8 mg/(kg soil). Of the 130 samples analyzed for total Se, only 13 contained Se in concentrations exceeding the detection limit of 2.0 mg/kg. The highest measured soil Se concentration was 3.4 mg/kg. It was concluded that the measured values of total As and Se and the hot-water extractable B were all within the range of expected values for soils of the region.

Although the unmanaged FIR soils are typically too saline for crop production, many of the soils are considered arable with appropriate levelling, irrigation and drainage (Dollarhide, 1975; CH2M Hill, 1989 draft). The CH2M Hill report identifies 3,115 acres within Fallon I (5,480 total acres) and 2210 acres within Fallon II (2,640 total acres) that are irrigable. Currently, approximately half of Fallon I is farmed. Approximately 600 acres of irrigated lands are presently drained by the A-Drain system which discharges into the TJ Drain. While irrigation and drainage technology are important considerations in the current and future developments of agriculture within the FIR, problems associated with TJ Drain water quality and drain water disposal have drawn more immediate concern.

3.3. Fallon Indian Reservation Soils Characterization and Monitoring

To evaluate interactions between irrigated soils and drain waters within the FIR, several soil profile monitoring sites have been selected and sampled. The soil profile sampling sites are shown in Fig. 1.2, in the fields labelled F2, IB and JA. Soil sampling and extraction procedures are described in Appendices A and B. A short discussion of problems associated with interpreting various soil analyses is provided at the beginning of Chapter 5. In this section, the rationale for selection of site sets F2, IB and JA will be presented, along with data on the soil profiles for which analyses have been performed.

The exceptionally poor quality water in the TJ Drain can be explained partially because it is a new drain (6 years old). Immediately after an irrigation-drainage system is installed, it is reasonable to expect high concentrations of a number of solutes moving out in

the drain waters. Over time, the average drainwater TDS and trace element loading is expected to exponentially diminish towards a composition limited by the combination of irrigation water quality and evapotranspirative concentration of the soil solution.

Given the importance of time as a variable in drain water quality, it would be ideal to take a series of soil profile samples through time at a single representative site, beginning before any irrigation water is ever applied and continuing annual sampling until a steady-state is reached. In this manner, the extent to which leaching from soil profiles contributes to poor drain water qualities can be determined. Restricted to a single site, such an effort could take tens of years. Furthermore, considerable variability of soil properties is commonly observed, even within a given field. A reasonable, expedient alternative is to select similar fields which have been irrigated and drained for different lengths of time. Under ideal conditions, a one-time sampling across a range of field ages would provide the desired information. Numerous interfering variables, most notably the natural variability found in soil properties, prevent simple interpretations of such an ideal sampling plan. Nevertheless, the sampling of fields with a wide range of irrigation histories provides us with perhaps the only realistic approach to quickly answering questions concerning time trends in drain water quality. In the western San Joaquin Valley, such an approach was taken in the research conducted by Fuji and Deverel (1989) in deducing the history of Se movement out of irrigated and drained fields.

In the present search for TJ Drain water disposal solutions, soils from fields designated F2, IB and JA were chosen to provide information of possible time trends in drain water quality. The F2, IB and JA fields have experienced approximately 1, 5 and 20 years of irrigation-drainage respectively (up to the time of sample collections, between December 1989 and March 1990).

3.3.1. F2 Sites

The F2 field is part of the New Tribal Farm area of Fallon II (Fig. 1.2). Although 56.3 hectares (139 acres) of this portion of Fallon II were initially levelled for irrigation,

only 34.2 hectares (84.5 acres) are presently being farmed. (The "F2" label is used in this study to refer to the New Tribal Farm region of Fallon 2.) During 1989, approximately 0.5 m (1.6 acre-ft/acre) of irrigation water was applied. This was the first irrigation of this site. Although the TJ Drain runs along the eastern boundary of the F2 field, the western edge of the field is about 0.8 km from the drain. The soil survey (Dollarhide, 1975) map indicates that most of the F2 field is a Sagouspe loamy sand (Aquic Xerofluvent subgroup). A small portion of the northern section of F2 is mapped as playa soils (undifferentiated). An old stream channel is suggested from the outline of the playa soil boundary. As a result of land levelling in preparing the F2 field, much of the original soil surface has been disturbed. Nevertheless, the remaining loamy sand and sandy loam surface soil textures which dominate F2 are consistent with the soil survey map. Despite the prevalence of loamy sand and sandy loam surface textures, clay contents in the surface and near surface soil horizons were observed to have a significant influence in distributions of soil moisture and consequently in distributions of winter rye seedling emergence. Also evident in some areas of F2 are stubble of native shrubs which have been cleared away.

Seven soil profile sampling sites were selected in F2. These sites were selected to provide a range of soil properties. Variations in surface features such as soil texture, seedling emergence and evidence of saline or sodic conditions were used to select the sites. Each site was hand-augered in duplicate core profiles down to approximately 3.00 m (0.10 m increments). The majority of profiles were predominantly sandy loam to fine sand, although stratification with finer-texture lenses was common. At one site, about 75% of the profile was comprised of loam to clay textures. During the sampling period (March 4 to 19, 1990), the water table at F2 was at about 2.9 m below the soil surface. Evidence for minor perched water table conditions was encountered at the F2 site.

To date, four profiles from the F2 field have been processed in the laboratory. These F2 sites are referred to as F2-s1, F2-s2, F2-s4 and F2-s7. Each site has been sampled in duplicate or triplicate to various depths. The F2-s1, F2-s4 and F2-S7 profiles appeared to be characteristic of most of the F2 field based upon superficial field observations of surface

soil texture, soil moisture and seedling emergence. The F2-s2 site had a dispersed crust of dark, organic matter coating mineral particles. This site, which appeared sodic, had no vegetation. Data from these four profiles are summarized in the following sections with respect to patterns of soil salinity, arsenic, boron and selenium concentrations.

3.3.1a. Soil Salinity at the F2 Sites

The salinity of the F2-s1, F2-s2, F2-s4 and F2-s7 soil profiles are shown in Fig. 3.1a, through electrical conductivity (EC) data. (The letters A, B, N and S following a soil site number represent replicates from the particular site.) In this figure, as well as in others to follow, the ECs of 1:5 soil:water extracts (corrected to 25°C) are linearly extrapolated to initial soil water contents. This procedure is not strictly valid for estimating actual field soil solution ECs due to both mineral dissolution and ion pairing effects, however it does provide a qualitative indication of both overall soil salinity and distribution of salts within the profile. The EC values from saturation-paste extracts (EC_e) would be roughly equal to between one third to one half of the corresponding EC values presented in these depth profile figures.

Two distinct patterns of salt distribution are shown in Fig. 3.1a. The majority of soil profiles exhibit a fairly consistent pattern with respect to salt distributions. However, the F2-s2 profiles differ significantly from the general patterns. The two types of salinity profiles will be discussed in turn.

In the case of the F2-s1, F2-s4 and F2-s7 sites, the uppermost 1.0 m of the soil profiles appears to have been leached of its initially high inventory of salts. It should be noted that data from adjacent unirrigated soil profiles (sites F2-O and F2-OS1, not shown in the figure) indicate that a zone of relatively lower soil salinity may be characteristic of the sandier F2 soils prior to its agricultural development. This lower salinity zone extends down to a depth of about 0.30 m in the two unirrigated sites. Sandier surface soils tend to be less saline because they are both easily leached by precipitation and they are less effective at evaporative "capillary" conduction of soil salts back towards the surface. Below a

depth of 1.0 m all of the F2 soil profiles remain high in salinity. As mentioned previously, about 0.5 m of irrigation water has been applied on the developed portion of Fallon II during 1989. Considerable variability in actual infiltration of irrigation waters is expected. A comparison of the average depths of the less saline surface zones in the irrigated and unirrigated sites indicates that approximately 0.50 m of the near-surface zone in the F2 fields have been leached during the first year of irrigation. This depth of soil profile leaching corresponds to a net infiltration (actual infiltration minus evapotranspiration) of about 0.1 m H₂O over the past season.

As noted previously, the F2-s2 site is atypical of the F2 field with respect to surface features. Unlike the F2 profiles just described, the F2-s2 site was expected to be highly saline based upon its finer surface soil texture and dispersed, structureless surface crust. Unlike the case of the sandier surfaces discussed previously, the finer textured soil surfaces retain salts because infiltration rates tend to be lower and evaporative concentration of salts tends to be more significant. Particle-size analyses of the F2-s2 soils indicated that the clay and silt size particle-size fractions accounted for between 30 to 95% of the soil mass in the upper 1.10 m of this site. None of the other sites shown in Fig. 3.1a had such high percentages of fine particle-sizes at the surface. Dispersed organic matter films at the surface of this site also suggested that the profile would be relatively higher in sodium content than the other F2 sites. No emergence of rye seedlings was observed at the F2-s2 site. The salinity profiles obtained by water-extractions of the F2-s2 soils are consistent with field observations. As seen in Fig. 3.1a, the salinity of the F2-s2 site is not diminished in the upper 0.80 m. Rather, a pronounced salinity maximum was measured over the depths ranging from 0.10 to 0.50 m. Such a distribution of soluble salts is indicative of a lack of leaching of this profile by the applied irrigation water. It appears likely that in this and other local regions of finer surface soil texture and low surface soil permeability, irrigation waters have run off to infiltrate into more permeable adjacent soils. These less readily leached areas can in fact have salt contents that are higher than other more permeable unirrigated soils. Another feature of the low permeability of such clay-influenced profiles is that

the rate at which salts and trace elements are leached into drainwaters is limited by the restricted infiltration.

The salinity profile data from the F2 field indicate that a net leaching of about 0.50 m of most of the soils was achieved as a result of the application of 0.50 m of irrigation water. Much lower rates of profile leaching can be expected in less permeable areas of the field. It is likely that the large majority of the soil profiles in the F2 field were highly saline at depths greater than 0.8 m at the time of sample collections. Approximately 4 more years under similar irrigation water applications would be required to leach the upper 3.0 m of the soil profile.

3.3.1b. Extractable Boron at the F2 Sites

Presently, only water-extracted and phosphate-extracted soil boron analyses have been performed. Boron concentrations obtained from phosphate extractions were usually slightly greater than those obtained with water extractions. Extractable B is also commonly correlated with clay content. The association of B with clays is well documented (e.g. Hingston, 1964; Sims and Bingham, 1968a,b; Elrashidi and O'Conner, 1982; Adriano, 1986).

The profiles of water-extracted B in the F2 field sites are summarized in Fig. 3.1b. The profiles are quite variable in B content, but a couple of general patterns can be discerned. Extracted B concentrations are relatively low within the upper portions of the readily leached soil profiles (as indicated in the data from site F2-s1, F2-s4 and F2-s7). The F2-s2 site B profiles contained high B concentrations in the surface soil zone, indicative of only limited leaching. This pattern is consistent with the corresponding patterns in salinity distributions noted previously. Profiles generally showed increases in B concentrations at deeper portions of the soil profile (2.0 to 3.0 m below the soil surface). The deeper portions of these soil profiles have water-extracted B concentrations in the range of 3 to 8 mg(kg soil)⁻¹ in the F2-s1, F2-s4 and F2-s7 sites. When normalized to the original water contents of these soils, B concentrations in the range of 9 to 30 mg L⁻¹. This range en

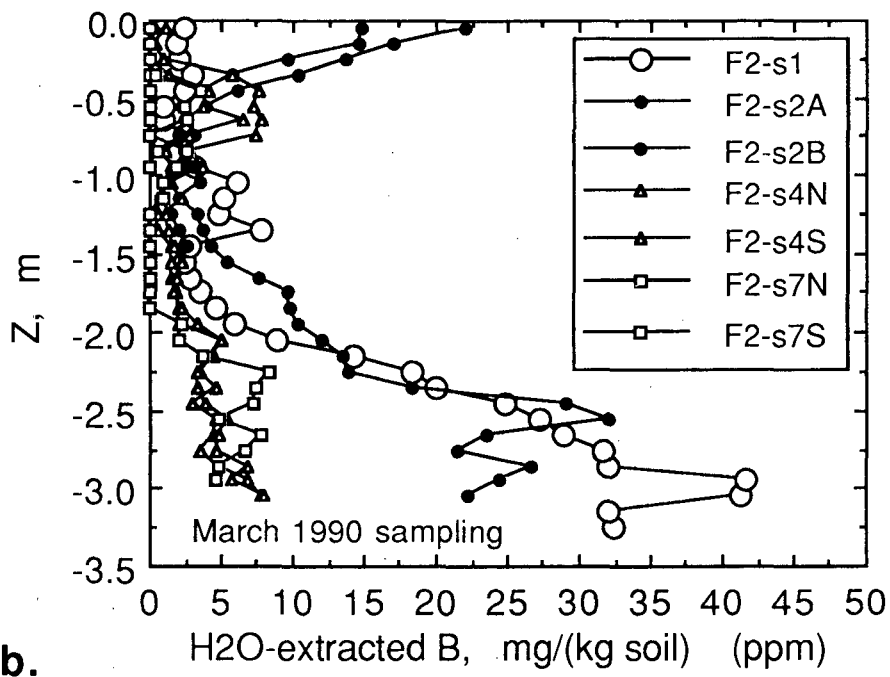
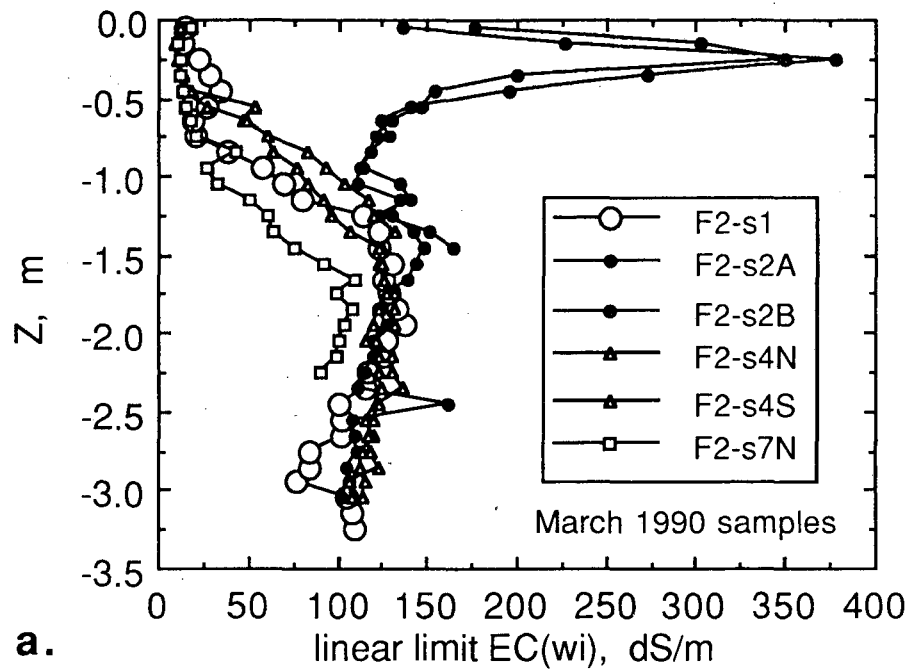


Figure 3.1. (a) Electrical conductivity profiles of soil water-extracts from irrigated F2 soil, linearly normalized to field soil water contents. (b) Water-extracted boron concentration profiles from irrigated F2 soil.

compasses most of the B concentrations measured in the A and TJ Drains. The relatively unleached F2-s2 profiles have maxima in B concentrations at both the soil surface and at the bottom of the sampled zone. The high B concentrations in the surface zone are consistent with the high salinity and minimal leaching of this site. High concentrations of B in the deeper portions of the soil profiles at site F2-s1 and F2-s2 are correlated with increased clay contents in these cases.

Hot-water extractions of soils are commonly employed for estimating plant-available B (Bingham, 1982). This technique has been tested in on a number of FIR soils. As noted in Appendix B, boron concentrations obtained with the hot-water extraction procedure generally yielded about 50% more than B obtained in the 1:5 soil water extracts commonly used in this work.

3.3.1c. Soil Arsenic at the F2 Sites

The profile of water-extracted soil As at the F2-s1, s2, s4 and s7 sites are shown in Fig. 3.1c. Water-extracted As concentrations commonly constitute only a small fraction of the total soil As inventory. Phosphate-extractions of soils generally provide greater releases of As from the combined effects of desorption and dissolution. Phosphate-extracted As profiles from site F2-s4N are shown along with corresponding water-extracted As data in Fig. 3.2a. An example of the correlation between water-extracted and phosphate-extracted As is provided in Fig. 3.2b. The total soil As concentrations are generally considerably higher than even the phosphate-extracted quantities. Total soil As data have only been obtained in a small number of samples. One of these sample sets is from site F2-s4. The limited number of total As data from this site are also plotted in Fig. 3.2a. The comparisons between water-extracted, phosphate-extracted and total As concentrations shown in this figure illustrate the commonly observed large differences between these various inventories. Only the water-extracted and phosphate-extracted fractions of As are expected to be available for slow leaching out of these soils. The total As analyses in the F2 soils is largely a measure of As in various less available solid phases.

The phosphate-extracted As concentrations have commonly been observed to be well correlated with clay contents of the individual soil samples. Such correlations have been well documented in the literature (e.g. Parfitt, 1978; Hingston, 1981). These correlations are especially high, given the measured variation in pH (Frost and Griffin, 1977), and the expected variations in composition of clays and arsenate concentration within a given profile. Similar correlations have been noted in a number of studies on As and other oxyanions (B, PO₄, SeO₃) in soils (e.g. Hingston, 1981). The As concentrations observed in the F2 field are not particularly high compared to other Lahontan Valley soils. However, the higher As concentrations in various portions of the profiles would exceed the Nevada surface water quality criterion if the As in the soil solution were to leach into the drain without attenuation.

3.3.1d. Soil Selenium at the F2 Sites

The water-extracted selenium profiles (Fig. 3.1d) are roughly correlated with the EC profiles in the upper 1.0 m of the F2 soil profiles. Since the extracted Se is primarily the freely soluble selenate (Se(VI)), such patterns are consistent with leaching of the soluble surface inventory. Only the F2-s2 profile has relatively high soluble Se concentrations in the near surface regions, consistent with the suggested lack of leaching at this site. Along with the soluble Se concentrations in the upper portion of the F2-s2 site, subsurface portions of the F2-s4 and F2-s7 sites also exhibit moderately high Se concentrations. These higher Se concentrations would clearly exceed effect levels if the Se remained in solution as soil waters leach into drains. However, the generally low Se concentrations obtained in all of the deepest samples at the F2 field suggest that Se is not being released into local groundwaters and drains at elevated concentrations. The large declines in extracted Se below about 2.2 m may be due to the presence of the shallow water table at about -2.8 m. Under reducing conditions often associated with this zone, removal of selenate from the soil solution via reduction to selenite (Se(IV)) and subsequent adsorption of selenite is plausible. In such situations a relative increase in the Se(IV):Se(VI) is expected. In the F2

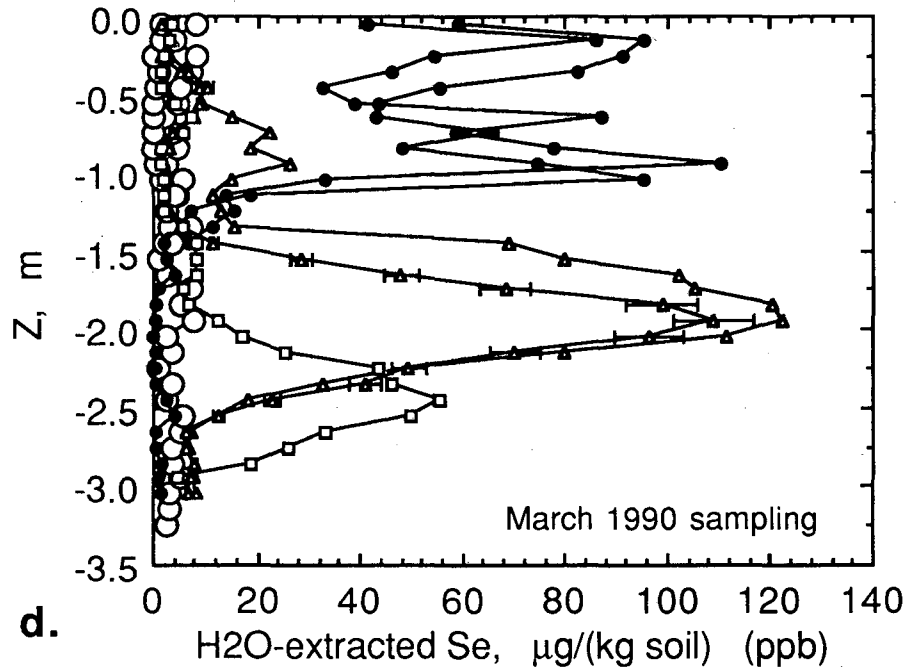
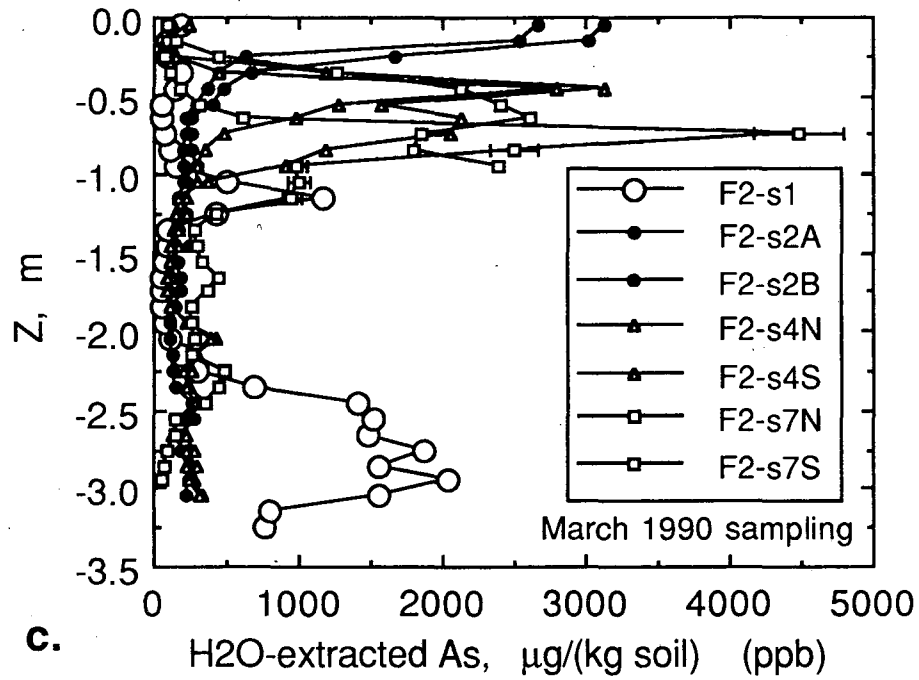


Figure 3.1. (c) Water-extracted arsenic concentration profiles from irrigated F2 fields. (d) Water-extracted selenium concentration profiles from irrigated F2 soil.

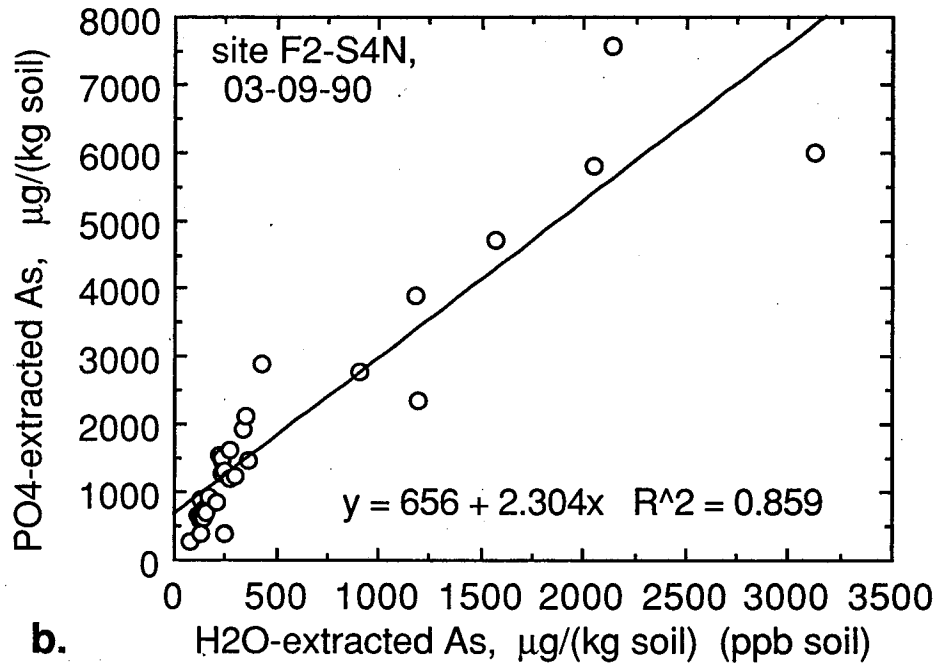
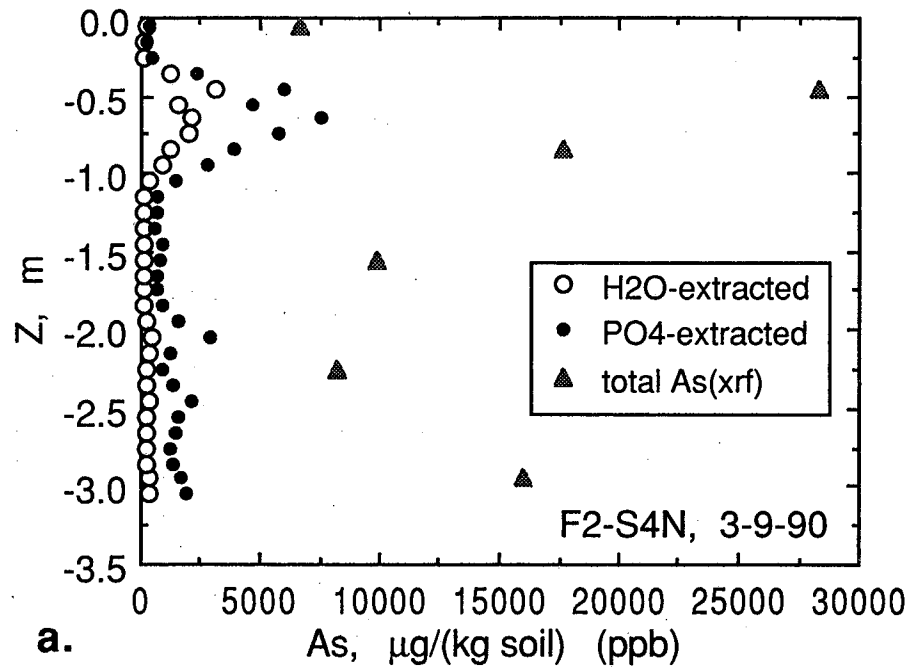


Figure 3.2. (a) Water-extracted, phosphate-extracted and total soil arsenic profiles from the F2-S4N soil sampling site. (b) Correlations between water-extracted and phosphate-extracted As concentrations from the F2-s4N site.

selenite data (not shown), such an increase is difficult to discern. The higher Se concentrations in the central region of the profile may compromise drain water quality in the event that such high concentrations of Se move into the drainwaters without reductive removal.

3.3.2. IB Sites

The IB field has been irrigated for 5 years. Like the F2 field, the IB field was leveled prior to irrigation. The TJ drain runs along the southern, eastern and about half of the northern boundary of the IB field so that all areas of the field are within about 0.2 km to a drain branch. Cement-lined irrigation canals also run along the south, west and about half of the north boundary of this field. The nominally 8 hectare (\approx 20 acre) field was initially cropped for rye, with somewhat poor yields. During the past two years, alfalfa has been grown. The soils of the IB field are mapped primarily as an Appian clay substratum—Tipperary Complex (Dollarhide, 1975). The Appian series is included in the Typic Natrarigid subgroup, while the Tipperary series belongs to the Typic Torripsamment subgroup. The southern edge of the IB field is mapped as a Dia loam (Fluvaquentic Haploxeroll).

A total of four sampling sites were selected in the IB field. Soil samples were collected between February 9 and February 13, 1990. Duplicate to triplicate auger samples were obtained at each of these sites. The profiles were primarily sandy loam to fine sand, although stratification with silty loam to clay textured soils was frequently encountered. In several cases these finer-textured horizons appear to exert significant influences on soil water drainage. The finer-textured strata were up to 0.7 m in thickness and located at depths ranging from 0.35 m down to 1.30 m below the soil surface. Minor perched water table conditions were observed at 2 sites. At one of the IB sites, mottles indicative of impeded drainage and variable redox conditions were observed throughout most of the upper 0.9 m of the profile. In small areas of the field alfalfa growth is poor and regrowth of invading *Tamarix*, *Kochia* and *Distichlis* has occurred. Three sets of soil samples have been processed in the laboratory. These samples are from the sites designated IBE-s1, IB-s2 and IB-s3.

The IB-E-s1 site is located 15 m west of the LBL IB-E series wells, in the northeast corner of the IB field. The soil profile was dominated by loamy sand to sandy loam, with minor finer textured inclusions between depths of 0.40 m and 0.60 m. The water table was approximately 2.5 m below the soil surface. Core samples obtained at the nearby IB-E well indicated that a clay layer would have been reached at a slightly deeper interval. Alfalfa growth at this site appears to have been moderately successful. The sample set consisted of a single profile sampled down to 3.00 m and supplemental profiles located 1.0 m and 2.0 m away from the main profile. The supplemental profiles were sampled down to 1.00 m.

The IB-s2 site, located about 149 m west of the IB-E-s1 site, is situated in a portion of the IB field which exhibited symptoms of poor drainage and very high salinity. Although such areas were not typical of this field, this site was selected in order to characterize a rather extreme case of impaired leaching. A discontinuous salt crust covered this site. Alfalfa growth at this site was poor and largely outcompeted by *Distichlis* and *Tamarix* shoots. The IB-s2 site was sampled in a manner similar to that used at the IB-E-s1 site. Localized lenses of finer texture soils were present, especially at the surface and at about 0.60 to 0.80 m below the surface. Evidence of perched water table conditions and localized bands of mottled soils were observed during sampling.

The IB-s3 site is located about 45 m east of the western edge and about 50 m south of the northern edge of the IB field. The site appeared superficially similar to much of the rest of the field, including site IB-E-s1. This site appeared to have yielded a moderate to good crop of alfalfa during 1990. Although the upper 1.00 m of this soil profile was loamy sand to sandy loam in texture, the soil profile graded to clay textures below about 1.3 m and a shallow water table prevented sample collections deeper than 1.50 m.

The remaining, unanalyzed site (IB-s4) was similar to the IB-E-s1 site in terms of field-observed characteristics. In the following subsections EC, B, As and Se profiles from water-extracts of IB field soils will be reviewed. The various analyses from replicate

samples were generally similar. Therefore, in the interest of clarity, only the data from the central sample profiles from each site will be shown in the graphs to follow.

3.3.2a. Soil Salinity at the IB Sites

Soil water-extract EC profiles are shown in Fig. 3.3a. The water-extract EC data were linearly normalized back to field water contents as described previously for the F2 soils. The IB-E-s1 and IB-s3 profiles reflect the influence of considerable leaching. Salinities throughout these profiles are generally less than those measured in the F2 sites. This was to be expected based upon differences in irrigation histories of the two fields. The exceptional case of the IB-s2 site is nevertheless consistent with the field observations of poor leaching. A salt crust at the soil surface and a shallow zone of persistently high salinity in the zone of finer texture soils was anticipated. It is interesting to note, that in the coarser texture subsoils, salinities at the IB-s2 site are relatively low. This may indicate that the lower permeability strata encountered at and near the surface have a very limited lateral extent. This is consistent with field observations of the surface soil textures at this site. (An interesting contrast will be illustrated in the JA field.)

3.3.2b. Soil Boron at the IB Sites

Water-extracted boron profiles from the IB field are presented in Fig. 3.3b. With the exception of the upper portion of the IB-s2 profile, these soils appear to be relatively well leached of B. In the IB-E-s1 and IB-s3 profiles, water extractable B concentrations of less than 1 mg/(kg soil) (< 1 ppm) were obtained in the upper sections of the profiles. The increased B concentration at the 1.35 m and 1.45 m samples at IB-s3 may have resulted from a combination of retention on clays which are dominant at that depth and slow leaching of the original soil solution at this site.

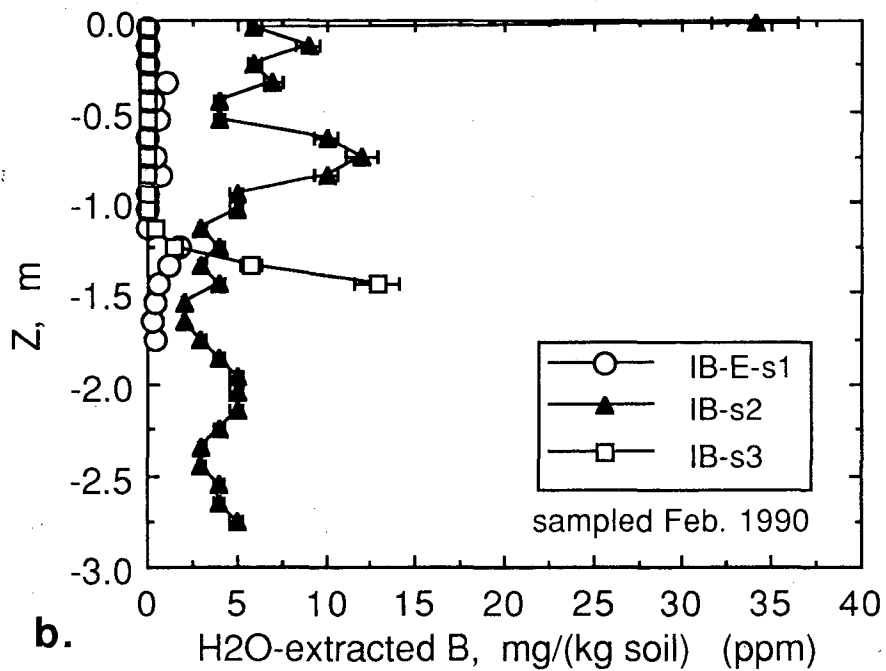
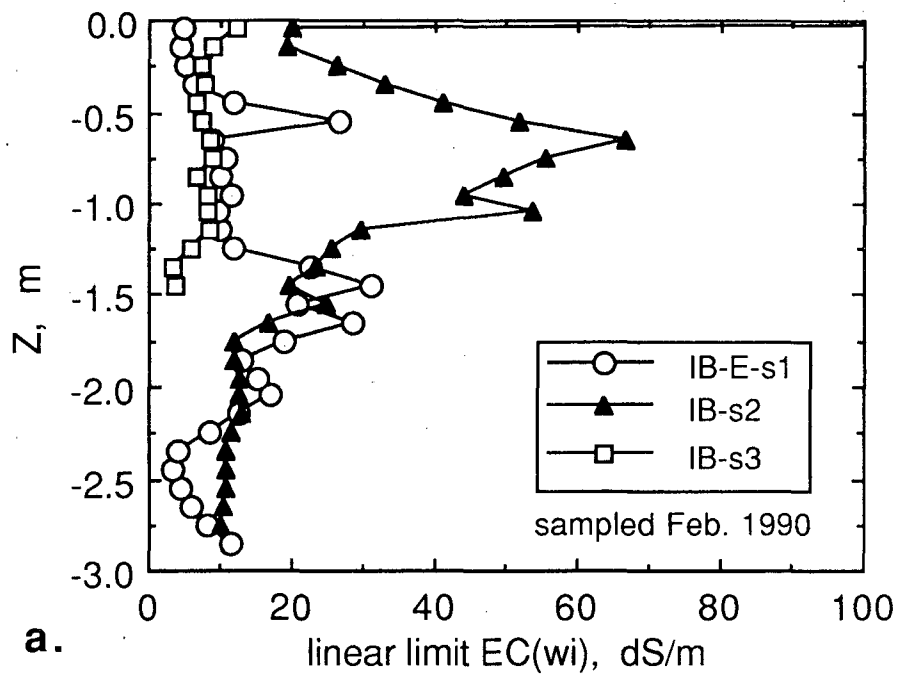


Figure 3.3. (a) Electrical conductivity profiles of soil-water extracts from the IB soil profiles, linearly normalized to field soil water contents. (b) Water-extracted boron concentration profiles from IB soil.

3.3.2c. Soil Arsenic at the IB Sites

Water-extracted arsenic profiles from the IB field are shown in Fig. 3.4a. The As depth profiles shown here are very irregular. Differences in initial soluble As inventories may explain part of the differences between sites. The IB-E-s1 and IB-s3 As profiles may have resulted from a combination of leaching of the surface zone and readsorption at greater depths. The IB-s2 profile may reflect a lower initial soluble As inventory. Much uncertainty is associated with these possible explanations where both transport and desorption (and/or dissolution) are significant sources of extracted As. A clearer example of both As and B profiles in which adsorption/desorption effects appear dominant is provided in Section 3.3.3b and c.

Water-extracted Se profiles from the IB field are presented in Fig. 3.4b. These soluble Se concentrations are on average only about one fifth of the concentrations obtained in the F2 soils. However, even the average extract Se concentrations from the IB field, when normalized back to concentrations per unit soil water mass would exceed effect levels by a factor of about 5. The high Se concentrations extracted from the deepest soil sample at the IB-s2 site indicate that reductive removal of Se(VI) has not occurred within the sampled interval.

3.3.3. JA Sites

The JA field has been irrigated and drained for at least 20 years and was selected in order to help assess the potential long-term quality of soil water, groundwater and drain water within the FIR. The field drains into the Paiute Drain, which runs along its eastern edge. The nominally 3.6 hectare (9 acre) field is cropped for alfalfa with generally high yields. At the time of sampling (December, 1989), the water table was located at about 1.7 m below the soil surface. Most of the field is well drained, although portions of the western edge are reportedly of low permeability and of low productivity (Mr. J. Allen, personal communication, December, 1989). Most of the JA field is mapped as East Fork clay loam,

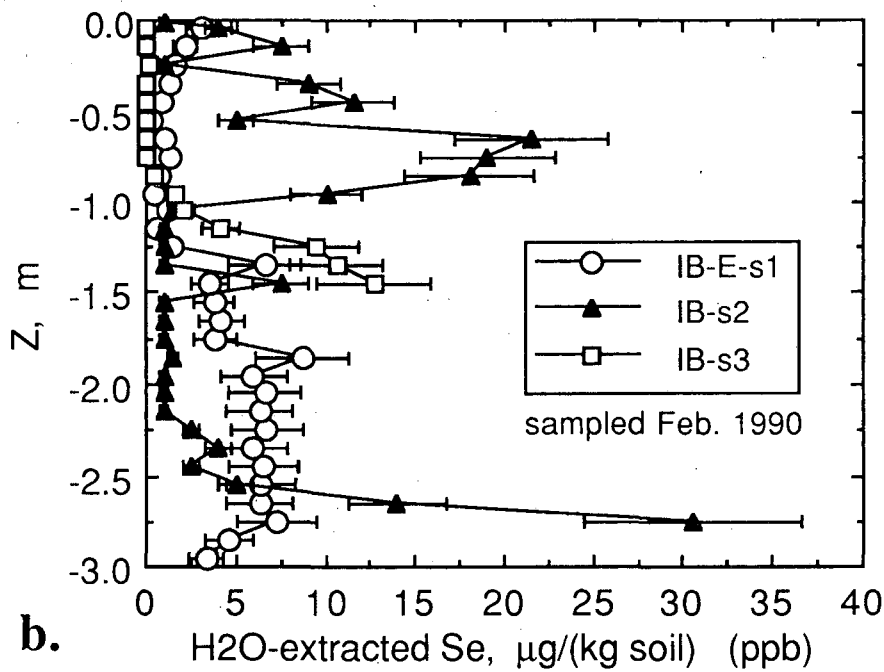
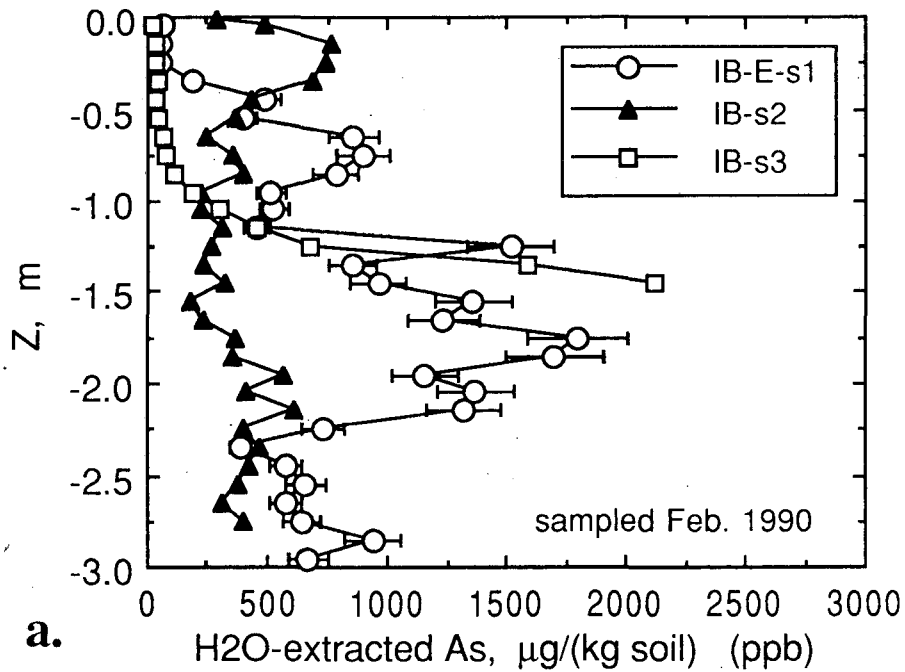


Figure 3.4. (a) Water-extracted arsenic concentration profiles from the IB field. (b) Water-extracted selenium concentration profiles from IB soil.

slightly saline (Fluvaquentic Haploxeroll) (Dollarhide, 1975). The soils of northern edge of the field are mapped as a Ragtown clay loam, slightly saline (Typic Torriorthent).

Two sampling sites were selected on opposite sides of the JA field. The east site, JA-E, is situated in soil considered typical of most of the field. The west site, JA-W, is centered in a less productive portion of the field. Although both sites are mapped under the East Fork series, the JA-W site exhibits profile characteristics associated more closely with the Ragtown series.

Particle-size analyses have been completed for the two JA sites and are depicted in Fig. 3.5a,b. The contrasts in textures between the two profiles is striking. The JA-E profile is primarily sandy loam in texture down to a depth of 0.7 m, where a thin (≈ 0.15 m) clay lense is encountered. Below the clay lense, the sandy loam texture quickly grades into fine sands. In the case of the JA-W site, the clay and clay loam textures dominates nearly the full length of the sample profile.

Electrical conductivity profiles of soil solution extracts linearly extrapolated to original soil water contents are shown in Fig. 3.6a for both sites. The contrast in salinity between the two sites is clear and can probably be related back to the textural differences. It is reasonable to expect that the primarily sandy loam profile of JA-E is generally more permeable and more effectively drained than the clay-dominated JA-W profile. Consequently, leaching of salts is expected to be much more efficient in the JA-E site. As a result of the probable low permeability at the JA-W site, much of the applied irrigation water is expected to run off towards adjacent, sandy loam profiles. The inflection point at about 0.4 m in Fig. 3.6a for the JA-W soil suggests that flow into the clay profile is only able to advance at about 0.02 m/year.

Water-extracted profiles of B in the JA fields are presented in Fig. 3.6b. (Phosphate extractions of B from the JA-E profile were not significantly greater than the water-extracted B.) A strong correlation between extracted B and clay content was found. This correlation is illustrated in Fig. 3.7a for the JA-W soils. Note that in this figure, the only

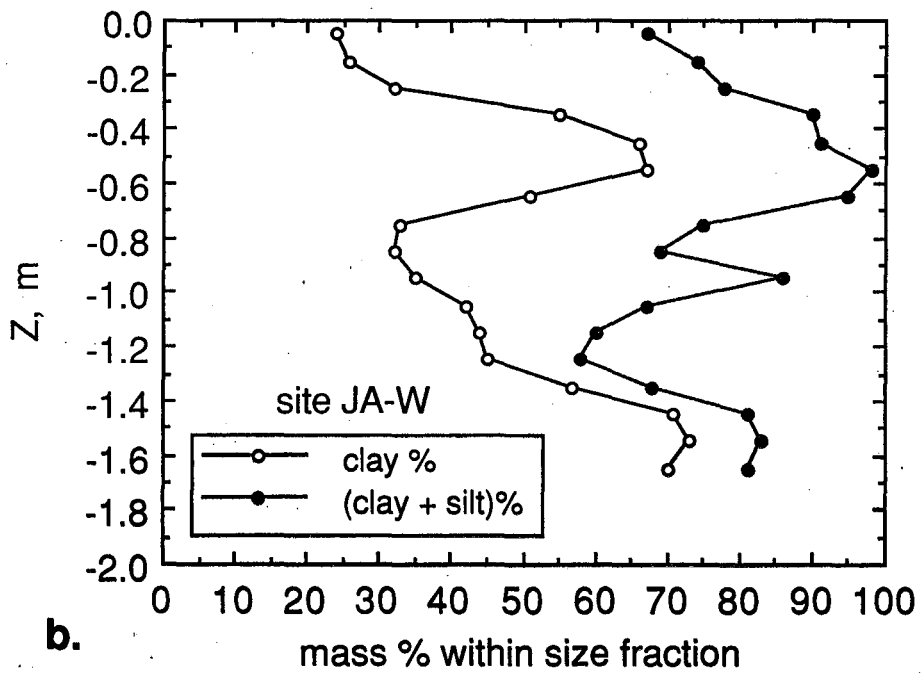
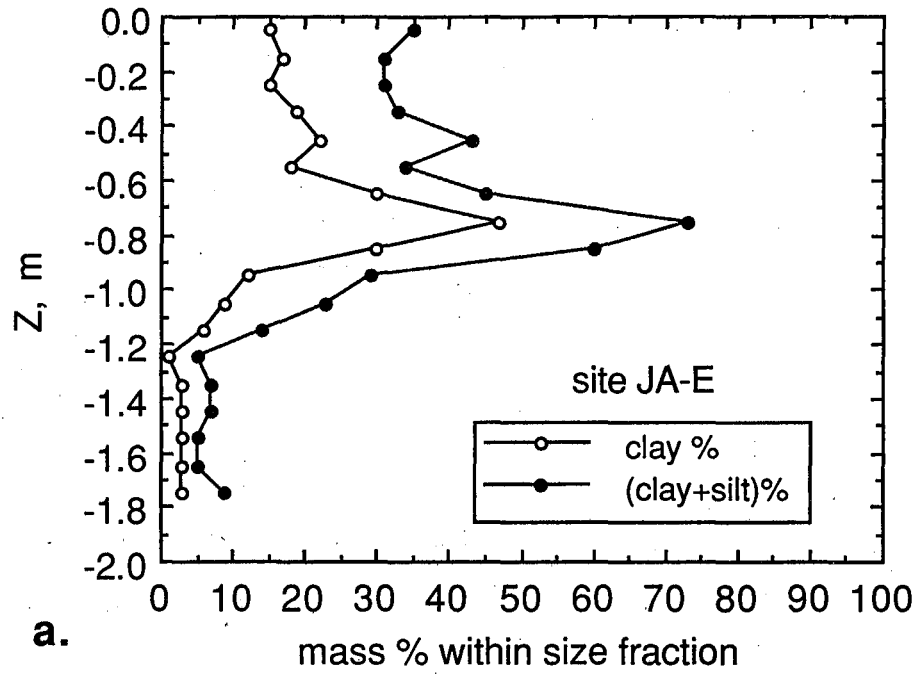


Figure 3.5. Clay and clay+silt profiles in the JA field, (a) site JA-E and (b) site JA-W.

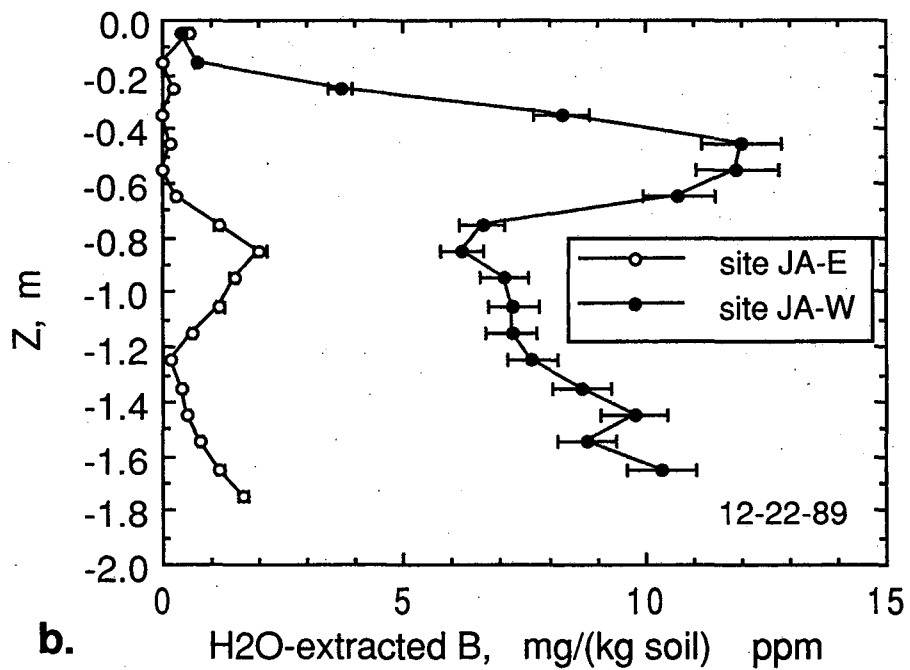
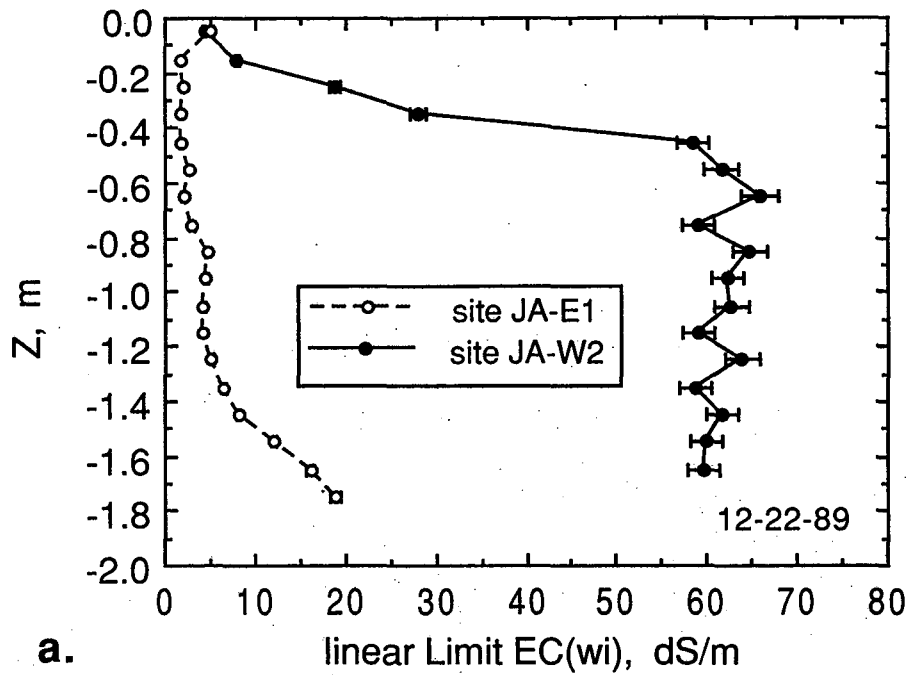
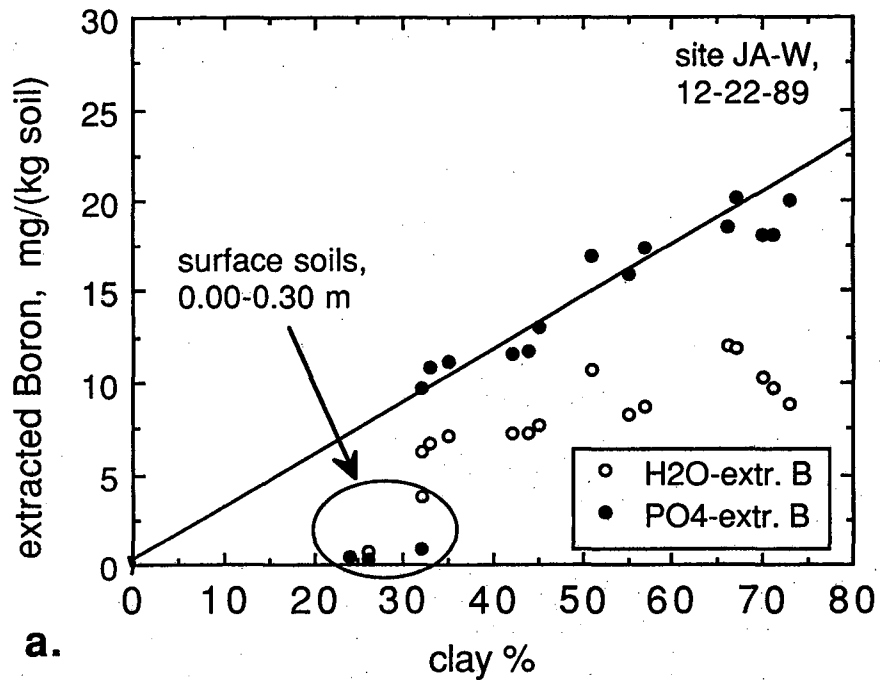


Figure 3.6. (a) Electrical conductivity (EC) profiles in JA-E and JA-W sites from water-extract data linearly normalized back to initial water contents. (b) Water-extracted boron profiles from the JA-E and JA-W sites.

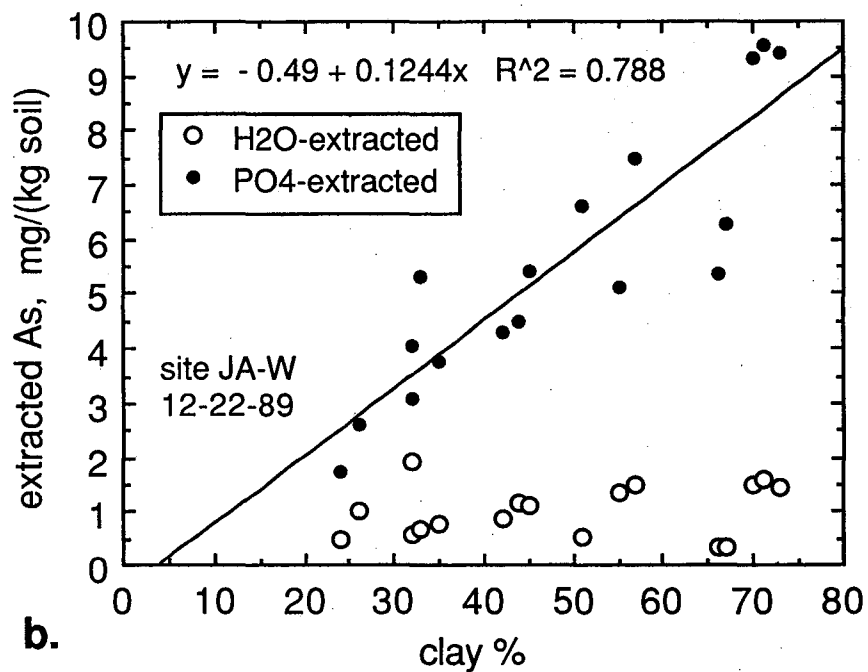
deviations from the linear regression forced through the origin are from the leached surface soil zone. (This is apparent from comparing Figs. 3.6a and b with Fig. 3.7a.) The strong association of B with clays has been well documented in the literature (e.g. Hingston, 1964; Sims and Bingham, 1968; Adriano, 1986).

Water-extracted As concentrations from the two soil profiles are shown in Figs. 3.8a and b. Phosphate-extracted As profiles are also shown. The differences between phosphate- and water-extracted As concentrations are suggestive of larger inventories of adsorbed and marginally soluble As. It is interesting to note that without the information on clay distributions, the arsenic concentration profile data alone would be suggestive of pulses of arsenic-enriched soil waters which have traversed different distances into the JA soils. In view of the particle-size information, such an interpretation is probably incorrect in these cases. Noting the similarity in the general shapes of the clay profiles (Figs. 3.5a and b) and the As profiles (Figs. 3.8a and b), one can deduce that the water-extracted As profiles are largely a result of As associations with clay and colloid surfaces discussed previously with respect to other FIR soils. Analyses of the extracted As from these and other FIR and SWMA soils using the method of Glaubig and Goldberg (1988) indicated that arsenate, As(V), accounts for most of the extracted As. Correlations between clay content and extracted As are presented in Fig. 3.7b. From the apparent strong association between As and clays, low rates of As displacement are expected from the JA profiles into drain waters. This does not rule out the possibility that much As has been leached out of the JA-E profile in the past. This latter scenario seems unlikely in the JA-W profile due to its persistent high salinity.

Water-extracted soil Se in the two JA sites are shown in Fig. 3.9a. The extracted Se was primarily Se(VI). Little more Se(VI) and almost no Se(IV) was recovered with phosphate extractions. In the case of the JA-E soils, no significant differences between the two extractions were detected. This was at least partially due to the larger relative uncertainties encountered at the very low measured Se concentrations in the JA-E profile.



a.



b.

Figure 3.7. (a) Correlations between (water- and phosphate-) extracted boron and clay content in the JA-W site. Note the locations of points in the leached surface soils. (b) Correlations between (water- and phosphate-) extracted arsenic and clay content in the JA-W site.

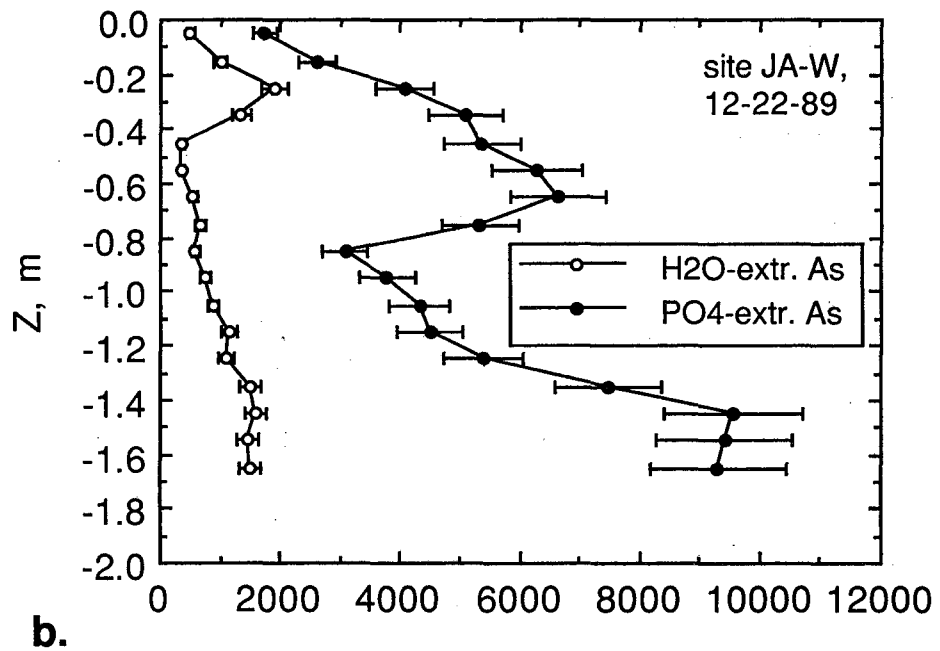
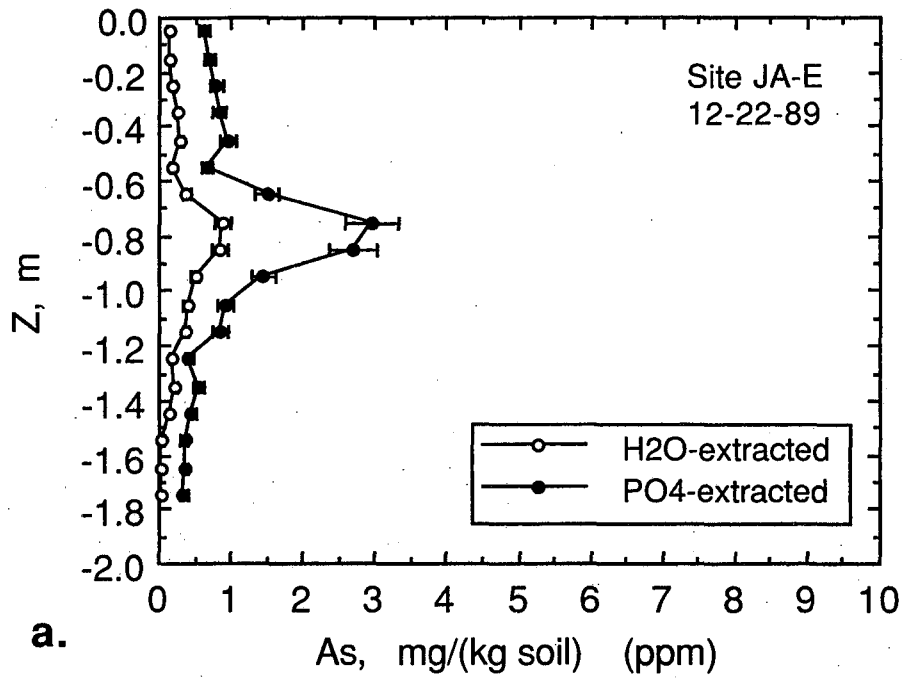


Figure 3.8. Water- and phosphate-extracted arsenic in the (a) JA-E site and (b) in the JA-W site.

The large differences in Se concentrations between the two profiles appear to be analogous to those of the EC profiles shown previously. In both cases, thorough leaching of the well-drained JA-E profile and impeded leaching of the JA-W profile can explain the low TDS and low soluble Se of the former soils and the high TDS and high soluble Se found in the latter profile. An important principle is illustrated through these two profiles. High concentrations alone, whether for trace elements or for TDS, do not result in high loading of drain waters. Rather, it is the product of concentrations and flow rates that govern drainwater loading from leaching soils.

Both water and phosphate-extracted Mo from the JA sites are presented in Fig. 3.9b. When compared with other extracted Mo concentrations from many other FIR sites, the extractable Mo associated with local concentration maxima shown here are relatively high. While the extractable Mo profiles in the JA-E site correlate fairly well with clay content, the relation does not hold in the JA-W site.

As previously noted, water-extractions and phosphate-extractions of soils often yield relatively small fractions of the total soil concentrations of various trace elements of interest. In Figs. 3.10a and b, comparisons of water-extractions, phosphate-extractions, and total element analyses for As and Se in the JA-W profile are presented. Water-extracted As in this profile amounts to only about 5% of the total soil As. Phosphate-extracted As amounts to only about 20% of the total soil As at this site. While rough correlations between water-extracted, phosphate-extracted, and total soil As were obtained at this site, generalization of these correlations to other sites can not be made without first inspecting a larger data base which includes these various analyses on other soils. From the data presented in Figure 3.10a, it is clear that total soil As concentrations are much higher than the soil As inventory immediately available for uptake in the food chain.

The data on water-extracted, phosphate-extracted, and total soil Se from JA-W shown in Figure 3.10b further illustrate the complexities involved in attempting to correlate concentrations obtained in various procedures. In this profile, water-extracted and phosphate-extracted Se concentrations were essentially identical. As noted earlier, such results are

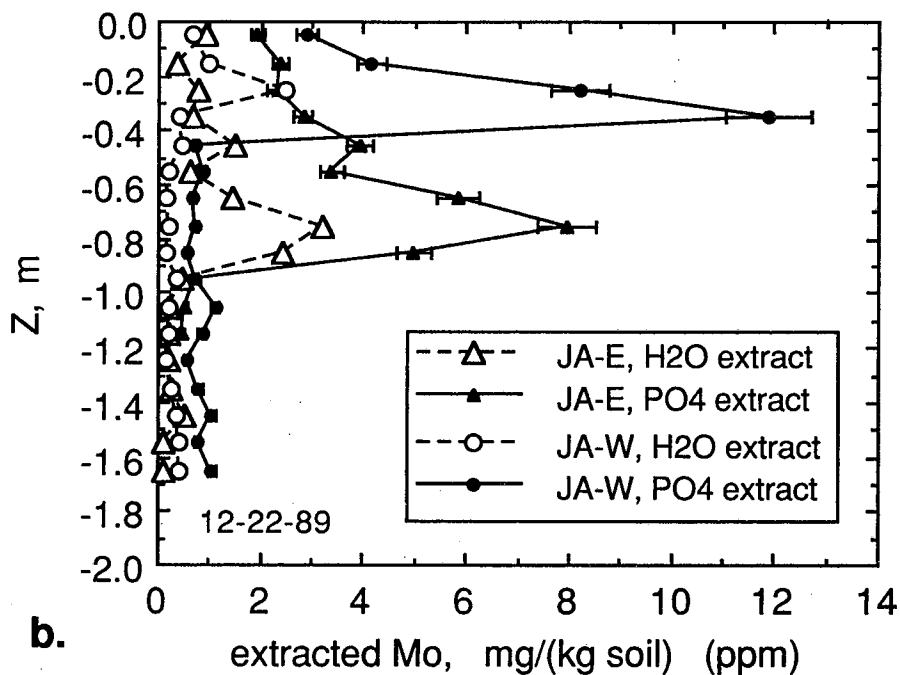
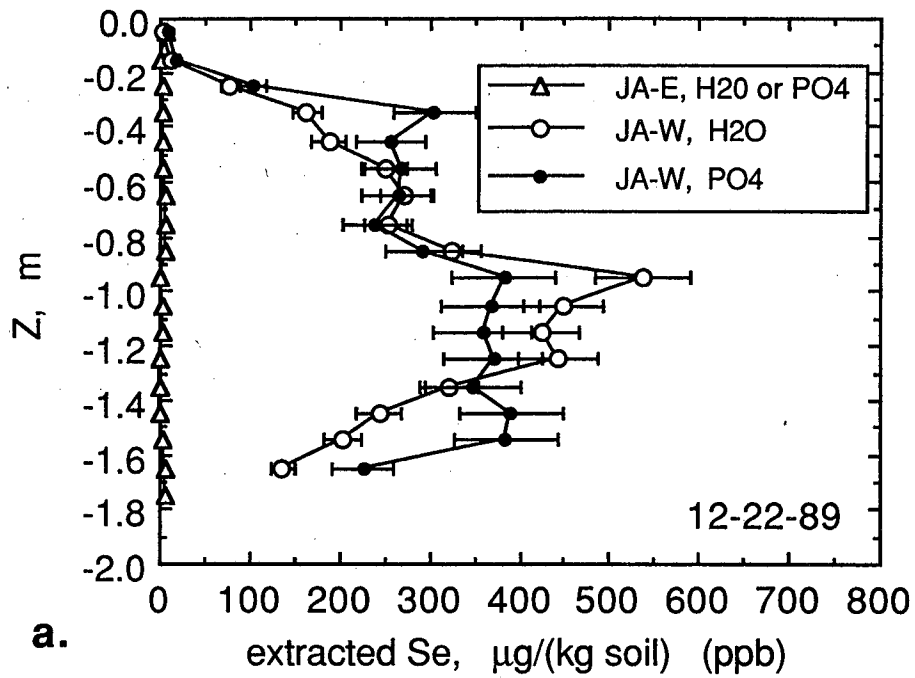


Figure 3.9. (a) Water- and phosphate-extracted selenium in the JA-E and JA-W sites. No significant differences between extractions were detected in the JA-E site. (b) Water- and phosphate-extracted molybdenum in the JA-E and JA-W sites.

reasonable when little readily available selenite is present. These extractions illustrate a case where total Se is not correlated with soluble Se. Note that the soluble Se distribution exhibits a maximum in the central portion of the profile while the total Se profile displays nearly the opposite pattern. A possible explanation for the lack of correlation between water-extracted and total Se at the bottom of the profile involves postulating a reducing environment in this region. In such case, selenate concentrations are expected to diminish via reduction to selenite (with possible further reduction to elemental Se and selenides). This possibility is consistent with the observed higher Se concentrations obtained in the phosphate-extracts relative to the water-extracts in the 1.40 to 1.70 m depths (Figs. 3.9a and 3.10b). However, this explanation is speculative because the redox potential was not measured. The lack of correlation between water-extracted and total Se in the aerated near surface zone can not be explained by the equilibrium redox behavior of Se. The poor performance of total element analyses for trace elements in predicting readily available trace element inventories in soils is discussed further in Chapter 5.

3.4. Laboratory Column Leaching Studies of FIR Soils

Determining the extent to which poor drain water quality is a result of salts and trace elements leached from agricultural fields versus from deeper groundwater sources is an important prerequisite for evaluating the feasibility of any drainage alternatives. Much of the information needed in this process is obtained from field characterization of the soils and groundwater and laboratory analyses of field-collected soil and water samples. The information obtained from water-extracts of soil samples provides a somewhat static assessment of the soil. Little information is gained with respect to rates at which various trace elements and salts will move through a soil profile and into the groundwater and drain systems. In order to fill this gap, a set of soil column leaching experiments have been initiated. The combined data from soil extracts and column leaching studies provide the information needed to determine leachable salt and trace element inventories in FIR soils. Such information furthermore is valuable in estimating rates of reclamation of the FIR soils presently

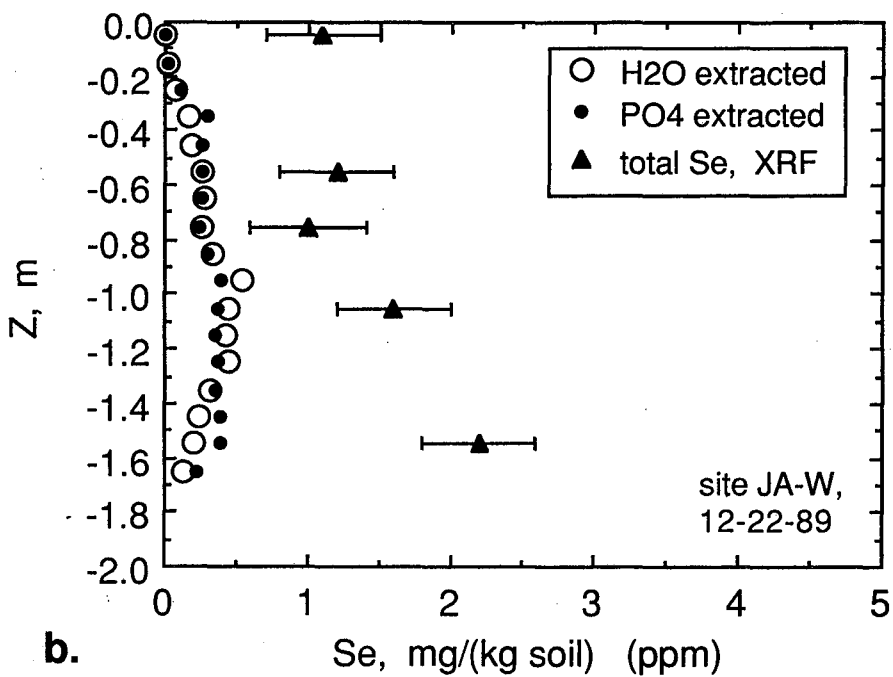
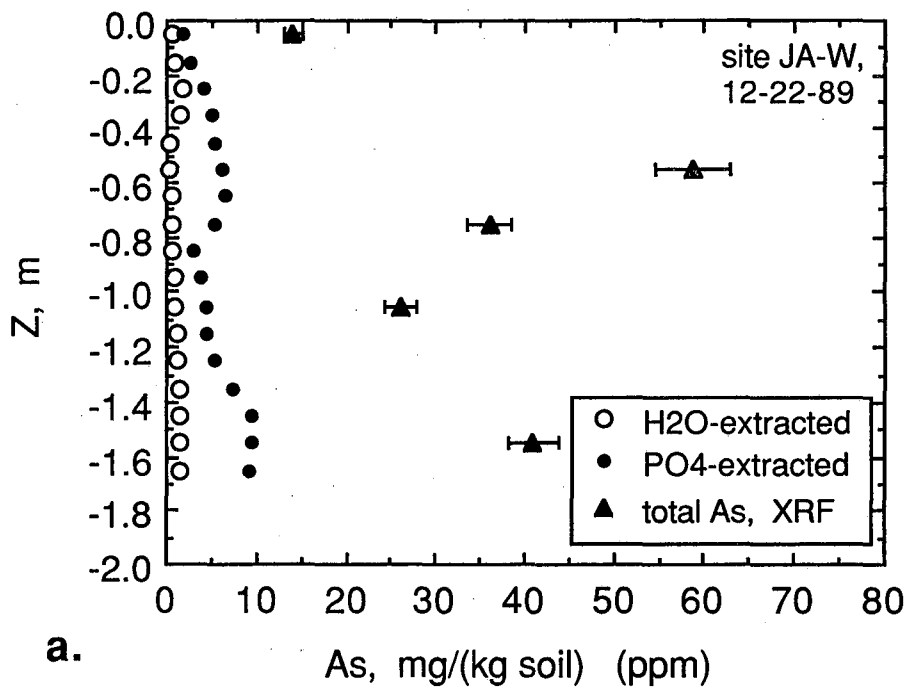


Figure 3.10. Site JA-W; (a) Water-extracted, phosphate-extracted and total arsenic and (b) water-extracted, phosphate-extracted and total selenium.

served by the TJ Drain. In this section, the general features of the experiments will be very briefly described and results from one set of such experiments are presented.

3.4.1. Experimental Design

Two types of column leaching experiments were designed to measure leach rates from FIR soils. One set of experiments is performed on short columns of repacked soils from particular depth intervals. A controlled rate of surface water application is established and leachate from the columns is periodically collected for chemical analyses. In another set of experiments, longer soil columns are employed in which the field soils are repacked into the columns according to their original stratification. Again, irrigation waters are applied and the effluent is periodically sampled. The short columns provide detailed local information, while the long columns provide leachate which will more closely simulate actual drainage waters. Information concerning local distributions of salts and various trace elements will be obtained from samples collected through ports along the length of the column. Effluent water quality data will provide an understanding of rates of leaching under various controlled conditions.

In both types of column experiments, soils from undeveloped lands in Fallon II were used since such soils would provide initially high inventories of the solutes of concern. In particular, soils from two sites neighboring the irrigated F2 fields were used. A larger diameter bucket auger was used for sample collections in anticipation of requiring large volumes of soil for replicate columns. These soils were sampled at 0.10 m intervals only within the uppermost 0.40 m, after which 0.20 m sampling intervals were used. Soils from site F2-OS1 were collected on May 9, 1990, in an area covered with low-relief dunes stabilized by native shrubs. The site is located 30 m west of the western edge of the irrigated portion of Fallon II and about 400 m south of the FIR-SWMA boundary. The soil was predominantly of fine sand texture, with a sandy clay loam horizon between depths of 0.10 and 0.40 m and clay interbedded with fine sands over the depths of 2.0 to 2.6 m. Weak mottling was observed at all depths below 0.5 m, with a zone of strongly mottled

fine sands encountered at depths ranging from 1.0 down to 1.5 m. The water table was reached at a depth of 2.5 m.

The second sampling site, designated F2-0, is located in a previously levelled but never irrigated portion of Fallon II, about 60 m south of the westward flowing irrigation ditch which is periodically used to irrigate the Fallon II fields to the north. The F2-0 soils were sampled on May 11, 1990. This soil was of generally finer texture than that obtained at the F2-OS1 site. The surface soils were sandy loams, grading to fine sands at a depth of about 0.30 m. A mottled clay horizon was encountered from about 0.8 m down to 1.8 m. Sandy loam to fine sand textures were characteristic of the remaining sampling intervals. The water table was reached at a depth of 2.85 m. Samples from depths greater than 3.0 m could not be retained. Carbonates were detected throughout much of both profiles.

Individual sampling intervals were prehomogenized prior to being packed into the columns. In view of the fairly deep sampling lengths, the long column soil packing was scaled down by 50%: the 0.20 m interval soil samples were packed into 0.10 m column sections, with the inside diameter equal to 37 mm. Six columns from each site will be used to provide duplicates of three different infiltration rates. Presently, only half of the long columns have been completely assembled and filled.

The short column experiments are used to measure effluent quality generated from particular depth intervals. To date, only a preliminary phase of these experiments has been completed. This first experiment was directed towards measuring changes in drainage water quality in three separate depth intervals from the F2-OS1 site and comparing the cumulative leached inventories of individual trace elements and ions with inventories estimated from soil-water extracts performed on the same materials. Such comparisons will assist in the interpretation of soil-water extraction data for these soils.

For this purpose, the 0.40–0.60 m, 0.80–1.00 m and 1.20–1.40 m intervals of the F2-OS1 samples were used. Duplicate, “full-scale” (i.e. 0.20 m long) columns were prepared. Column bottoms were lined with filter paper backed with stainless steel screen and fit with 3 mm inside diameter tubes through which effluents were collected. Soils were

maintained at their field moisture content at all stages prior to the application of irrigation waters. The 0.40–0.60 m samples were packed to an equivalent dry bulk density of 1380 kg/m³. The 0.80–1.00 and 1.20–1.40 m samples were both packed to achieve equivalent dry bulk densities of 1490 kg/m³.

For leaching the columns, a simulated S-Line Canal water was prepared, which approximately matched an average Newlands project water composition, without the inclusion of trace elements. Newlands project water quality data were obtained from Table B-1 of the USBR (Sept. 1987) FIR Water Quality Report. Water was applied separately to the top of each column via peristaltic pumps, at a rate of 0.10 m/hour. Water was supplied until at least 8 pore volumes of effluent was collected from each column. (A “pore volume” is equal to the volume of water contained in the soil column during the leaching experiment and provides a useful “common denominator” for comparisons between similar leaching experiments. In this experiment, the pore volume does not refer to the total volume of pores in the soil. The columns had between 12% and 18% of their pore space occupied by trapped air (determined gravimetrically), therefore the pore volumes refer to volumes ranging from 81% to 88% of a potentially available pore space.)

3.4.2. Results

Presently, only the effluent EC, B and Se concentration data have been processed. Analyses have also been performed for major ions, arsenic, silica, lithium and molybdenum. The processed data are presented in two forms. First, the directly measured effluent concentration is reported as a function of the (cumulative) number of pore volumes leached at the time of collection. A second method of presenting the data is in terms of the cumulative quantity of solute removed from the column.

In the case of a particular solute being released by the soil, the product of the concentration times the volume of the effluent sample is equal to the mass of the solute in the particular sampling of the effluent. Summing such products (for a particular solute) over all the samples collected provides a measure of the cumulative mass of the solute released from

the soil. One final normalization is commonly performed by dividing the cumulative mass of the solute collected in the effluent stream by the mass of the soil sample being leached. This type of normalization again provides a means for comparing different experimental data sets. In the present work, such normalization will be compared with “water-extractable” concentrations obtained by equilibrating soil/water mixtures (as described in Appendix B).

Effluent EC values from duplicate columns from each of the three depths are presented in Fig. 3.11a and b. The second (semi-log) figure is provided to illustrate the measurable deviations of EC values from that of the inflowing waters. The rapid declines in effluent EC indicate that the readily soluble salts are rapidly leached. A significant slow release of salts is still detectable, especially in the 0.40–0.60 m soils. The slow release of additional salts in these columns appear to be a result of slow dissolution of gypsum and calcite. Nevertheless, by the time that 3 pore-volumes have passed through the soils, ECs are less than 1.5 dS/m, indicating that total dissolved solids in the effluent are less than 1000 mg/L.

Concentrations of B in the column effluents are plotted in Fig. 3.12a. Note that B concentrations in the leachate diminish very rapidly and declining below 1 mg/L by the time one pore volume of effluent water has been collected. It should be noted that such rapid declines in effluent B are partially a result of the relatively high leaching rate. With slower percolation, less soluble forms of B have sufficient time to dissolve into the percolating water resulting in higher effluent B concentrations. Experiments designed to determine the magnitude of this effect in FIR and SWMA soils are planned. The cumulative amount of B released in the three soils is plotted in Fig. 3.12b. On the right side of the graph, horizontal lines indicating the cumulative B quantities predicted from 1:5 soil-water extracts are shown. The comparisons suggest a strong influence of B resorption from solution, which is minimized in leaching processes.

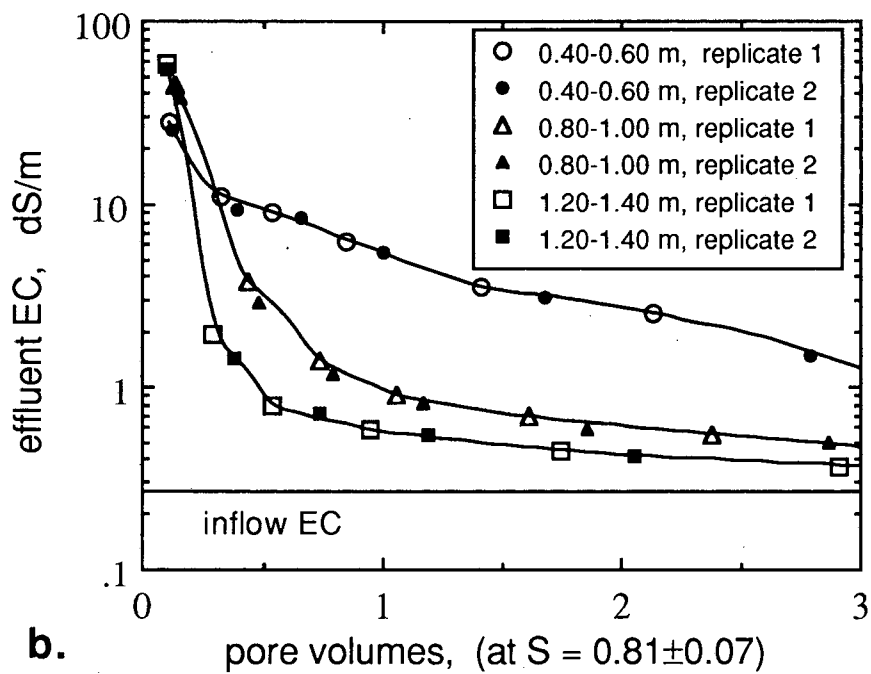
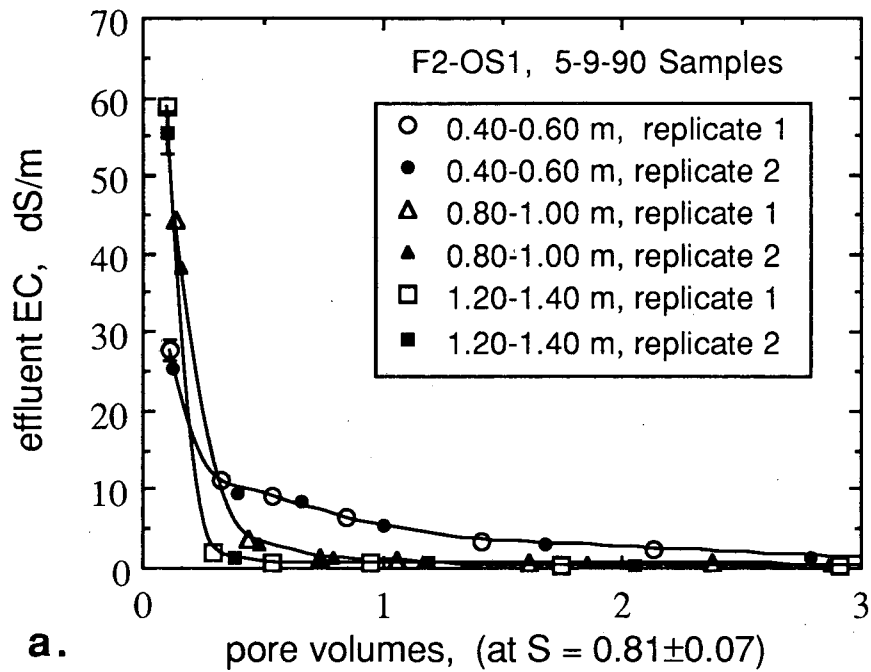


Figure 3.11. (a) Electrical conductivity (EC) of F2 soil column leachate versus pore volumes leached, (b) same data on semi-log scale.

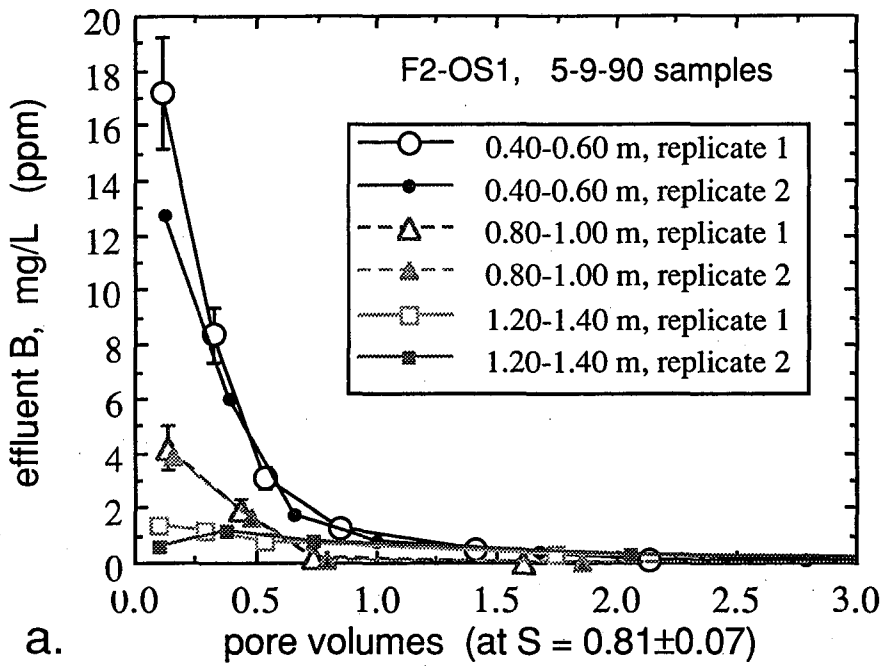
The data on Se concentrations in the leachate are presented in Fig. 3.13. Of these soils, only the 0.40–0.60 m interval had enough soluble Se to detect with confidence. This limited inventory was readily leached from the column.

The column leaching data are quite limited at this stage. However, they do demonstrate that with respect to salinity, well drain profiles can be relatively quickly desalinized. A continued slow release of B is also expected, both from the evidence provided in the literature and from the discrepancies between B extracted in batch equilibrations and B extracted by continuous leaching.

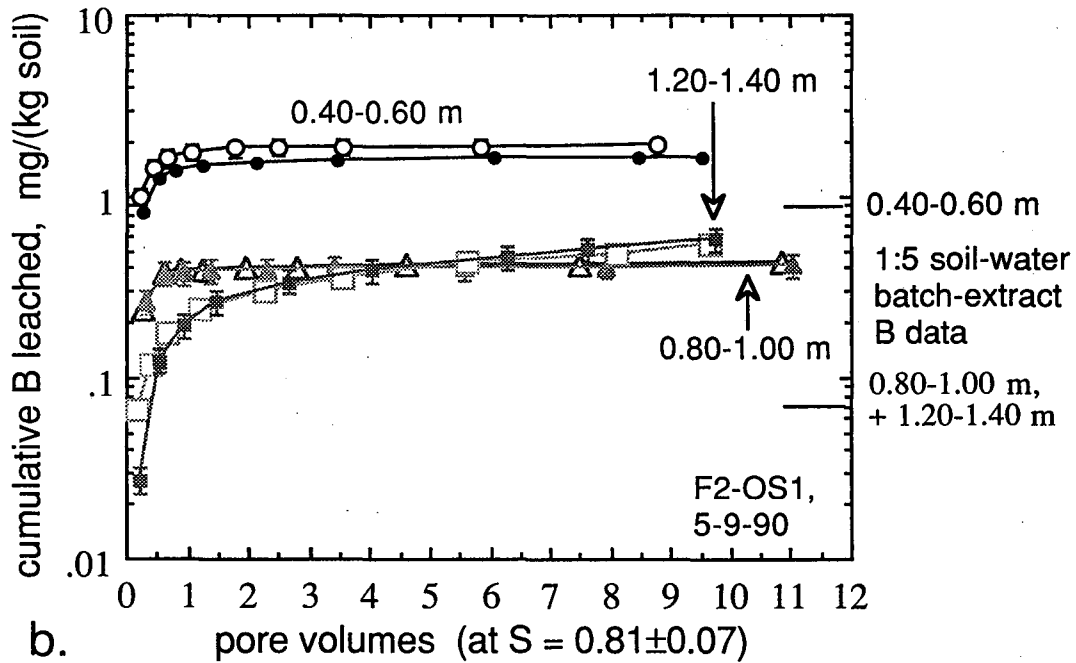
3.5. Summary of FIR Soil Profile Salinity and Select Trace Element Data

In the opening sections of this chapter, the pervasiveness of high salinity and high concentrations of trace elements in undeveloped Lahontan Valley soils was recognized. In the third section, distributions of salts and certain trace elements in different irrigated fields within the FIR were described. Within each of the different age fields, considerable spatial variability with respect to salinities and trace element inventories was recognized. Nevertheless, comparisons of data from different fields were suggestive of reasonable time trends resulting from leaching salts and some trace elements from soil profiles. The laboratory soil column data presented in the previous section demonstrated the expected improvements in leachate water quality over time under controlled conditions. An overview of all of the analyzed FIR soil profile data, in conjunction with the laboratory column data provides an opportunity to summarize the general patterns in salt and trace element leaching from area soils. From this overview, a better understanding of the nature of the agricultural drainage water contribution to TJ Drain waters can be developed.

In order to facilitate the assessment of possible time trends in soil profile characteristics resulting from irrigation, it is sometimes useful to consider profile-averaged quantities. It is clear from the detailed soil profile data presented in Section 3.3 that aside from the site to site variability of numerous properties, considerable variability in chemical composition is also observed with depth at most sites. Profile-averaging of a particular chemical prop-



a.



b.

Figure 3.12. F2 soil column leachate; (a) Boron concentration in instantaneous outflow and (b) cumulatively leached boron.

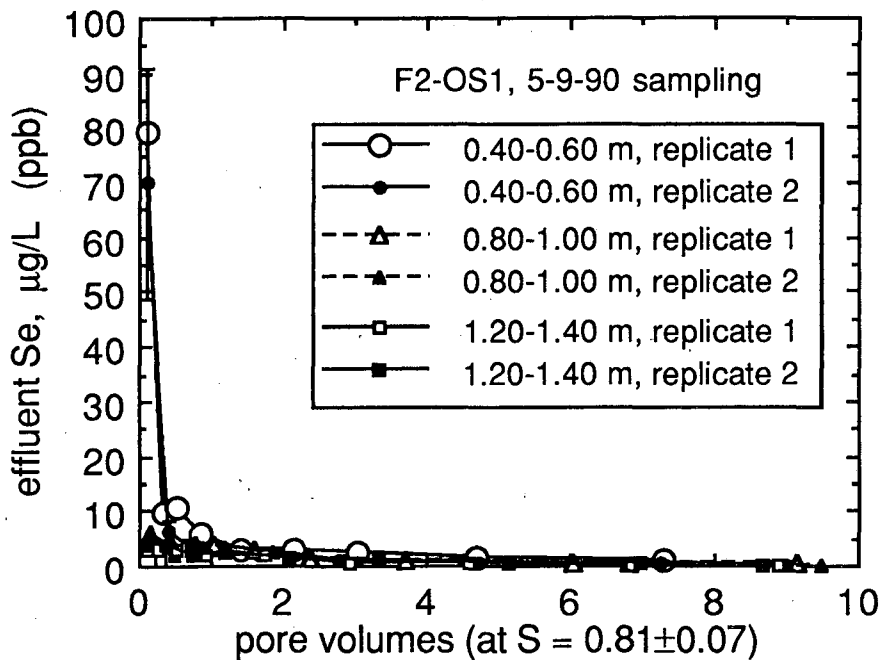


Figure 3.13. F2 soil column leachate selenium concentrations.

erty refers here to calculating the expected chemical concentration or EC for the case where component depth intervals in a profile are mixed to achieve a homogeneous profile. Thus site to site comparisons of any particular property involve comparisons of only one characteristic value from each site. The statistical significance of any apparent trends can then be easily tested.

For purposes of the present discussion, average properties of the upper 1.70 m depth intervals were calculated for each of 10 sites. In one additional site, IB-s3, only the upper 1.50 m was sampled, hence the averaging was performed down to this interval. While most sites have been characterized to depths of about 3.0 m, the 1.50 to 1.70-m averaging depths provided a uniform framework with which to make meaningful comparisons among all profiles. More importantly, this depth interval corresponds more closely to the drained soil profile depth needed to sustain crop production. Profile-averaged values of various properties were obtained simply by summing values from individual 0.10 m depth inter-

vals, then dividing by the total number of intervals (17, except for site IB-s3 in which the sum is divided by 15). The validity of this procedure requires that variations in soil bulk density with depth be small relative to depth variations in the properties of interest. For the profiles considered here, this procedure appears to be reasonable. The profiles to be compared include 2 unirrigated sites (F2-0 and F2-OS1), 4 sites with 1 year of irrigation (F2-s1, F2-s2, F2-s4, and F2-s7), 3 sites with 5 years of irrigation (IB-E-s1, IB-s2, and IB-s3), and 2 sites with about 20 years of irrigation (JA-E and JA-W).

Apparent time trends in profile-averaged properties were considered. In these comparisons, profile-averages of EC, B, As, and Se obtained in water-extracts of soils were plotted against the number of years that the particular sites have been irrigated. It is emphasized that these data lead to inferred time trends rather than actual time trends since the data come from fields with different irrigation histories, rather than from time series data from a particular site. As noted in Section 3.3, the JA-W site was unique in many respects because of its high clay content, low permeability, and consequent impeded leaching of salts and trace elements. Thus while the JA-W site is within a field irrigated for at least 20 years, the depth of leaching at this site is more characteristic of a site irrigated for no more than 1 season. For this reason, the JA-W site was left out of all correlation analyses.

Profile-averaged soil salinity is graphed as a function of the number of years of site irrigation in Fig. 3.14a. The salinity is expressed as the average of linear limit EC data described previously. The EC versus time data from all sites except site JA-W were fit to an exponential decay function as shown in the figure. (The data from site JA-W are also included in this and similar figures for comparison.) The exponential function obtained in the least squares fit, along with the correlation coefficient are included in the figure. The exponential function was selected since it appears to be a reasonable choice among the simplest functions, and provides a simple test on the significance of possible correlations. The exact expressions for effluent composition-time functions for even the simplest of soil leaching processes are much more complex (see for example, van Genuchten and Wierenga, 1986). It is evident from Fig. 3.14a that considerable variability in profile-

averaged EC is found within any given field. Nevertheless, for the 10 profiles used in generating the regression equation, the correlation coefficient $r = 0.79$ supports the hypothesis that the data are significantly correlated at the 1% confidence level. The fact that a significant correlation is obtained despite the clearly large EC variations within individual fields lends support to the use of data from a collection of different fields for deducing time trends. The denominator in the exponential provides an estimate of the number of years of irrigation needed to decrease the profile-average salinity to 37% of its initial value. In this case, the regression equation suggests that about 6.6 years will be required to reach this stage. The exponential regression equation can also be interpreted in terms of the time required to diminish a profile-average salinity to 50% of its current level. From this perspective, the average soil salinity is diminished to half of its current level every 5 years. The exponential function is expected to slightly over-estimate the time required to reach this stage of leaching since it best fits systems in which resident and inflow fluids mix completely while leaching proceeds.

The effluent EC data from the column leaching experiments provides further evidence that the exponential curve-fit may over-estimate leaching times. Note in Fig. 3.11b that the column drainage water EC curves are all concave upwards in the semi-log plot, whereas exponentially decreasing functions would plot as straight lines. Thus an exponential fit to the data would over-estimate EC values at intermediate times (and under-estimate initial and long-time EC data).

The simple least-squares fit leads to further over-estimates of leaching times because the individual profile averages are assumed to be normally distributed about the mean value of a large population of profile-averages. Probability distributions of soil salinity and other chemical characteristics tend to have mean values skewed towards lower values (for example, log-normal distributions). In such case, average leaching times will again be somewhat shorter than indicated in the curve-fit shown in Fig. 3.14a.

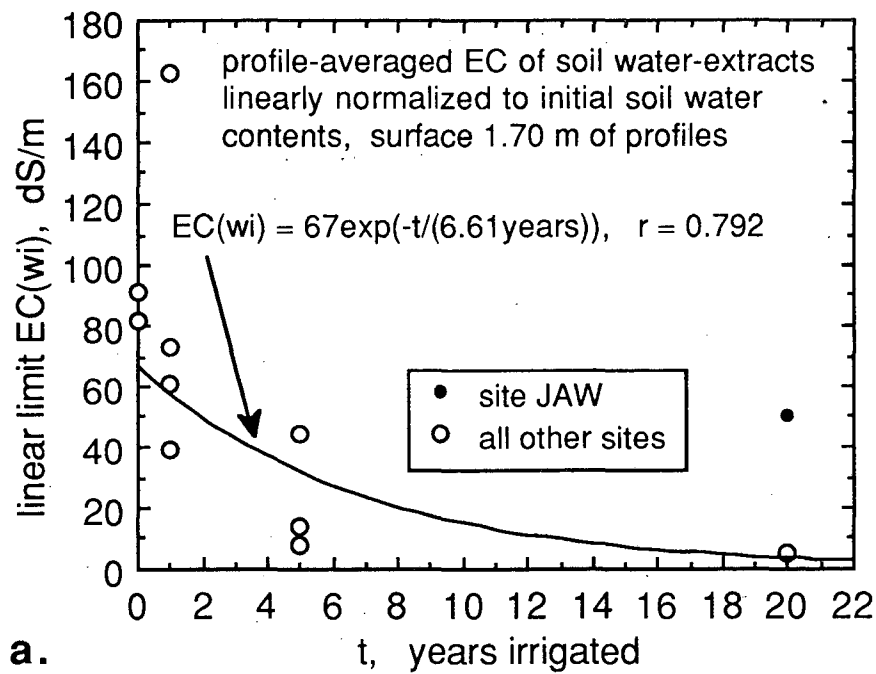
The correlation analyses for profile-average B in the various fields is shown in Fig. 3.14b. The much greater scatter in the data is reflected in the low and insignificant correla-

tion coefficient. This result is probably explained by the high correlation between extracted B and clay content noted in Section 3.3. The high affinity of B to colloid surfaces and solid phases hinders its leaching out of the soil profiles. Thus, the large inventory of B in soils throughout the Lahontan Valley can only be slowly depleted in leaching environments. On the other hand, such slow releases also imply that B concentrations in soil waters of older irrigated fields may be relatively low. The column experiments presented in the previous section support this conclusion. After about 1 pore volume of drainage from the columns (equivalent to about 5 years of drainage under irrigation), B concentrations in the column leachate are less than or equal to 1 mg/L. Field soil solution sampling as part of the upcoming shallow tile drain study will provide further information in this area.

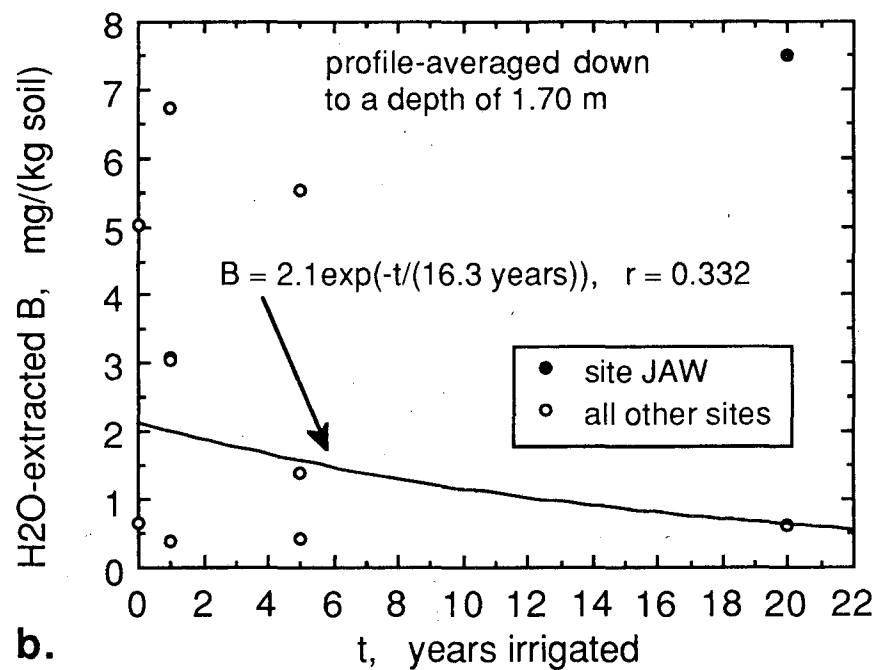
The correlation analyses for water-extracted As in the FIR fields is presented in Fig. 3.15a. The high variability and poor time-correlation ($r = 0.07$) of profile-averaged As is similar to the case of B discussed previously. Again, the influence of high As affinity to surfaces and solid phases rather than simple leaching dominates. The profile-averaged water-extracted As concentrations are not significantly correlated with the number of years the soils have been irrigated.

The correlation analysis for water-extracted Se in the various FIR fields is presented in Fig. 3.15b. A weak correlation ($r = 0.58$) is suggested. However, it should be noted that the low Se concentrations in the extracts introduce additional uncertainties since as detection limits are approached. (Some of the Se data from individual depth intervals are less than $0.2 \mu\text{g/L}$ as analyzed.)

Other analyses of profile-averaged properties have also been done. Possible correlations between profile-averaged salinity and profile-averaged extracted B, As, and Se have been tested. Results of these tests have shown that water-extracted B and As are at best only weakly correlated to salinity. The water-extracted Se data appeared correlated with salinity (at the 5% confidence level), yet the previously cited uncertainties in the low Se concentrations diminish the significance of this relation.



a.



b.

Figure 3.14. (a) Profile-averaged soil EC plotted versus the number of years the sites have been irrigated. See text for details on sites and averaging. (b) Profile-averaged water-extracted boron plotted versus years of irrigation.

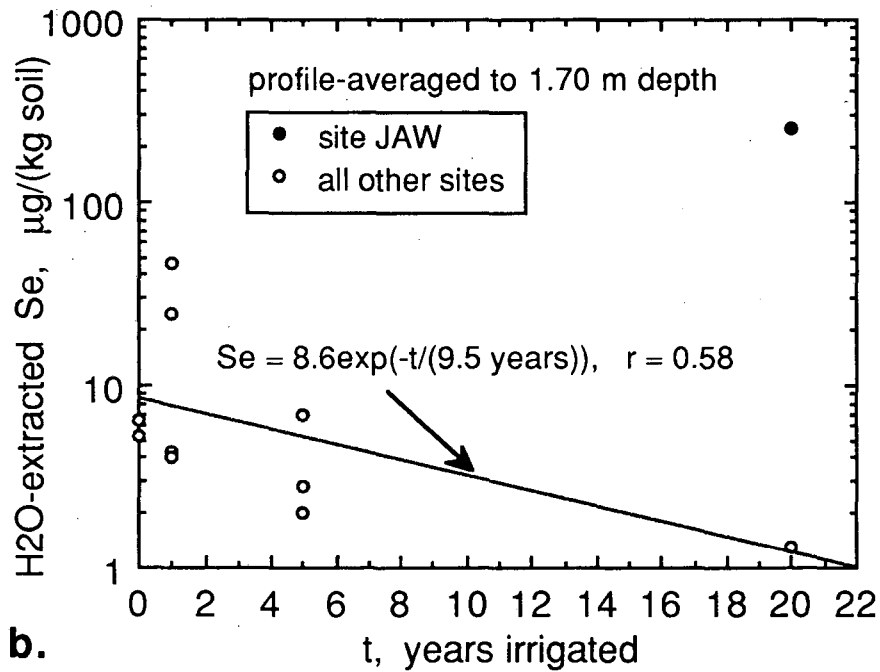
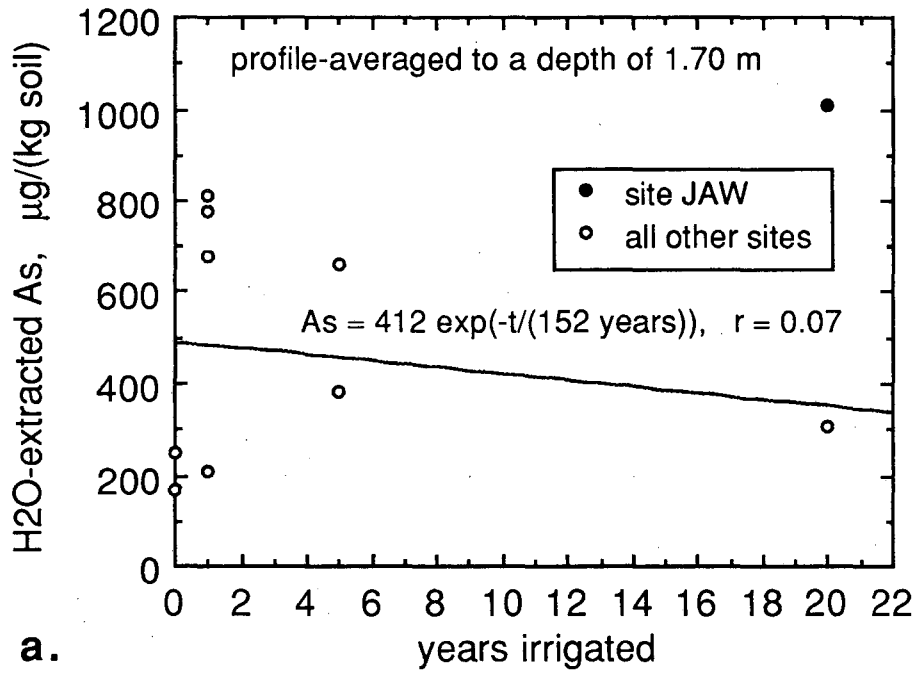


Figure 3.15. (a) Profile-averaged soil water-extracted arsenic plotted versus the number of years the sites have been irrigated. See text for details on sites and averaging. (b) Profile-averaged water-extracted selenium plotted versus years of irrigation.

4. GROUNDWATER INVESTIGATIONS AT FALLON INDIAN RESERVATION

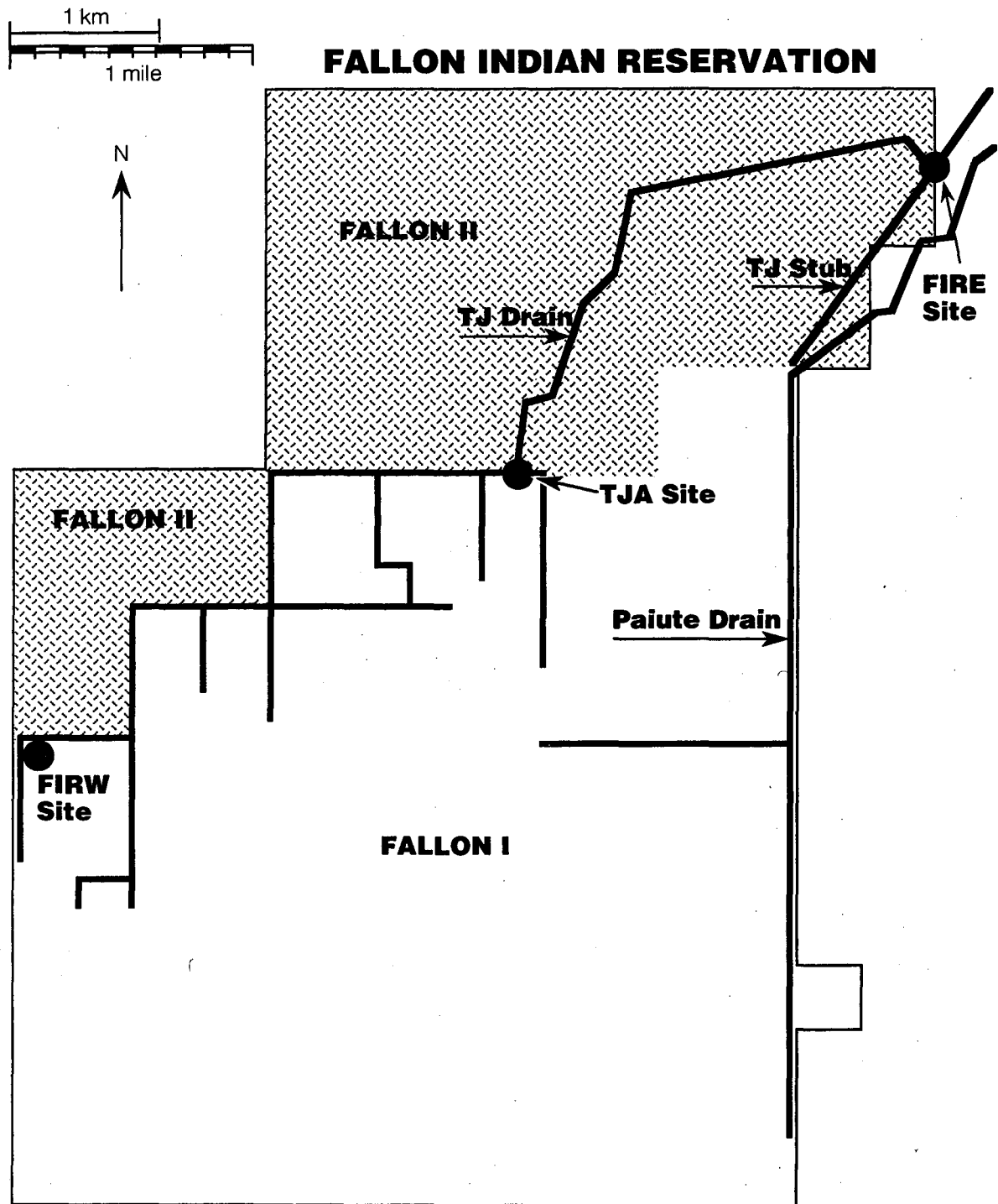
4.1. Site Investigations

As part of the LBL effort a detailed investigation of the groundwater system underlying the FIR was undertaken. Objectives of the investigation were two-fold. First, since shallow groundwater under the FIR is the major source of salts and trace elements to the TJ Drain, this effort was designed to provide additional information on the extent and chemical characteristics of the saline groundwater discharging into the TJ Drain. Second, the shallow aquifers under the Reservation may be suitable for reinjection of TJ drain waters; the program was tailored to address this question.

The basic information required to accomplish the objectives of the investigation include the following:

- Information on the geological characteristics of the subsurface,
- Information on the structure of the aquifers and aquitards,
- Information about the hydrologic properties of the aquifers and aquitards, including hydraulic conductivities and storage coefficients of the hydrostratigraphic units,
- Information about fluid potential (water level) in the hydrostratigraphic units and
- Information about salt and trace element distributions in the groundwater system.

To gather this information, clusters of wells were installed for testing and evaluation at three sites in FIR. Fig. 4.1 shows the locations of these three sites. Together the sites provide for a 7.5 km (4.5 miles) SW-NE trending cross-section of the area under the FIR. Twelve wells were drilled and tested, with a minimum of three wells at each site. The maximum depth penetrated at each site was about 100 m (330 ft).



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Figure 4.1. Location map showing the sites, FIRW, TJA and FIRE, where clusters of wells were installed for the groundwater investigations.

More complete descriptions of the drilling and testing program are provided in the following paragraphs.

4.1.1. Drilling Program

Of the 12 wells drilled during this program, 4 were located at the FIRW site, 5 at the TJA site and 3 at the FIRE site. Clusters of wells, each penetrating to different depths below the ground surface, were installed to obtain depth-profiles of water quality, fluid potentials (water levels) and hydrogeologic characteristics at each site. Cross-sectional and plan views of the wells at each site are shown in Figs. 4.2 to 4.4. Table 4.1 summarizes information on well depths and screened intervals.

Table 4.1. Depths and screened intervals for wells drilled as part of the LBL groundwater investigation at the FIR.

| Well Name | Depth (m) | Length of Screen (m) | Top of Screen (m) | Bottom of Screen (m) |
|-----------|-----------|----------------------|-------------------|----------------------|
| FIRW-33A | 36.6 | 15.2 | 18.3 | 33.5 |
| FIRW-33B | 36.6 | 15.2 | 18.3 | 33.5 |
| FIRW-76 | 79.2 | 21.3 | 56.4 | 79.2 |
| FIRW-95 | 97.5 | 15.2 | 79.2 | 94.5 |
| TJA-25 | 28.3 | 9.1 | 16.1 | 25.3 |
| TJA-30A | 30.5 | 12.2 | 15.2 | 27.4 |
| TJA-30B | 30.5 | 15.2 | 15.2 | 30.5 |
| TJA-75 | 75.9 | 9.1 | 64.0 | 73.1 |
| TJA-100 | 96.0 | 9.1 | 83.8 | 93.0 |
| FIRE-33 | 33.5 | 15.2 | 15.2 | 30.5 |
| FIRE-75 | 61.0 | 12.2 | 45.7 | 57.9 |
| FIRE-100 | 85.3 | 9.1 | 73.1 | 82.3 |

The drilling program was initiated in April 1990 and completed approximately 2 months later, in July 1990. Wells were drilled by the USBR crew using a Failing 1500

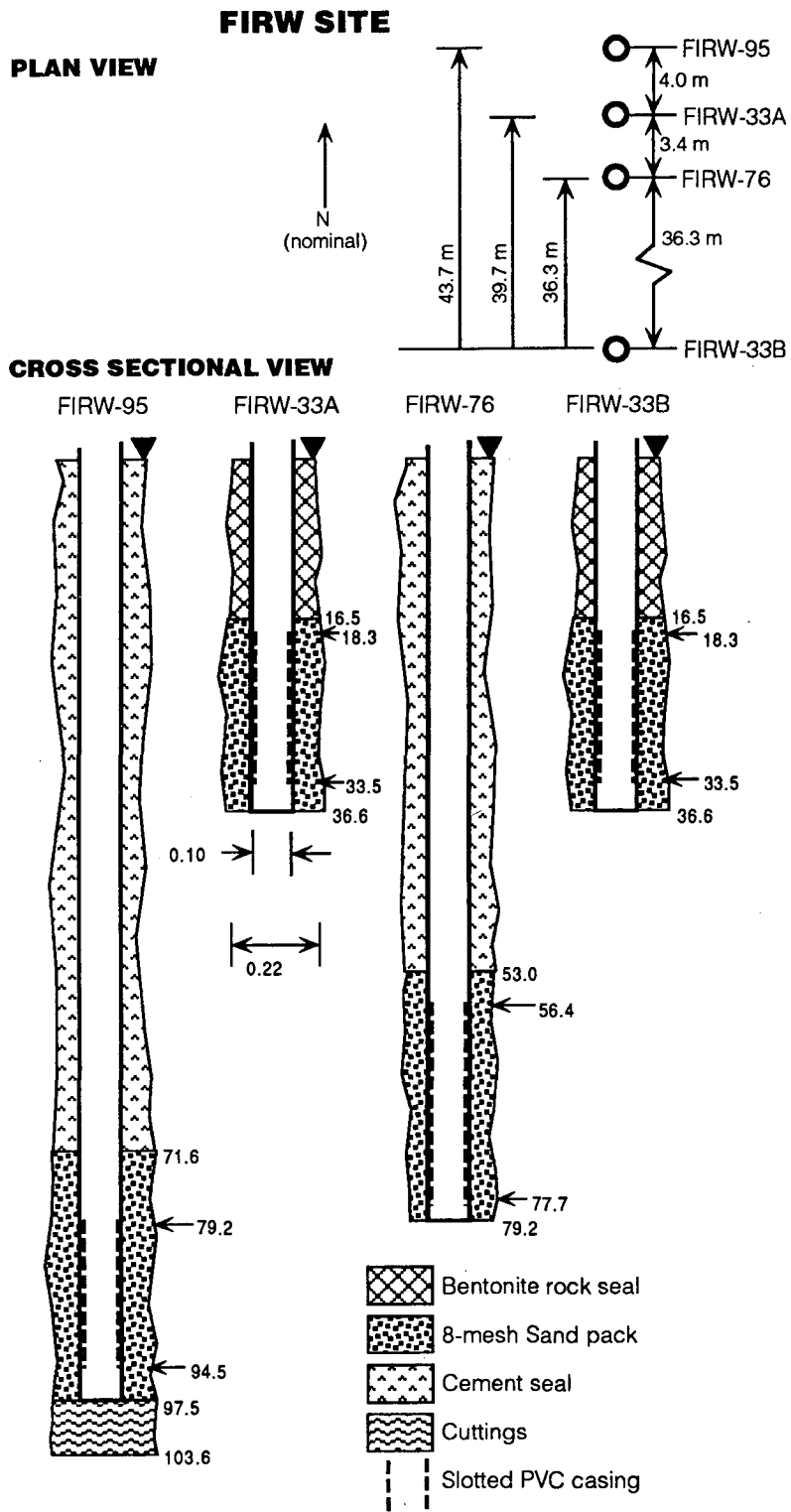


Figure 4.2. Plan and cross-sectional view of wells at the FIRW site. Inter-well distances are provided as the top of the figure. See Fig. 4.1 for site location.

Figure 4.3. Plan and cross-sectional view of wells at the TJA site. Inter-well distances are provided as the top of the figure. See Fig. 4.1 for site location.

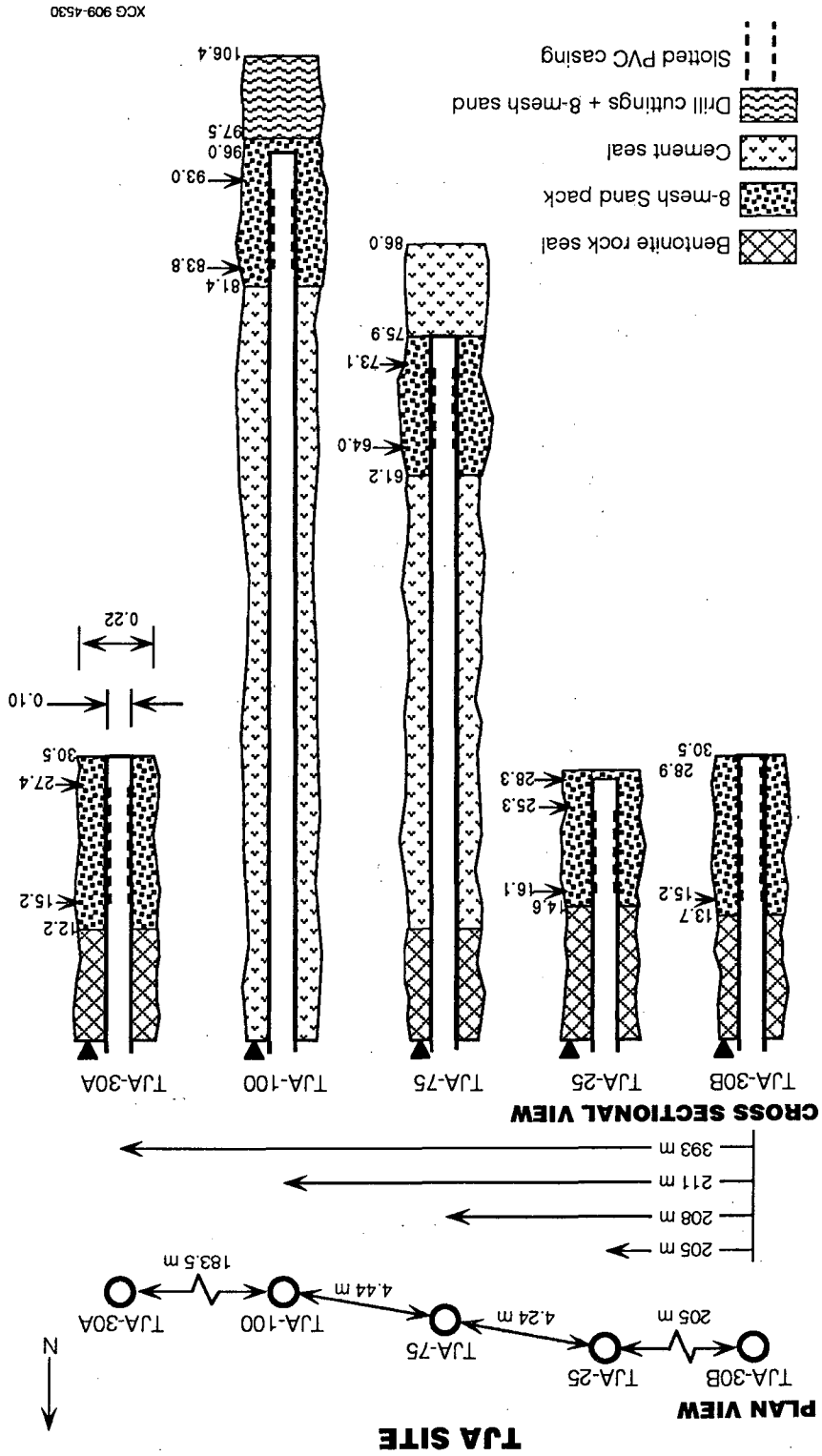
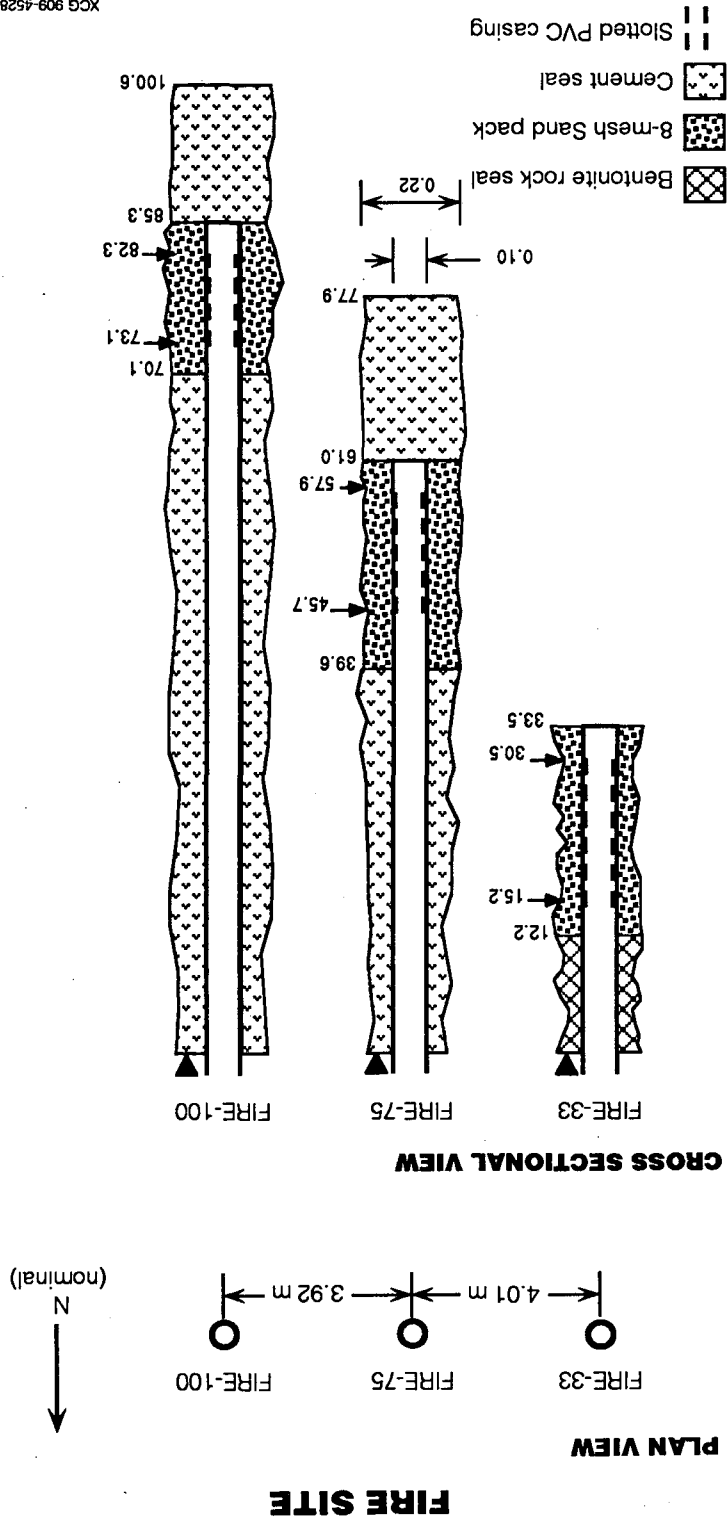


Figure 4.4. Plan and cross-sectional view of wells at the FIRE site. Inter-well distances are provided as the top of the figure. See Fig. 4.1 for site location.

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Holemaster rotary drilling rig. Wells were drilled with a 10 days-on, 6 days-off schedule. A typical work-day lasted from 10 to 12 hours.

A LBL geologist was on location throughout the drilling program, collecting geological specimens and overseeing well construction activities. Drill cuttings from the boreholes were logged continuously by the on-site geologist. A polymer drilling mud (EZ Mud manufactured by NL Baroid, Inc.) was used to maintain borehole stability and circulate drill cuttings out of the borehole. An 8.5 inch drag bit with a 6 m-long (20 ft.) drill collar was used to drill all the wells.

Geologic formations penetrated during the drilling program consisted solely of unconsolidated sediments; in each case a sequence of sands, silty and clayey sands, silts and clays were penetrated. Geologic descriptions of the drill cuttings consisted of visual and tactile textural analysis of the sediments (U. S. Dept. of Interior, Earth Manual, 1980). Detailed records of the color of the sediments were also kept to help in correlating lithostratigraphic units from well to well and site to site. Because of the mud-based rotary drilling system, it was often difficult to accurately characterize the texture of the sandy and silty units. Therefore, intact cores of the sediments were collected occasionally to help in constructing accurate geologic columns from these wells.

The drilling sequence at each site was as follows. Initially, a 100 m (330 ft.) deep borehole was drilled. When completed, geophysical logs, including caliper, neutron, natural gamma and electrical conductivity (using the Geonics, Inc. EM39 system) were obtained. Geologic descriptions of drill cuttings, intact cores and geophysical logs were used to identify large-scale features of the groundwater system. Based on this, target depth intervals for screening this and the remainder of the wells at each site were chosen. Wells were screened in sandy intervals likely to yield groundwater.

Immediately after logging the borehole, preparations for well completion began. A collarless drag-bit was lowered into the borehole to dislodge cuttings stuck to the wall and remove the last of the sediment-laden drilling mud. The drill bit was then removed and a

0.1 m (4 inch) diameter PVC casing was lowered to the bottom of the borehole. The bottom 3 m (10 ft.) of the casing is a "rathole" where sedimentary debris can collect without blocking the screened interval of the well. The well screen (slotted PVC) was located immediately above this. Screen lengths vary from site to site but are generally 10 m (30 ft.). After installing the casing, Clorox Bleach was mixed into the circulation system to breakdown the viscosity of the polymer drilling mud. The annulus of the screened interval of the well was then gravel-packed. Eight-mesh sand (8-mesh Monterey Sand, RMC Lonestar, cleaned, grated and kiln dried) was poured through a tremmie pipe into the annulus of the well. The tremmie pipe was first set at the bottom of the screened interval and gradually raised as the sand filled up the annulus. When the sand was from 1.5 to 3 m above the top of the screened interval, the tremmie pipe was raised another 6 m (20 ft.) and used to pump cement into the remainder of the borehole annulus. Once cement was observed coming out of the borehole annulus at the surface, pumping stopped and the tremmie was removed. The following morning the well was checked to see if the cement remained at the ground surface and iron protective casing was installed over the well. A benonite seal was used to fill the annulus if the cement had receded more than 3 ft from the ground surface.

After the first well was drilled, the remainder of the wells at each site were drilled and completed. Drill cuttings from all wells were logged to improve the description of the geologic column. Intact cores were collected in certain intervals to resolve any ambiguity between the geologic and geophysical logs. In wells where the top of the gravel-pack was less than 15 m below ground surface (50 ft.), the annulus was sealed with bentonite clay instead of cement (Hole Plug, Coarse Grade Wyoming Bentonite, produced by Baroid Drilling Fluids, Inc.).

4.1.2. Well Testing Program

The well testing program was designed to provide accurate information on the hydraulic conductivity and storage coefficient of the sandy units (aquifers) underlying the FIR. Also, to a limited extent, the testing program provided information on the extent to

which the clay layers (aquitards) prevent hydraulic communication between the sandy aquifers. The following four types of tests were used to gather this information:

- Single-well steady state pumping tests to obtain an initial estimate of the hydraulic conductivity of the screened interval of a well,
- Single well pressure-transient tests to evaluate the condition of the well and obtain a more accurate measurement of the hydraulic conductivity,
- Pressure interference tests to measure hydraulic communication between wells, to obtain larger-scale data on hydraulic conductivity and to measure the storage coefficient of the aquifer and
- Vertical pressure interference tests to measure hydraulic communication across the aquitards separating the aquifers.

Single-well steady state tests were performed on all wells with a trash pump. Single-well transient tests were performed on 7 wells, including all those that could be pumped at greater than 0.5 L/s (about 10 gallons per minute) with a submersible pump set at a depth of less than 30 m (100 ft.). Pressure interference tests were conducted at the FIRW and TJA sites, where two or more wells were drilled and screened in the same depth intervals. Vertical interference tests were conducted at the FIRW and TJA sites. Table 4.2 summarizes the well tests that were conducted as part of this program. The following describes the test methods employed in greater detail.

Before testing, wells were developed and cleaned of debris by a combination of airlifting and pumping. Airlifting involved injecting air down a narrow plastic tube whose outlet was located at the bottom of the well. The air-water mixture then flowed up the well where it was discharged to the surface. The 100-psi air compressor used to airlift the wells created water discharge rates between 1 to 5 L/s (20 to 100 gallons per minute). Some of the wells were also developed with a trash pump, with periodic on-off cycles used to surge the well and remove drilling debris from the sand pack. From one to several hours of airlifting and pumping was required before water coming out of the well was free of sediment

and any traces of the polymer drilling mud. After development, the depth of the well was measured to ensure that the screened interval was not blocked by sediments.

Table 4.2. Summary of the well tests conducted at the FIRW, TJA and FIRE sites as part of the LBL groundwater investigations at the FIR.

| Well Name | Single-well Steady State | Single-well Transient | Pressure Interference | Vertical Pressure Interference |
|-----------|-----------------------------|--------------------------|--------------------------|--------------------------------------|
| FIRW-33A | x | | Observation | Observation |
| FIRW-33B | x | x | Pumping | Pumping |
| FIRW-76 | x | x | | Pumping and Observation |
| FIRW-95 | x | | | Observation |
| TJA-25 | x | x | Observation | Pumping and Observation |
| TJA-30A | x | | Observation | |
| TJA-30B | x | x | Pumping | |
| TJA-75 | x | | | Observation |
| TJA-100 | x | x | | Pumping and Observation |
| FIRE-33 | x | x | | |
| FIRE-75 | x | | | |
| FIRE-100 | x | | | |

Single-well steady state pumping tests were then conducted on each well. This involved pumping the well at a constant flow rate for about one hour with a trash pump. Pumping rates, which were measured volumetrically, ranged from 0.1 to 1.3 L/s (1.5 to 20 gallons per minute). Accuracy of the pumping rate measurements was $\pm 10\%$. Waterlevel measurements were made in the well with a resistance-type sensor before and during pumping. The “steady-state” drawdown was defined as the difference between the waterlevel in the well before pumping and the waterlevel after one hour of pumping. Although truly “steady-state” conditions were not achieved in this one hour period, measurable changes in the waterlevel ceased (resolution of the waterlevel sensor was 2 cm). Using this

approximate definition of the “steady-state,” drawdowns ranged from 2 to 3 m (6.5 to 10 ft.). Analyses of the steady-state pumping tests are described in Section 4.3.1.

Single-well pressure transient tests were conducted in all wells with productivity indices (flow rate/drawdown) of greater than 0.06 (L/s)/m (0.3 gallons per minute per foot of drawdown). These tests require pumping the well at a constant rate while the pressure drawdown in the pumping well is measured periodically with a very accurate pressure gauge. After a specified period, the pump is turned off and the pressure recovery is also monitored. A computer-automated system was used to carry out these tests (Benson, 1986). The system, housed in a delivery truck, consists of quartz crystal piezoelectric pressure transducers, a Hewlett Packard computer and associated electronics for converting input signals from the sensors to digital information. The resolution of the pressure sensors is 6 Pa (0.001 psi). This high resolution provides extremely accurate pressure transient data from which the hydrologic properties of the geologic formations can be determined. To carry out these tests the pressure sensor and submersible pump were lowered into the well to a depth of at least 15 m (50 ft). The pressure recording equipment was then activated. Once a stable pressure reading was observed, the pump was turned on. Pumping durations ranged from 6 to 24 hours. Pressures were measured every second for the first 100 seconds, every 10 seconds for the next 1000 seconds, every 60 seconds until 3600 seconds and then every 600 seconds until the pump was turned off. A similar recording schedule was used to monitor the pressure recovery. Flow rates were measured with an impellor flow meter with an accuracy of $\pm 5\%$. An example of the pressure transient test data from one well is provided in Fig. 4.5. As shown, before pumping, the pressure was steady. At 3600 s the pump was turned on (0.8 L/s, 13 gallons per minute) and the pressure dropped quickly. The pressure continued to decline slowly until the pump was shut off. A similar pattern is apparent when the pump is turned off. Analyses of the single-well pressure transient tests are described in Section 4.3.2.

At the FIRW and TJA sites pressure interference tests were also conducted. These test are carried out by pumping one well while measuring the pressure drawdown in a

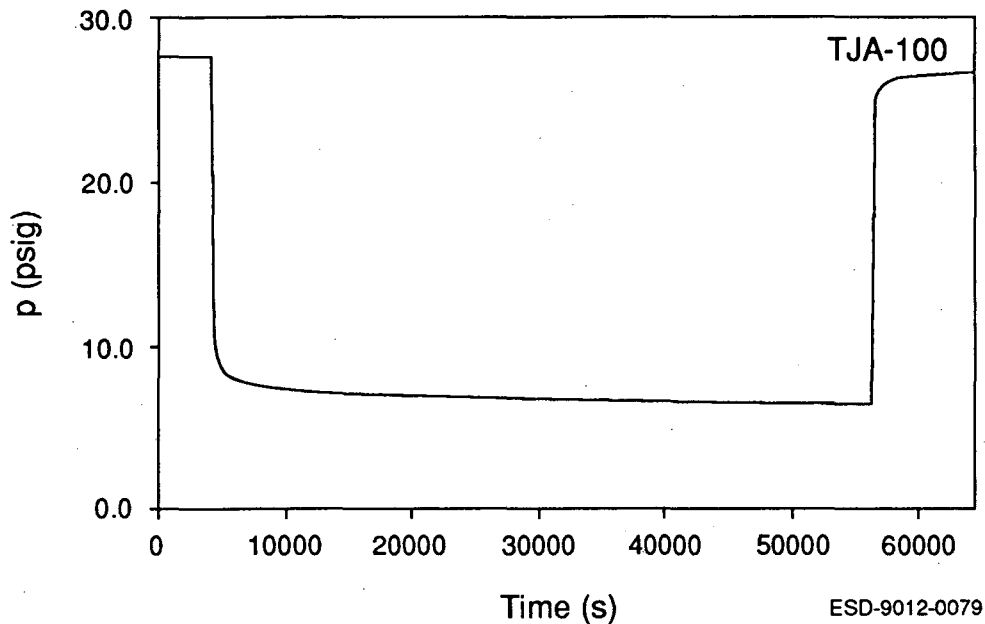


Figure 4.5. Single-well pressure transient test data from well TJA-100.

nearby passive monitoring well located in the same aquifer. The same instrumentation and monitoring system was used for these tests as for the single-well pressure transient tests described above. At the FIRW site well FIRW-33B was pumped while pressure drawdown was measured in FIRW-33A. The pressure drawdown data at FIRW-33A, located 39.7 m (130 ft.) from FIRW-33B, are shown in Fig. 4.6. Over the 16 hour period while FIRW-33B was pumped at a rate of 2.14 L/s (34 gallons per minute) the pressure at the observation well dropped about 5000 Pa (0.8 psi). Similar data were obtained from the TJA site where wells TJA-25 and TJA-30A were monitored while TJA-30B was pumped. Analyses of the pressure interference tests are described in Section 4.3.3.

Vertical pressure interference tests were also conducted at the FIRW and TJA sites. These tests involve pumping a well located in one aquifer while measuring the pressure drawdown in another aquifer located above or below the aquifer from which water is pumped. At the TJA site, TJA-25 was pumped for 20 hours at 0.76 L/s (12 gallons per minute) while the aquifer pressure was measured in TJA-75 and TJA-100 (see Fig. 4.3 for

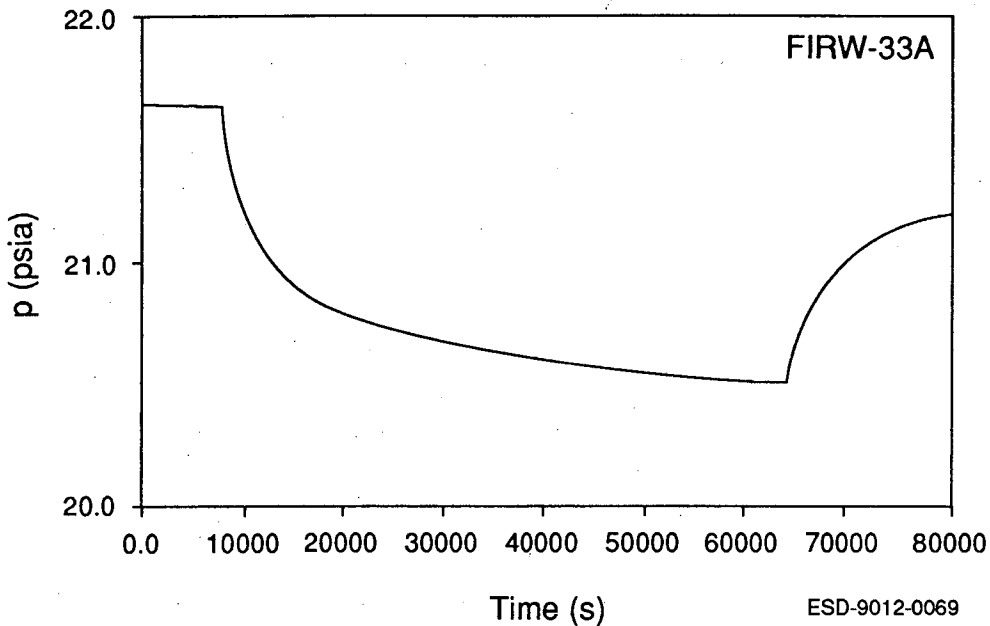


Figure 4.6. Pressure interference data at well FIRW-33A while well FIRW-33B was pumped at a rate of 2.14 L/s (34 gallons per minute).

well locations). No detectable pressure drawdown occurred, indicating that the vertical hydraulic conductivity is extremely low. Stated another way, the clay layers in the subsurface form effective aquitards, limiting vertical migration of fluids under the FIR. Similar results were obtained when TJA-100 was pumped and pressures were measured in TJA-75 and TJA-25. At the FIRW site two vertical pressure interference tests were also conducted. For one test, pressures were monitored in FIRW-95 and FIRW-76 while FIRW-33B was pumped for 16 hours at 2.14 L/s (34 gallons per minute). In the other test, FIRW-76 was pumped while pressures were monitored in FIRW-33A and FIRW-95. Again, no pressure drawdown was observed in either of these tests. Because of the limited quantitative information these tests provided they will not be addressed further in this report.

4.1.3. Groundwater Quality Testing Program

Groundwater samples were collected from each well drilled as part of this program. Usually, at least two sets of samples were collected, one shortly after the well was devel-

oped and another while the well tests were conducted. Samples were collected from the discharge pipe of either the trash pump or submersible pump. In general, three sub-samples were obtained each time the well was sampled; a filtered sample, a filtered and acidified sample and an unaltered sample. Groundwater samples were collected into double-rinsed 100 ml bottles. Filtering was done with a double-rinsed syringe connected to 0.45 μm cellulose nitrate filter cartridge. To acidify the samples, 5 ml of HCl was added to the sample bottle. A detailed description of the sampling protocol is provided in the LBL Quality Assurance and Control Program (ESD, 1991).

Groundwater samples were transported back to the LBL where they were analyzed in-house for major ions and selected trace elements. Inductively coupled plasma atomic absorption spectrophotometry (ICP-AAS) was used to measure Na, Ca, Mg, Mn, B, Fe, S and Mo concentrations. Hydride generation atomic absorption spectrophotometry (HG-AAS) was used to measure As, Se, Li and K concentrations. Chloride concentrations were measured by silver nitrate titration. Quality assurance and quality control measures include analysis and documentation of duplicate samples, standards and spikes. Quantitative assessment of the performance of our analytical laboratory is overseen by a dedicated QA/QC manager. A detailed description of our QA/QC program for the trace elements As, Se and B is provided in the LBL Quality Assurance and Control Program (October, 1990).

Data and discussion of the results of the groundwater sampling program are provided in Section 4.4.

4.2. Lithostratigraphy Under FIR

The Fallon Indian Reservation (FIR) overlies sediments deposited by an alternating sequence rivers and deep lakes in the Lahontan Valley. The geology of the area is most easily understood in this context. At least twice within the last 60,000 years, lands of the FIR have been inundated by Lake Lahontan (Morrison, 1964; Benson, 1978). During these periods, which lasted from thousand to tens of thousands of years, clays were deposited in a lake that was over 100 m deep (Morrison, 1964). When the Sierran glaciers

receded, Lake Lahontan dried up and much like today, rivers, streams and marshes criss-crossed the area. Sands carried from the Sierra Nevada were deposited by the rivers and streams, silts and clays were deposited in marshes and wind-blown sediments fell to the ground. Nearly continuous records of this history are present in the sediments underlying the FIR.

Numerous published reports are available describing studies of the groundwater hydrology and geology of the Lahontan Valley. The most comprehensive geologic evaluation of the Carson Desert, including the Lahontan Valley, was undertaken by Morrison (1964). Much of the interpretive work in this report draws heavily on this earlier report. More recently, the need for potable groundwater has motivated local and regional assessment of the hydrogeology and water quality in the Lahontan Valley (Sinclair and Loeltz, 1963; Lico *et al.* 1986; Glancy, 1986; and Welch *et al.* 1989). The search for geothermal sources of hot water and steam also provided relevant data on the subsurface geology of the FIR (Morgan, 1982). In particular, Morgan (1982) identified three potential aquifers within the surface-most 70 m under FIR. This motivated the present search for aquifers contributing salts and trace elements to shallow drainage; these aquifers may also be used for reinjection of drainage water. Studies specific to FIR also provide information on water quality of shallow groundwater (<10 m), soils and geology (CH2M Hill, 1989; USBR, 1987; and Hoffman *et al.*, 1990). All the above sources of information have been drawn upon in the LBL investigations.

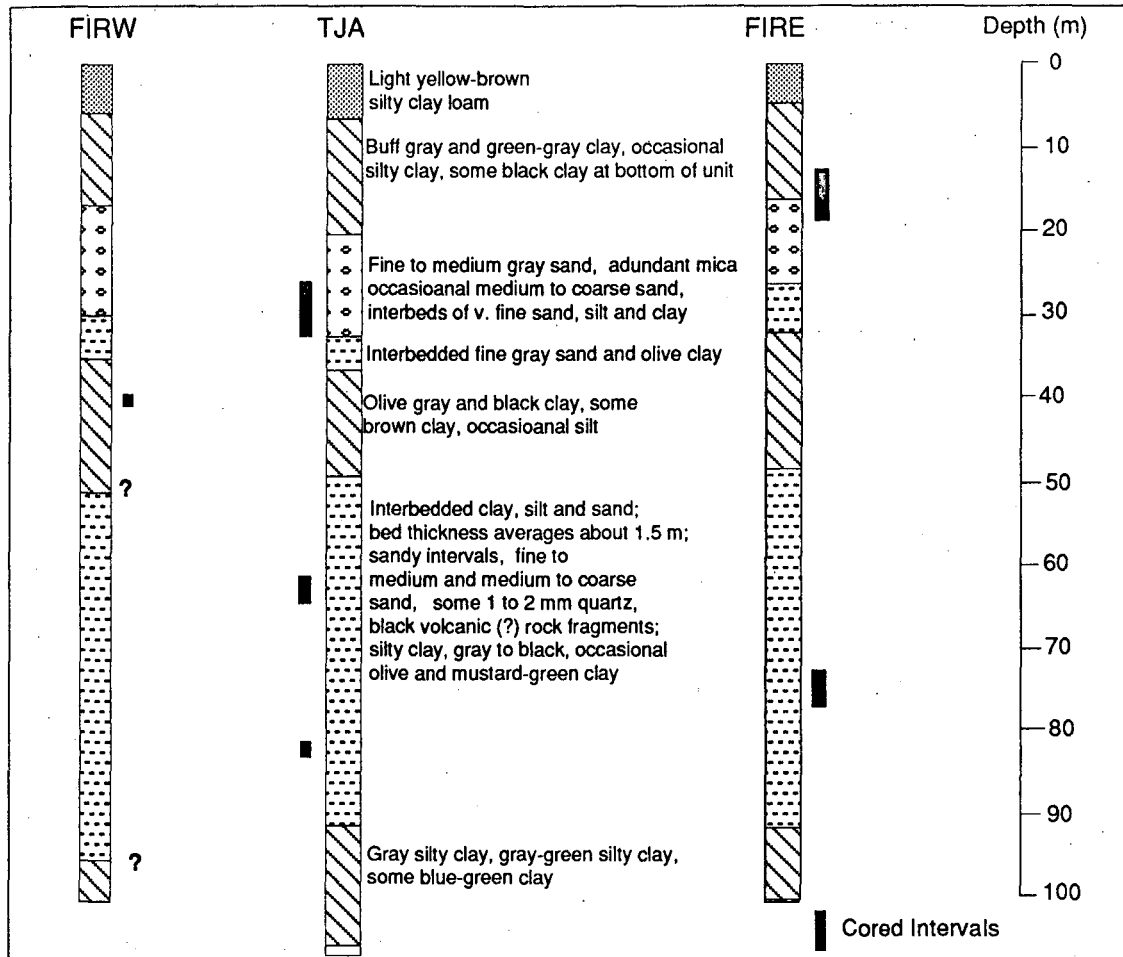
4.2.1. Geological Logging of Drill Cuttings and Cores

The subsurface geology under FIR is characterized by a sequence of sands, silts and clays deposited under the conditions described above. Figure 4.7 illustrates a schematic of the major features identified by logging drill cuttings and inspecting intact cores. The black bars adjacent to the geologic columns indicate depth intervals where intact cores were obtained. As shown in Fig. 4.7, the overall geologic features from site to site are remarkably similar.

In the paragraphs to follow, each unit indicated in Fig. 4.7 is described. As mentioned in Section 4.1.1., detailed textural characterization of the sandy and silty units was often difficult because of mixing between the drill fluids and formation sediments. The mixture was typically a slurry, without individual "cuttings" of the formation. Intact cores were extremely useful in these regions. Characterization of the clay units was relatively easy because drill cuttings were large enough (0.5 cm by 1 cm by 2 cm) to inspect visually. Contacts between the coarser grained units (sand) and finer grained units (silt and clay) were usually easy to identify by changes in the drilling rate.

The first 3 to 5 m consist of light yellow-brown loam, similar to the surface soils at FIR. Although clay, sand and silt fractions vary from site to site and the depth to which this unit extends is variable, it is distinctly different than the massive clay unit underlying it. Similar observations have been made from other FIR investigations (CH2M Hill, 1989; Dollarhide, 1975). This unit is presumably the PA-1 unit (Potential Aquifer-1) described in Morgan (1982). However, new information suggests that at FIR it is unlikely that this forms a significant aquifer. Perhaps locally, where it is relatively thick and coarse textured, water could be extracted at reasonable pumping rates. Nevertheless, water quality in this unit is on-average extremely poor (USBR, 1987; CH2M Hill, 1989).

A massive clay unit extends from about 5 to 18 m below the ground surface. The light-gray to green-gray high plasticity clay is very similar from site to site. Lack of internal structure and wide-spread uniformity suggest that this was deposited in a deep lake environment. Near the bottom of this unit a black organic-rich clay with occasional silty-clay stringers is present. Presumably this represents the transition from a marsh environment to a deep-lake environment. This unit is tentatively correlated with the Seho formation described by Morrison (1964). It is also probably the same unit that Morgan (1982) identified as CB-1 (Confining Bed-1). The bottom of this unit is marked by a sharp transition to a sandy unit below it.



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Figure 4.7. Geologic columns from the FIRW, TJA and FIRE sites. Columns represent composites of data from all wells at each site. Black bars adjacent to the geologic columns represent depth intervals from which intact cores of the sediments were obtained.

A fine to medium grained sandy unit extends from about 18 to 32 m below the ground surface. Silty and clayey stringers are not uncommon in this unit, particularly at the FIRE site. Occasional beds of medium to coarse grained sand are present, indicating deposition in relatively high-energy stream channels. Overall, sediments are finer-grained at the bottom of this unit, with a gradual transition to interbedded sands, silts and clays. This unit is tentatively correlated with the Wymaha Formation of Morrison (1964) and the PA-2 (Potential Aquifer-2) identified by Morgan (1982). In the city of Fallon it is an important aquifer for supplying drinking water to local residents. Glancy (1986) describes it as the

“shallow-intermediate” aquifer. The interbedded sands and clays at the bottom of this unit provide a gradual transition to second deep-lake period described in the paragraph that follows.

The second massive clay unit underlying FIR extends from a depth of about 35 to 50 m. Again, it is remarkably uniform from site to site. The massive olive gray highly plastic clay grades to black and brown clay near the bottom of the unit. Occasional brown silty clay beds are present. The massive nature and uniformity of this unit suggests that it was deposited in a deep-lake environment. Gradational changes at the bottom are associated with slowly rising waterlevels, from a marsh to deep-lake environment. This unit is tentatively correlated with the Eetza formation of Morrison (1964) and the CB-2 (Confining Bed-2) of Morgan (1982). The bottom of this unit is marked by a sharp transition to sandy sediments deposited in Pre-Lahontan times.

From a depth of 50 to about 95 m below the ground surface sediments are characterized by an irregular sequence of interbedded sands, clays and silts. Transitions between beds are sharp, as suggested by rapid changes in drilling rates and drill-bit chatter. Bed thickness averages about 1.5 m (5 ft). The textural and mineralogical composition of the sand beds was highly variable. Some beds consisted of medium to coarse sand, with black (volcanic), red and white rock fragments (1 to 2 mm). Others consisted of relatively uniform fine-grained gray sand. Clay beds were less plastic than clays in the massive units described previously; colors ranged from light to dark gray, olive, black and mustard-green. Although the overall character of this unit was similar from site to site, it was not possible to correlate individual beds over these distances. Most likely, these sediments were deposited in a fluvial environment with periodic spatial and temporal changes in depositional patterns. This unit is tentatively correlated with the Paiute formation described by Morrison (1964) and the PA-3 unit (Potential Aquifer-3) described by Morgan (1982). The bottom of this unit is marked by a gradual transition to a third massive clay unit.

At a depth of about 95 m (310 ft) all the wells penetrated a massive gray-green highly plastic silty clay. The bottom of this unit was not clearly encountered in any of the wells so

it is not possible to place a lower bound on its extent. Most likely, these sediments were deposited in a Pre-Lahontan Lake. Limited evidence for such a lake is provided by Morrison (1964).

4.2.2. Geophysical Logs

Geophysical logs, including caliper, natural gamma and neutron, were obtained from the deepest well at each site before the casing was installed. Attempts were also made to run short-normal and long-normal resistivity logs; these were unsuccessful due to the extremely high electrical conductivity of groundwater in this area. As an alternative, bulk-conductivity logs (Geonics Inc. EM39) were run after the wells were cased and cemented. Natural gamma, neutron and bulk-conductivity logs were also run in two of the shallower wells (TJA-30A and FIRW-33B) after they were cased. Geophysical logs obtained from these wells are shown in Figs. 4.8 through 4.12.

The caliper log provides a measure of the borehole diameter. Ideally, the bore diameter is the same as the diameter of the drill bit (in this case, 0.22 m, (8.5 inches)). However, it is not uncommon for sandy units to slough into the well, creating "caved" regions where the bore diameter is bigger than expected. Also, clays typically "squeeze" into the borehole, reducing its diameter. Both of these features are apparent in Figs. 4.8 through 4.10. The borehole diameter is more irregular in the sandy units than in the clay units. Also, at the transition from the second massive clay unit (about 50 m), the borehole is enlarged when the first sands are encountered. The deepest clay unit (>95 m) has a tendency to "squeeze" into the borehole. In spite of these variations, the majority of the borehole is close to the expected value of 0.22 m in all the wells.

Gamma logs measure the natural decay of radioactive elements in the sediments. Finer grained sediments such as silt and clay have higher contents of potassium, uranium and thorium bearing minerals. Therefore, high gamma counts are usually associated with silts and clays and low gamma counts are associated with sands. Exceptions to this general rule may be found in sands with large heavy mineral accumulations. The gamma logs

shown in Figs. 4.8 through 4.12 are interesting in two respects. First, they show little coarse-scale correlation with the lithologic descriptions provided in Section 4.2.1. For example, gamma ray signals are not particularly high in the 5 to 18 m depth interval where the first massive clay is. Also, large positive anomalies are present in the gamma signal that are not obviously correlated from well to well. Intact sediment cores were collected to resolve this ambiguity. Tentatively, positive gamma anomalies are attributed to high concentrations of radioactive elements associated with organic-rich marsh sediments. Because of their unusual characteristics, gamma logs were not very helpful in correlating subsurface conditions from site to site. The positive gamma anomalies that Morgan (1982) attributed to a volcanic ash layer at a depth of 20 to 30 m were not observed in these wells.

The neutron log proved to be the most useful for well to well correlations. The neutron log provides a measure of the water content of the sediments (or porosity in a fully saturated soil). Clay intervals have the highest water content, which is indicated by lower neutron counts. Conversely, sandy intervals have lower porosity and higher neutron counts. The neutron log data presented in Figs. 4.8 through 4.12 are plotted with increasing neutron counts to the left of the graph; therefore, clayey intervals are indicated where data are closest to the right-hand side of the graph. Based on comparison of drill cuttings and the neutron logs, intervals with neutron counts of less than 200 cps (counts per second) are identified as clays. As illustrated in Fig. 4.8, neutron logs provided detailed corroboration of information obtained from geological evaluation of the drill cuttings. Correlation of data from the neutron logs is discussed in Section 4.2.3.

Bulk-rock electrical conductivity (σ_B) in a fully water-saturated sediment is influenced by a number of parameters, as indicated by the following relationship:

$$\sigma_B = a \sigma_w \phi^m \quad [\text{Eq. 4.1}]$$

where a (about 0.9 in this case) and m (about 1.4 in this case) are constants determined by the rock type; σ_w is the electrical conductivity of the groundwater and ϕ is the porosity of the rock (Archie, 1942). Depending on which of these parameters is most variable, a bulk-

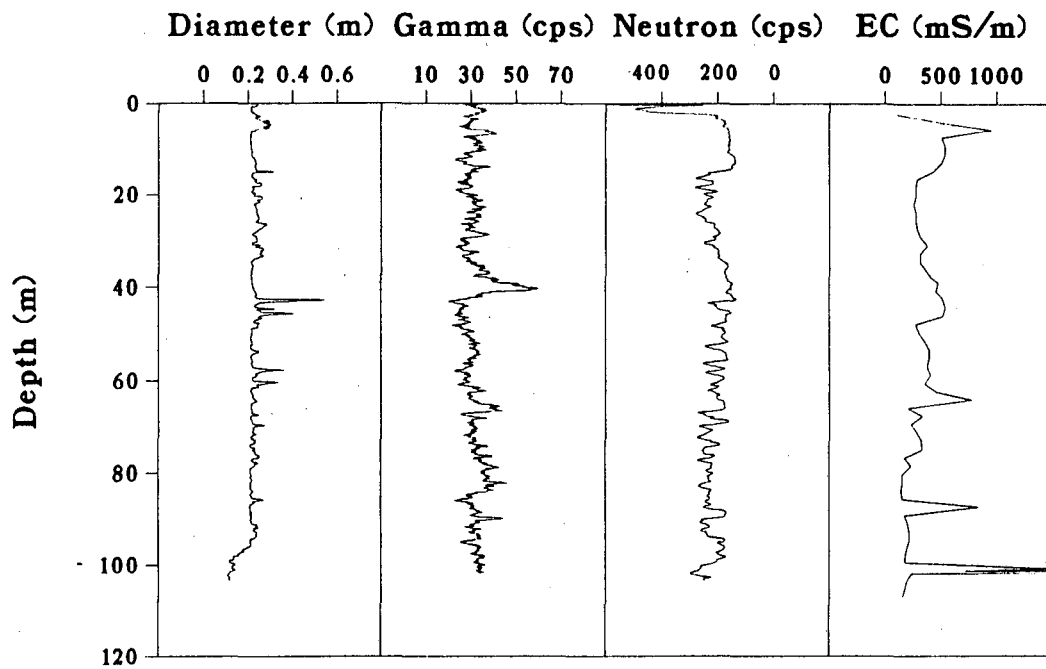


Figure 4.8. Geophysical logs from FIRW-95. The first column is the diameter of the borehole, as measured with a borehole caliper tool. The second column is the natural gamma log. The third is the neutron log. The last column is the apparent bulk-rock electrical conductivity, as measured with the Geonics Inc. EM39 system. The first three logs were obtained in an un-cased borehole. The last was obtained after the casing was installed.

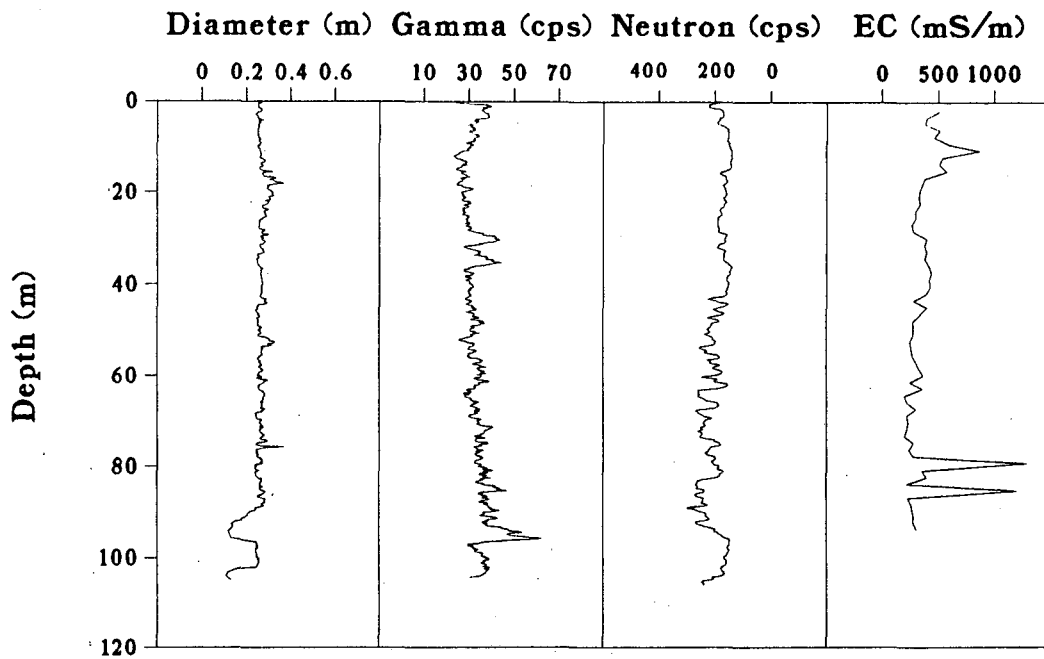


Figure 4.9. Geophysical logs from TJA-100. See Fig. 4.8 caption for description.

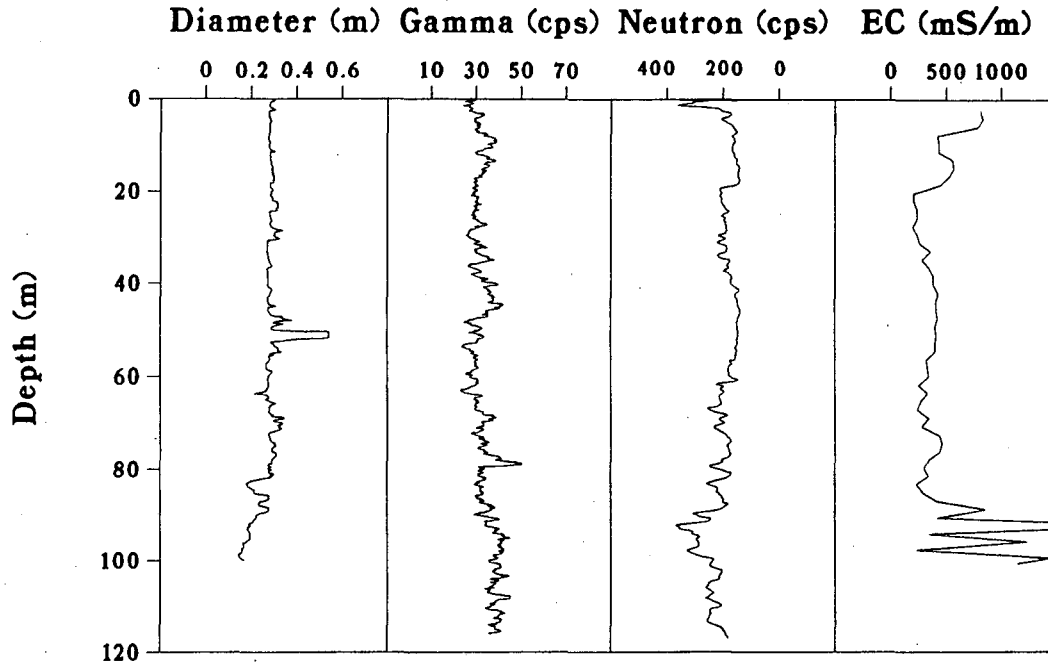


Figure 4.10. Geophysical logs from FIRE-100. See Fig. 4.8 caption for description.

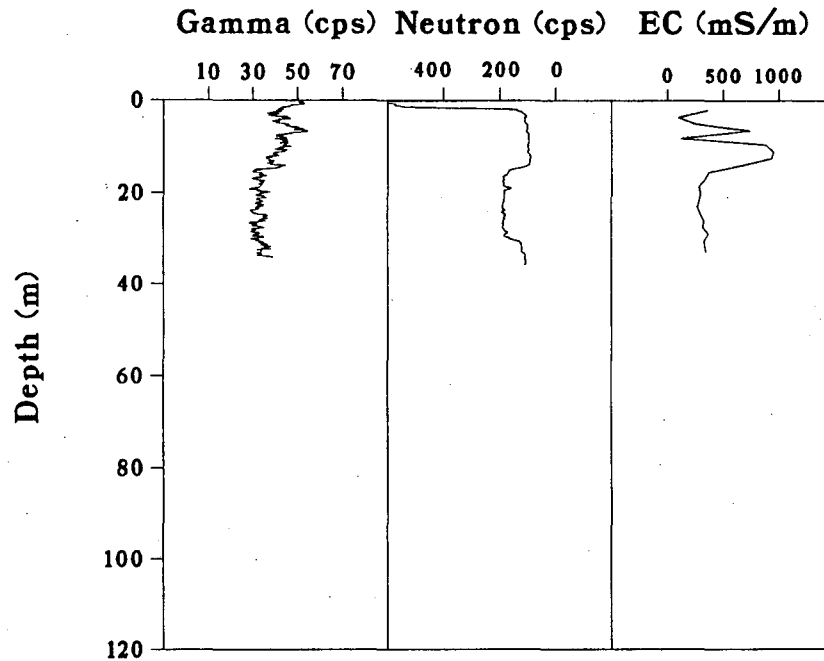


Figure 4.11. Geophysical logs from FIRW-33B. Note that all three logs were obtained after the casing was installed. The first column is the natural gamma log. The second is the neutron log. Differences between this gamma log and the log from FIRE-95 (Fig. 4.8) are attributed partially to the influence of the well casing and deeper water level when the logs were run in this well. The last column is the apparent bulk-rock electrical conductivity, as measured with the Geonics Inc. EM39 system.

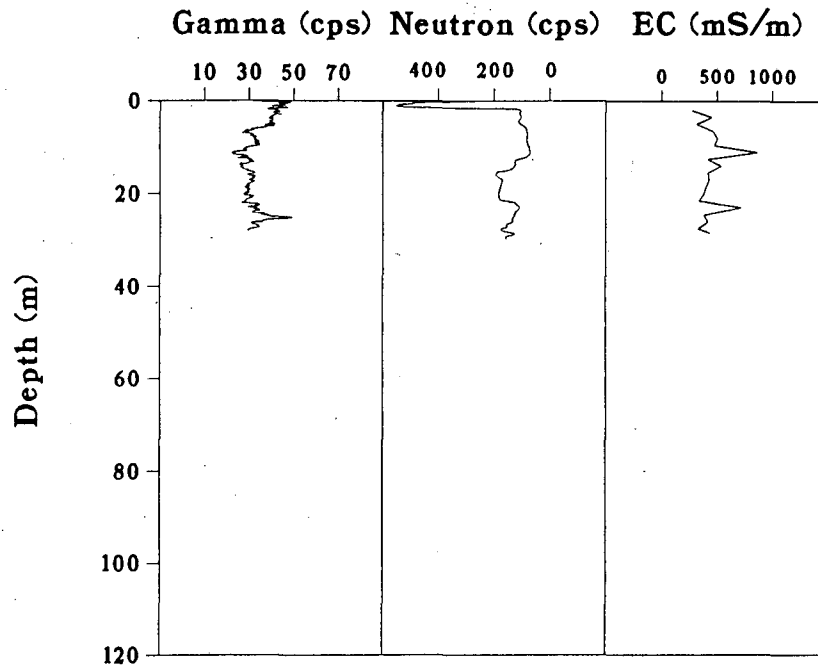


Figure 4.12. Geophysical logs from TJA-30A. Note that all three logs were obtained after the casing was installed. The first column is the natural gamma log. Differences between this gamma log and the log from TJA-100 (Fig. 4.9) are attributed partially to the influence of the well casing and deeper water level when the logs were run in this well. The second is the neutron log. The last column is the apparent bulk-rock electrical conductivity, as measured with the Geonics Inc. EM39 system.

where a (about 0.9 in this case) and m (about 1.4 in this case) are constants determined by the rock type; σ_w is the electrical conductivity of the groundwater and ϕ is the porosity of the rock (Archie, 1942). Depending on which of these parameters is most variable, a bulk-conductivity log will provide a qualitative measure of how it varies with depth. In the subsurface environment under FIR, the electrical conductivity of the water (σ_w) is the most variable. Therefore, bulk-conductivity can be used to obtain a qualitative measure of how the salinity, which is directly related to σ_w , varies with depth. The bulk-conductivity logs shown in Figs. 4.8 through 4.12 show a general pattern of decreasing conductivity with depth. Stated another way, these logs indicate that the most saline waters are present in the surface-most 20 m. This information is generally consistent with water quality data provided in Section 4.4. Another interesting feature of these logs is the sharp positive conductivity anomalies observed sporadically. These are indicative of “pockets” of highly

these sediments. Both explanations are plausible in this environment and attest to the historically high-salinity of Carson Desert sediments.

4.2.3. Correlation of Lithostratigraphic Units

By combining neutron logs with geological data, lithostratigraphic correlations between the sites have been made. The resulting SW-NE trending cross-section is shown in Fig. 4.13. Five major units have been identified in this cross section: three clay units, C₁, C₂ and C₃; and two predominantly sandy units, S₁ and S₂. Geologic descriptions of these units are provided in Fig. 4.7 and Section 4.2.1.

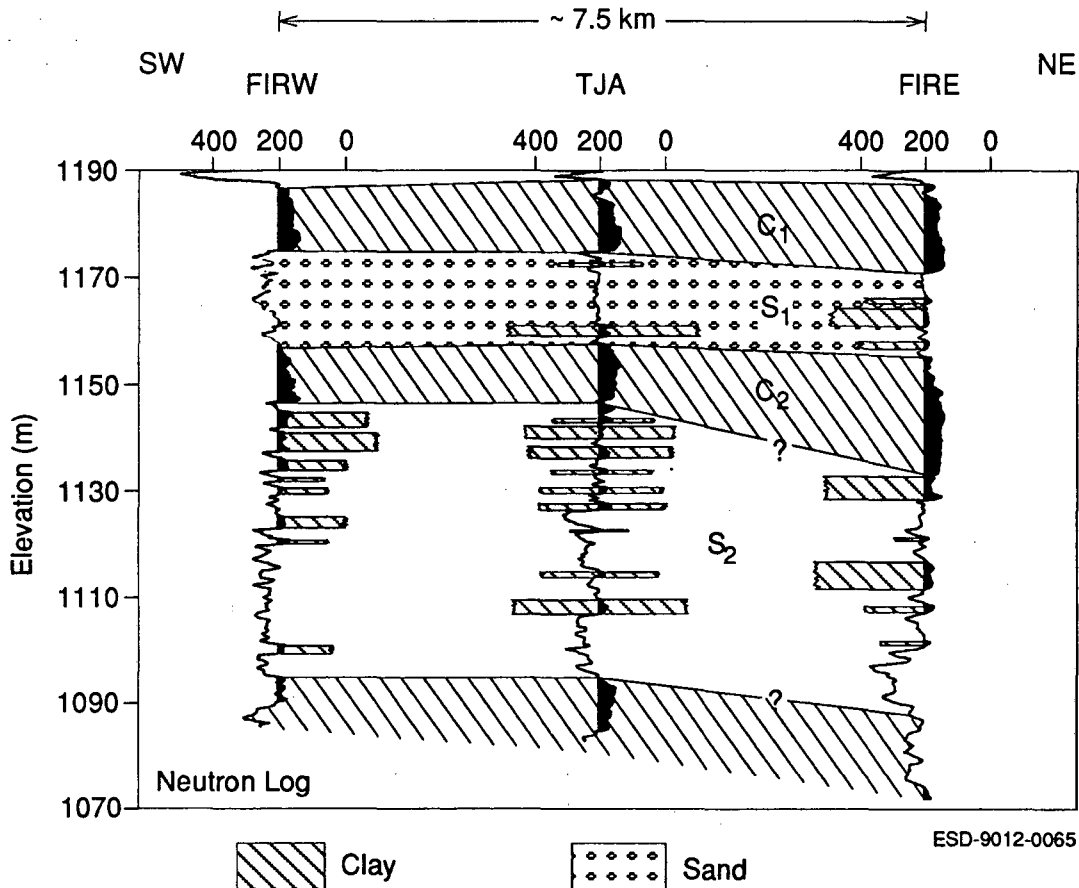
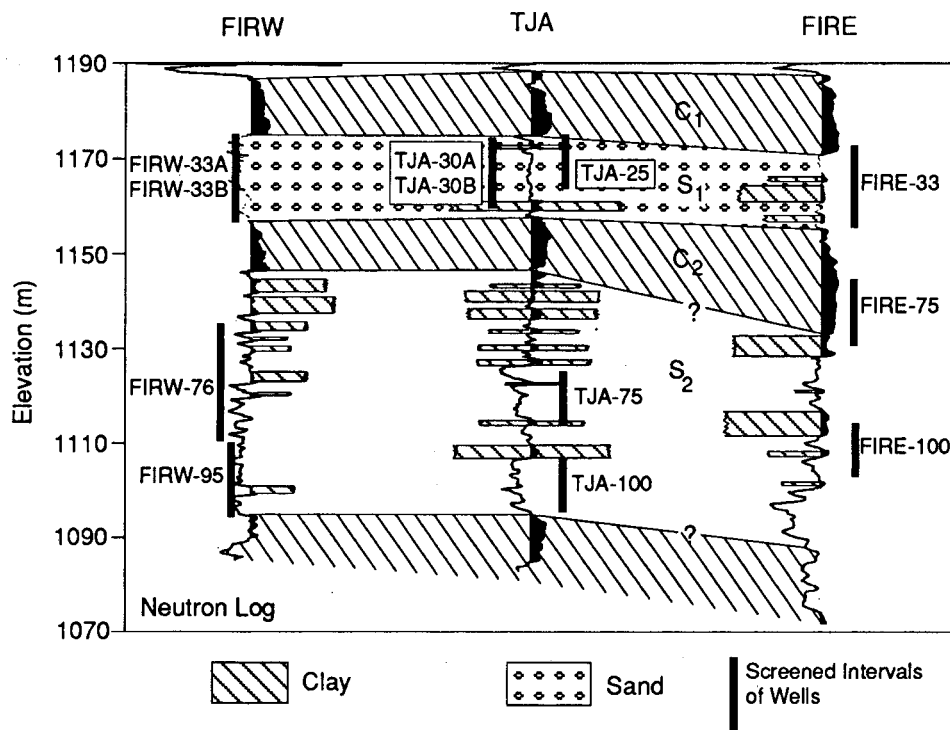


Figure 4.13. SW-NE trending cross-section of the subsurface geology under FIR. Correlations were obtained using the neutron logs (shown for each site) and information from drill cuttings and intact cores.

To highlight clay units identified by the neutron log, all values less than 200 cps have been shaded. This clearly illustrates the massive nature of the clay units identified as C₁ and C₂ and emphasizes the complex stratigraphy of the S₂ unit. Also, a general trend towards a predominance of finer grained sediments to the NE is also suggested, particularly in the surface-most 50 m. This general pattern is consistent with a larger-scale trend identified by Glancy (1986).

4.2.4. Fluid Potentials and Hydrostratigraphy

At each site wells were screened in the S₁ and S₂ units. A schematic showing depth intervals of well screens relative to the various lithostratigraphic units is provided in Fig. 4.14. At the FIRW site two wells are screened at the same depth in the S₁ unit (FIRW-33A and FIRW-33B) and two are screened at different depths in the S₂ unit (FIRW-76 and FIRW-95). (A list of well depths is provided in Table 4.1; Section 4.1.1.). At the TJA site



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Figure 4.14. SW-NE trending geologic cross-section showing the depth intervals where wells were screened at the FIRW, TJA and FIRE sites.

three wells are screened in the S₁ unit (TJA-25, TJA-30A and TJA-30B) and two in the S₂ unit (TJA-75 and TJA-100). Only three wells were drilled at the FIRE site. One well was screened in the S₁ unit (FIRE-33), one in the transitional region between the C₂ and S₂ units (FIRE-75) and one in the S₂ unit (FIRE-100).

Waterlevels were measured in each well after they were developed. At a given site, comparison between waterlevel depths provides a means of assessing the direction and magnitude of vertical hydraulic gradients. Comparison of the watertable elevation (ground surface elevation minus depth to the waterlevel) from site to site can be used to assess the magnitude and direction of horizontal hydraulic gradients. Unfortunately, at present, ground surface elevation data from the study sites are not available and therefore, horizontal gradients can not be evaluated.

Waterlevel measurements for all the wells are provided in Table 4.3. For wells screened in the S₁ unit, waterlevels are from 1.45 to 1.92 m below the ground surface. Waterlevels in wells screened in the S₂ unit are considerably shallower at the FIRW site. At the TJA and FIRE sites, wells screened in the S₂ unit are artesian. Standpipes were put on artesian wells to estimate artesian pressures; these values are also provided in Table 4.3. When left unsealed, TJA-100, FIRE-75 and FIRE-100 flowed at rates of about 0.2 L/s (several gallons per minute).

The relatively large upward hydraulic gradient (0.04 m/m) measured at the FIRW site indicates that the C₂ clay is an effective aquitard that limits upward migration of groundwater. Similarly and perhaps even larger vertical hydraulic gradients at the other sites support this conclusion. Data from the TJA and FIRE sites suggest a relatively large upward gradient in the S₂ unit, attesting to the limited vertical permeability of this unit as well; interbedded clays form effective local barriers to upward migration of groundwater.

The well tests described in Section 4.1.2. provide characterization of the hydrologic properties of the S₁ and S₂ units identified in Fig. 4.13. As shown in Fig. 4.14, the array

of wells provides broad coverage of both these units. The results of the well tests are summarized in Section 4.3.

Table 4.3. Waterlevel measurements from the FIRW, TJA and FIRE sites (measured in September, 1990).

| Well Name | Waterlevel Depth Below Casing Top (m) | Height of Casing Top (m) | Waterlevel Depth Below Ground Surface (m) |
|-----------|---------------------------------------|--------------------------|---|
| FIRW-33A | 2.59 | 0.67 | 1.92 |
| FIRW-33B | 2.46 | 0.58 | 1.88 |
| FIRW-76 | 0.79 | 0.70 | 0.09 |
| FIRW-95 | 0.69 | 0.70 | +0.01* |
| TJA-25 | 2.21 | 0.76 | 1.45 |
| TJA-30A | 2.31 | 0.83 | 1.48 |
| TJA-30B | 2.29 | 0.61 | 1.68 |
| TJA-75 | Artesian | 0.69 | Est. +1.5 |
| TJA-100 | Artesian | 0.63 | Est. +1.75 |
| FIRE-33 | 2.26 | 0.56 | 1.70 |
| FIRE-75 | Artesian | 0.81 | Est. +2.25 |
| FIRE-100 | Artesian | 0.69 | Est. +2.5 |

* Artesian, (+) value is above level of the ground surface

4.3. Hydrologic Properties of the S₁ and S₂ Units Under FIR

In this section analysis methods and results from each type of test are described sequentially: single-well steady-state tests (Section 4.3.1); single-well pressure transient tests (Section 4.3.2); and multiple-well pressure interference tests (Section 4.3.3). In Section 4.4. results from all tests are compared and summarized.

4.3.1. Single-Well Steady-State Pumping Tests

Single-well steady-state pumping tests were conducted by the procedure described in Section 4.1.2. Measured waterlevel drawdowns and pumping rates are provided in Table

4.4. Productivity indices, defined as the pumping rate divided by the drawdown, are also provided in Table 4.4.

Permeability values (k) and hydraulic conductivities (K) were calculated using Thiem's equation (Todd, 1980)

$$k = \frac{q\mu}{2\pi\Delta h\rho gH} \ln \frac{r_e}{r_w} \quad [\text{Eq. 4.2}]$$

where q is the pumping rate, μ is the viscosity of water (10^{-3} Pa-s), Δh is the waterlevel drawdown, ρ is the density of water (1000 kg/m^3), g is the gravitational constant (9.81 m/s^2), H is the thickness of the screened interval, r_e is the radius of hydraulic influence and r_w is the well radius. The radius of influence (r_e) can be approximated by (Earlougher, 1977)

$$r_e = 2 \left(\frac{kt}{\phi\mu c_t} \right)^{1/2} \quad [\text{Eq. 4.3}]$$

where t is the time since pumping began (3600 s for these tests), ϕ is the formation porosity and c_t is the formation compressibility. Calculated averages for k and ϕc_t obtained from the pressure transient tests described in Sections 4.3.2 and 4.3.3 were used to estimate r_e for the steady-state analysis with Eq. 4.3. Using these averages, r_e is estimated to be 64 m. Fortunately, even if r_e is over or under estimated by as much as $\pm 50\%$, only $\pm 10\%$ error is introduced into the calculated permeability or hydraulic conductivity values. This is small in comparison to other errors introduced into permeability values calculated by this type of test. Section 4.3.4 provides additional discussion of the accuracy of reported permeability values calculated from steady-state pumping tests.

The hydraulic conductivity is directly related to the formation permeability by

$$K = \frac{k\rho g}{\mu} \quad [\text{Eq. 4.4}]$$

where all of the terms have been defined previously. Permeability values and hydraulic conductivities calculated using this procedure are listed in Table 4.4.

Table 4.4. Drawdown and pumping rate data from the single-well steady-state pumping tests. Calculated values of the formation permeability (k) and the hydraulic conductivity (K) are also provided.

| Well Name | Drawdown (m) | Pumping Rate (m ³ /s) | Productivity Index (m ³ /s/m) | Permeability k (m ²) | Hydraulic Conductivity K (m/s) |
|-----------|--------------|----------------------------------|--|----------------------------------|--------------------------------|
| FIRW-33A | 1.70 | 2.7×10^{-4} | 1.5×10^{-4} | 1.2×10^{-12} | 1.1×10^{-5} |
| FIRW-33B | 1.35 | 3.2×10^{-4} | 2.3×10^{-4} | 1.8×10^{-12} | 1.8×10^{-5} |
| FIRW-76 | na | na | na | na | na |
| FIRW-95 | 1.90 | 3.0×10^{-4} | 1.6×10^{-4} | 1.2×10^{-12} | 1.2×10^{-5} |
| TJA-25 | 1.83 | 2.7×10^{-4} | 1.5×10^{-4} | 1.9×10^{-12} | 1.9×10^{-5} |
| TJA-30A | 1.65 | 3.2×10^{-4} | 1.9×10^{-4} | 1.8×10^{-12} | 1.8×10^{-5} |
| TJA-30B | 1.67 | 5.1×10^{-4} | 3.1×10^{-4} | 2.3×10^{-12} | 2.3×10^{-5} |
| TJA-75 | 5.08 | 1.1×10^{-4} | 2.1×10^{-5} | 2.7×10^{-13} | 2.6×10^{-6} |
| TJA-100 | 5.39 | 2.9×10^{-4} | 5.4×10^{-5} | 6.9×10^{-13} | 6.7×10^{-6} |
| FIRE-33 | 2.16 | 1.1×10^{-4} | 5.0×10^{-5} | 3.8×10^{-13} | 3.7×10^{-6} |
| FIRE-75 | 4.64 | 8.2×10^{-4} | 1.8×10^{-4} | 1.7×10^{-12} | 1.7×10^{-5} |
| FIRE-100 | 6.23 | 3.0×10^{-4} | 4.3×10^{-5} | 5.6×10^{-13} | 5.4×10^{-6} |

na - not available

Productivity indices for the wells range from 2.1×10^{-5} to 3.1×10^{-4} (m³/s)/m (0.1 to 1.5 gallons per minute per foot of drawdown). In the Fallon area, Glancy (1982) reported a range of 0.5 to 12 (gal/min)/ft. for wells screened in the shallow part of the intermediate-depth alluvial aquifer. Values measured at FIR fall at the low end of this range. The average productivity index for wells screened in the S₁ unit (1.8×10^{-4} (m³/s)/m or 0.87 (gal/min)/ft.) is about twice as high as the average value (9.1×10^{-5} (m³/s)/m or 0.44 (gal/min)/ft.) measured in the S₂ unit.

Hydraulic conductivity values range from 3.7×10^{-6} to 2.3×10^{-5} m/s (1 to 6.5 ft/day). These are typical values for fine sands and sand, silt and clay mixtures (USBR, 1977). The average hydraulic conductivity of the S₁ unit is 1.5×10^{-5} m/s (4.3 ft/day). This is nearly double the average value of 8.6×10^{-6} m/s (2.4 ft/day) measured in the S₂ unit.

4.3.2. Single-Well Pressure Transient Pumping Tests

Single-well pressure transient pumping tests were conducted in 6 wells. Of these, 4 were in the S₁ unit and 2 in the S₂ unit. Procedures for carrying out these tests were described in Section 4.1.2. Advantages of pressure transient tests over single-well steady-state tests include the following:

- Damage to the formation (caused by drilling) can be detected;
- Heterogeneities in the formation, such as flow barriers or faults can be detected;
- A larger volume of the formation is influenced by these tests; and
- Because of all the above, they provide more accurate hydraulic conductivity data.

Due to the relatively long time these tests take to set up and conduct, they were carried out only on a limited number of wells. Comparison between the transient and steady-state tests provides for an independent check on the validity of the steady-state tests. This comparison is provided in Section 4.3.4.

Test data were analyzed using conventional semi-log pressure drawdown or buildup plots (Earlougher, 1977). A graph of pressure versus the logarithm of time since pumping began (Δt) is prepared. If the formation behaves like a homogeneous aquifer, a straight line is observed on the graph. The slope of this straight line (m) is related to the formation permeability by

$$k = 0.183 \frac{Q\mu}{m} \quad [\text{Eq. 4.5}]$$

Note that all these equations are in Standard International Units. If the pressure buildup data are analyzed instead of the pressure drawdown data, the quantity $(t+\Delta t)/\Delta t$ is substituted for Δt , where t is the length of time the well was pumped before it was shut off and Δt is the time since shutting of the pump. The skin factor of the well (s), an indicator of formation damage (e.g., plugging due to drilling), is calculated by

$$s = 1.151 \left(\frac{\Delta p_{1s}}{m} - \log \left(\frac{k}{\phi \mu c_t r_w^2} \right) - 0.351 \right) \quad [\text{Eq. 4.6}]$$

where Δp_{1s} is the pressure drawdown on the semi-log straight line when Δt is one second.

Semi-log graphs of the pressure transient tests are shown in Figs. 4.15 to 4.19. In each case, a well-defined semi-log straight line is present, indicating that over the test period, the formation is behaving like a homogeneous aquifer. Table 4.5 lists the slopes of these straight lines, the values for Δp_{1s} and pumping rates. Calculated permeabilities, hydraulic conductivities and skin factors are also listed in Table 4.5.

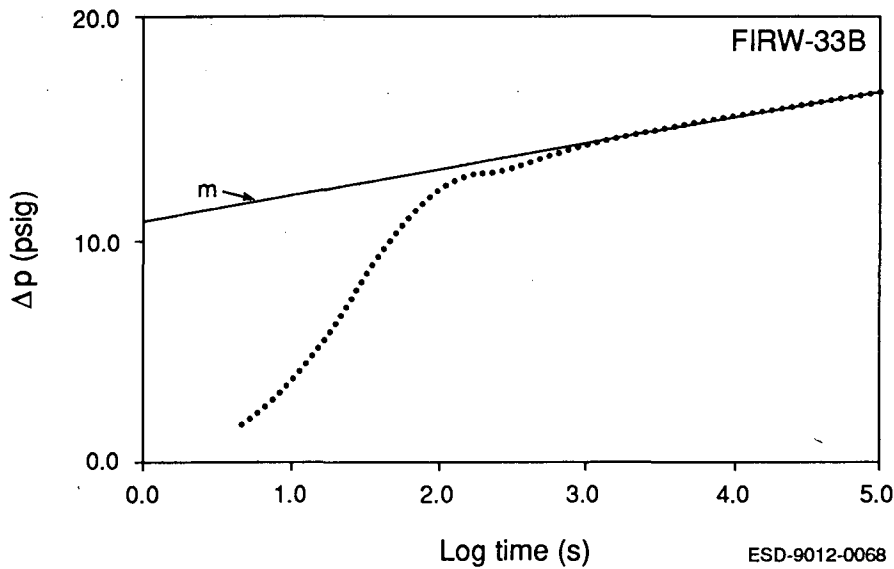


Figure 4.15. Semi-log plot of the pressure drawdown data from FIRW-33B.

Table 4.5. Data and analyses of the pressure transient pumping tests conducted at FIR.

| Well | Pumping Rate Q (m ³ /s) | Slope m (Pa/cycle) | Δp_{1s} (Pa) | k (m ²) | K (m/s) | s |
|----------|------------------------------------|--------------------|----------------------|-----------------------|----------------------|------|
| FIRW-33B | 2.1×10^{-3} | 8619 | 7.40×10^4 | 3.0×10^{-12} | 2.9×10^{-5} | 6.6 |
| FIRW-76 | 2.2×10^{-3} | 4309 | 7.68×10^4 | 4.3×10^{-12} | 4.3×10^{-5} | 17.0 |
| TJA-25 | 6.9×10^{-4} | 4750 | 4.09×10^4 | 2.9×10^{-12} | 2.9×10^{-5} | 6.6 |
| TJA-30B | 2.1×10^{-3} | 9048 | 4.81×10^4 | 2.8×10^{-12} | 2.8×10^{-5} | 2.8 |
| TJA-100 | 8.2×10^{-4} | 7614 | 1.15×10^5 | 2.2×10^{-12} | 2.1×10^{-5} | 14.2 |
| FIRE-33 | 1.3×10^{-4} | 4324 | 7.27×10^4 | 3.5×10^{-13} | 3.4×10^{-6} | 17.1 |

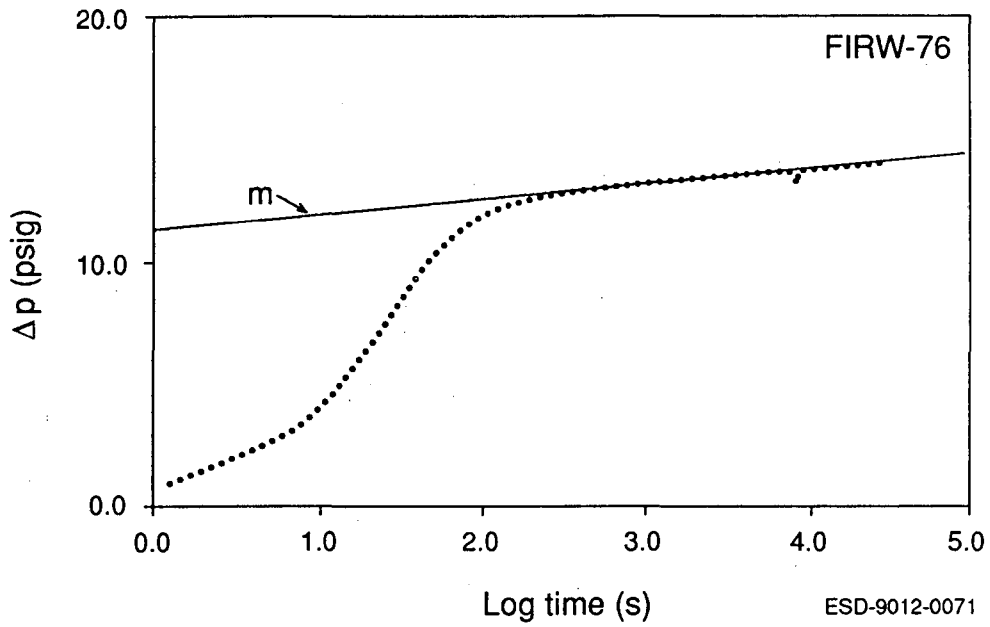


Figure 4.16. Semi-log plot of the pressure drawdown data from FIRW-76.

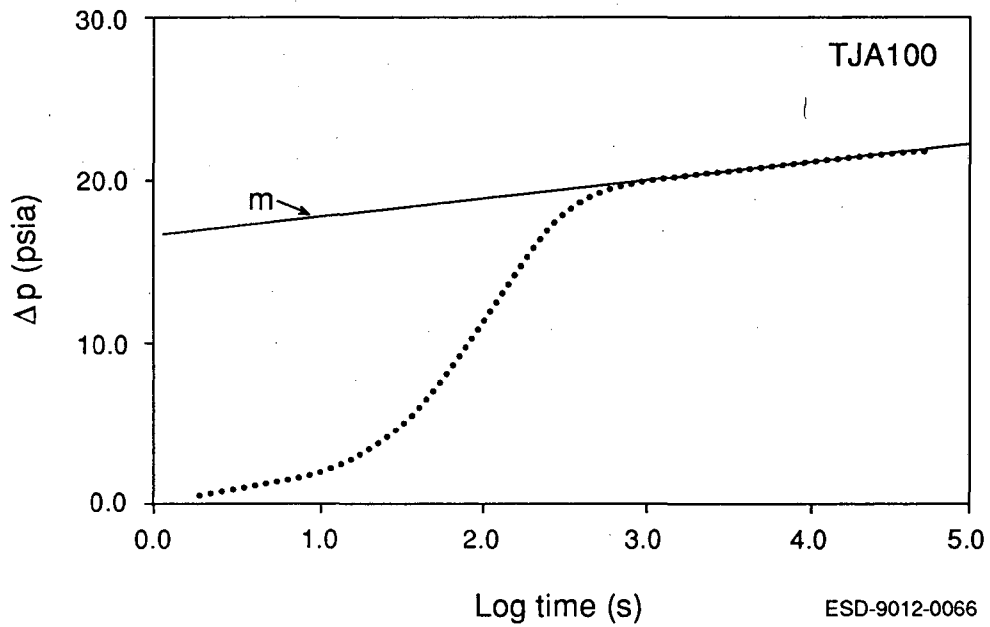


Figure 4.17. Semi-log plot of the pressure drawdown data from TJA-100.

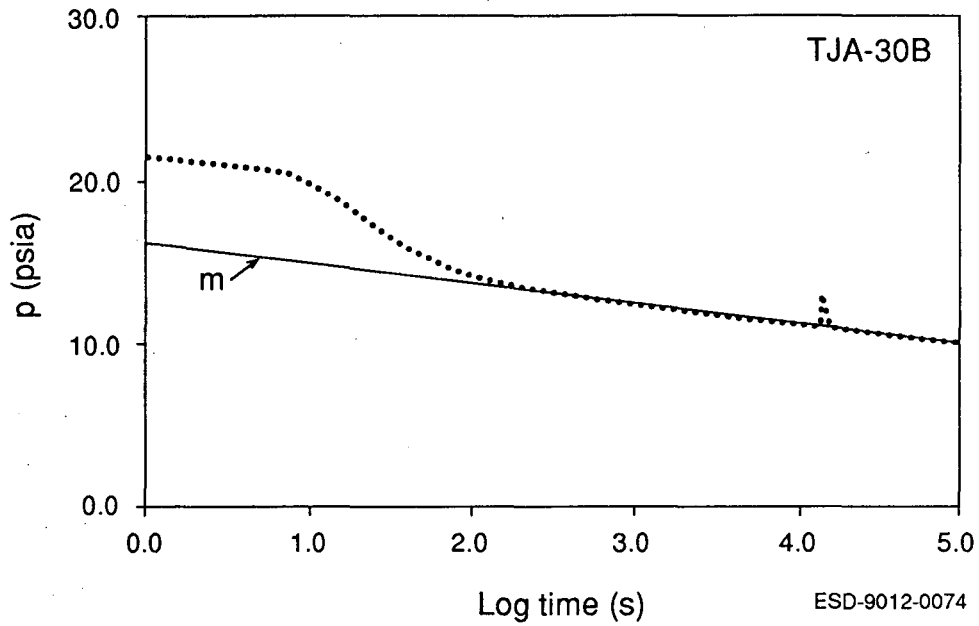


Figure 4.18. Semi-log plot of the pressure drawdown data from TJA-30B.

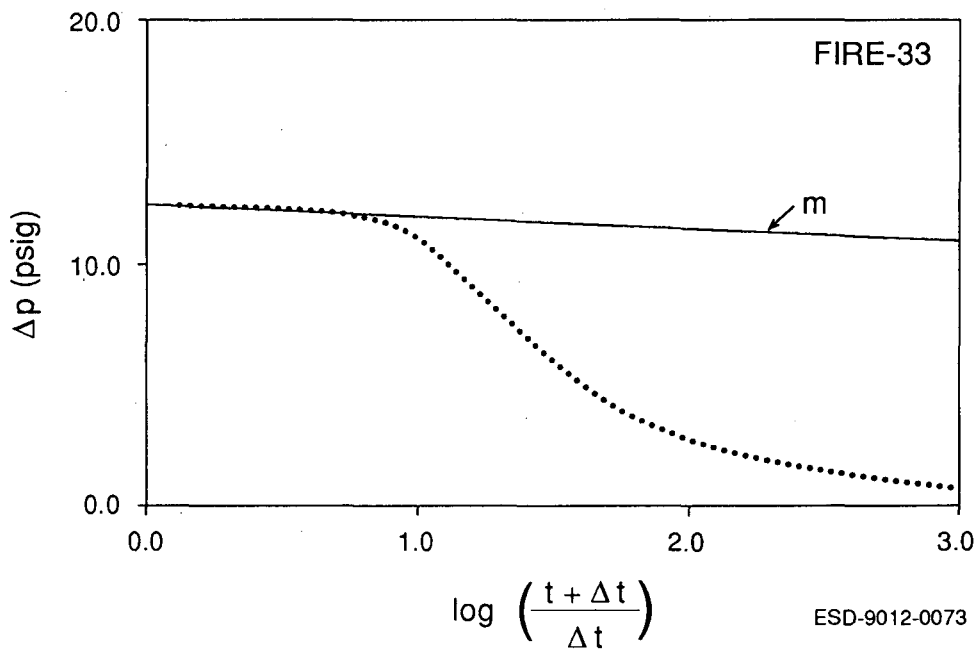


Figure 4.19. Semi-log plot of the pressure drawdown data from FIRE-33.

With the exception of FIRE-33, hydraulic conductivities calculated from these tests fall in the remarkably small range of 2.1×10^{-5} to 4.3×10^{-5} m/s (6 to 12 ft/day). The value of 3.4×10^{-6} m/s measured in FIRE-33 is significantly lower. The average hydraulic conductivity for the 4 wells tested in the S₁ unit is 2.0×10^{-5} m/s (6.3 ft/day). Because only two wells were tested in the S₂ unit, the average is not meaningful and in fact, may be misleading because these tests were carried out in the wells with the highest productivity indexes.

The large skin factors shown in Table 4.5 indicate that flow from the formation into the wells is impaired in all these wells. Skin factors range from 2 to 17. A well without any damage should have a skin factor of zero (van Everdingen, 1953). Wells with skin factors in the range of 2 to 5 are considered to be moderately damaged. Wells with skin factors greater than 20 are severely damaged. The source of these large skin factors is puzzling. All wells were gravel-packed with 8-mesh sand, which is many times more permeable than the formation sands. Consequently, these wells are expected to have a negative skin factor, indicating a higher permeability near the wellbore. Extreme care was taken to emplace the gravel pack, as we were immediately aware of the large skin factors in these wells. Moreover, wells were vigorously developed by air-lifting, another step that should contribute to removing any drilling-associated formation damage.

While it is possible that the formation was plugged by the drilling and well completion process, an alternative explanation for the large positive skin factors is also plausible. The presence of thin and laterally discontinuous clay lenses may limit vertical groundwater flow in the immediate vicinity of the well. With increasing distance from the well, it is likely that groundwater can flow circuitously through the entire thickness of the aquifer. This will create the same effect as wellbore damage. Near the well the formation appears to have relatively low permeability, compared to farther away from the well. This will create an apparent "skin effect" For example, the skin factor of a well can be estimated by

$$s = \left(\frac{k}{k_s} - 1 \right) \ln \left(\frac{r_s}{r_w} \right) \quad [\text{Eq. 4.7}]$$

where k_s is the permeability of the skin-damaged region and r_s is the radial distance that it extends from the well (Hawkins, 1956). Assuming that the region near the well has an effective permeability one half that of the aquifer and that this region extends 10 m from the well, the well would have a skin factor of 4.3. This is in the range of values observed in FIR wells. Although this explanation is not clearly superior to the more conventional one, it does provide an alternative. The ability to complete wells with minimal skin damage will be extremely important if reinjection is a serious option for drainage water disposal.

4.3.3. Multiple-Well Pressure Interference Tests

Two multiple-well pressure interference tests were conducted in the S_1 unit, one at the FIRW site and another at the TJA site. Procedures for conducting these tests were described in Section 4.1.2.

Pressure drawdown data from the interference tests were analyzed using conventional log-log type curve matching (Earlougher, 1977). Measured values of $\log(\Delta p)$ are plotted as a function of $\log(\Delta t)$, where Δp is the pressure drawdown in the observation well and Δt is time since the pump was turned on. This plot is then compared to theoretical curves, in this case, to the Theis (1935) equation. The Theis equation describes the pressure drawdown in a infinite confined aquifer of infinite extent and constant thickness, with wells that fully penetrate the aquifer. Comparison between the theoretical and measured drawdown curves is used to calculate the permeability (or hydraulic conductivity) and the storage coefficient of the aquifer.

A log-log plot of pressure drawdown at FIRW-33A caused by pumping FIRW-33B at $2.1 \times 10^{-3} \text{ m}^3/\text{s}$ (34 gal/min) is shown in Fig. 4.20. FIRW-33A is 39.7 m from the pumping well. Over the one-day test period, the pressure in the aquifer declined by the equivalent pressure of 0.8 m of water (1.2 psig). Note that reported pressures are in units of psig. The shape of the drawdown curve is typical of the drawdown in the Theis-type aquifer described above. The match between the theoretical curve and measured data is

shown in Fig. 4.21. Calculated values for the hydraulic conductivity and storage coefficient are given in Table 4.6.

Table 4.6. Calculated hydraulic conductivities and storage coefficients from the multiple-well interference tests.

| Observation Well | k (m ²) | K (m/s) | Storage Coefficient (m/Pa)* |
|------------------|------------------------|----------------------|--------------------------------|
| FIRW-33A | 5.0×10^{-12} | 4.9×10^{-5} | 1.2×10^{-7} |
| TJA-25 | 4.5×10^{-12} | 4.4×10^{-5} | 3.9×10^{-8} |
| TJA-30A | 4.5×10^{-12} | 4.4×10^{-5} | 3.9×10^{-8} |

* Storage coefficient is defined as $\phi c_t H$, where H is 15 m for the S₁ unit

At the TJA site, well TJA-30B was pumped at a rate of 2.1×10^{-3} m³ /s (33.5 gal/min) while pressure drawdown was monitored in TJA-25 and TJA-30A. Fig. 4.3 shows a plan view of wells at this site. TJA-25 and TJA-30A are 205 m and 396 m from TJA-30B, respectively. Pressure drawdowns at both wells are shown in Fig. 4.22. During the one-day test, pressures dropped by an equivalent of 0.6 and 0.2 m of water, in TJA-25 and TJA-30A, respectively. Note that in Fig. 4.22, $\log(\Delta p)$ is plotted as a function of $\log(\Delta t/r^2)$. The close match between these two data sets indicates that the aquifer is fairly homogeneous. The departure of these two curves near the end of the test may indicate leakage from the confining clay layers, C₁ and C₂. However, without additional supporting evidence this conclusion is premature. A single theoretical curve matches both data sets, as shown in Fig. 4.23. Calculated values for the hydraulic conductivity and storage coefficient are given in Table 4.6.

4.3.4. Summary of the Hydrologic Testing Program

The combination of geological and hydrological data provide a coherent description of the groundwater system under FIR that is consistent with regional studies published by Morrison (1964), Morgan (1982) and Glancy (1986). The two sandy units, S₁ and S₂, identified by the drilling program form moderately permeable aquifers. The S₁ unit is correlated with the Wyemaha formation of Morrison (1964) and is an important source of

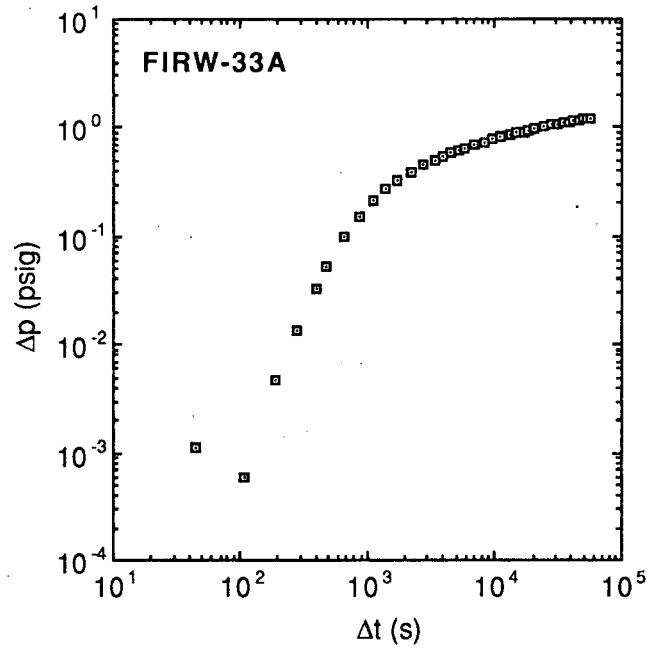


Figure 4.20. Log-log plot of the pressure interference data for FIRW-33A. Well FIRW-33B was pumped at a rate of $2.1 \times 10^{-3} \text{ m}^3/\text{s}$ (34 gal/min) during this time. See Fig. 4.2 for a plan-view of the test site.

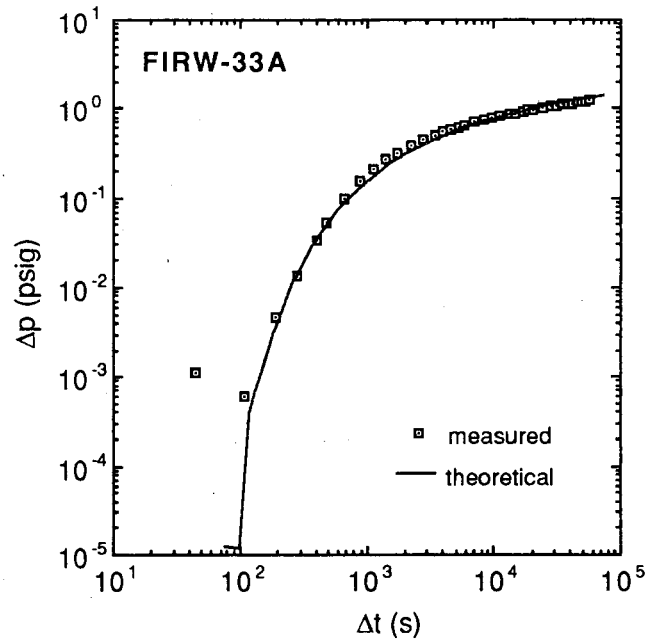


Figure 4.21. Type-curve match of the FIRW-33A pressure interference data.

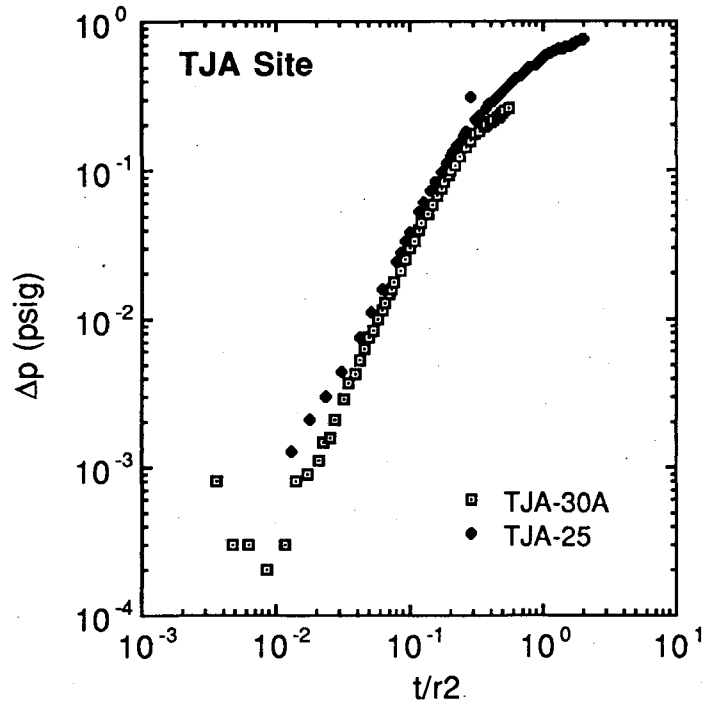


Figure 4.22. Log-log plot of the pressure interference data for TJA-25 and TJA-30A. Note that the data are plotted as a function of t/r^2 , where r is the distance between the pumping well and the observation wells. Well TJA-30B was pumped at a rate of $2.1 \times 10^{-3} \text{ m}^3/\text{s}$ during this time (33.5 gal/min). See Fig. 4.3 for a plan-view of the test site and inter-well distances.

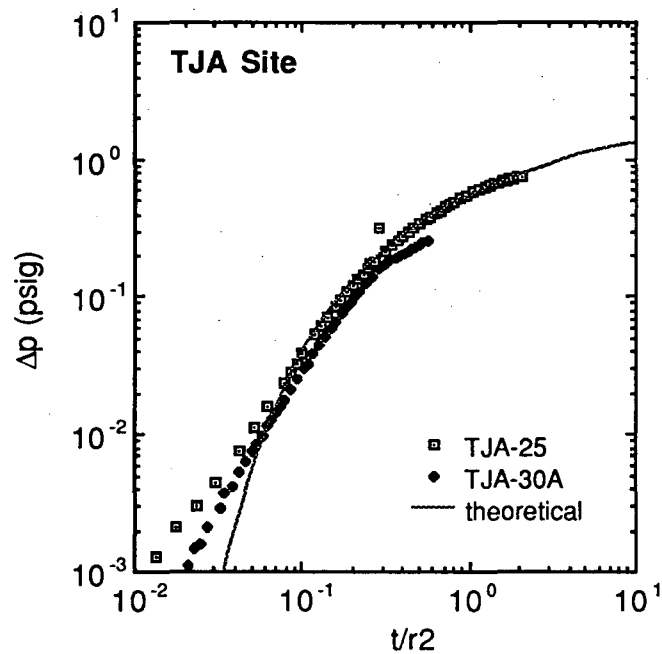


Figure 4.23. Type-curve match for the pressure interference data from wells TJA-25 and TJA-30A.

drinking water in the City of Fallon (Glancy, 1986). The S₂ unit is tentatively correlated with the Paiute formation of Morrison (1964) and is not a primary source of groundwater in the Fallon area (Glancy, 1986). The massive clay units, C₁ and C₂, are aquitards that limit vertical flow between the S₁ and S₂ units. Moreover, they limit transport of deep groundwater to the shallow drainage system.

The hydraulic conductivities measured in these two formations are typical of fine sand and mixtures of sand, silt and clay. Measured values range from 2.6×10^{-6} to 4.9×10^{-5} m/s (0.7 to 14 ft/day). Overall, there is good agreement between results from the three different methods used to calculate hydraulic conductivity. Table 4.7 lists average hydraulic conductivity values for the S₁ and S₂ units, as measured with the 3 methods described in Sections 4.3.1 to 4.3.3. Average values for the S₁ unit range from 1.5×10^{-5} to 4.7×10^{-5} m/s (4.2 to 13 ft/day). The average value measured from the steady-state tests in the S₂ unit is 8.6×10^{-6} m/s (2.4 ft/day).

The pressure transient tests generally yield higher estimates of hydraulic conductivity than the other tests. This is readily explained by the large positive skin factors measured in the single-well transient tests. All the single-well pressure transient pumping tests indicate that the hydraulic conductivity is lower near the wellbore than in the formation as a whole. Steady-state tests can not differentiate between near-well and formation properties and, in fact, steady-state measurements are strongly biased towards the near-well conductivity. Measurements from pressure transient tests are biased towards the average conductivity of the formation. In spite of the systematic trend of increasing conductivity with larger volumes of investigation, the range of conductivities is still remarkably small. This, taken together with Theis-like shape of the pressure interference data suggest that, on-average, the S₁ unit is reasonably homogeneous. In the immediate vicinity of the well, flow is impaired by formation damage or the presence of many thin and laterally discontinuous layers of silt or clay.

Hydraulic conductivity of the S₁ unit is about twice that of the S₂ unit. This is consistent with the geological characteristics of the two units. The S₁ unit is predominantly sandy. The S₂ unit consists of interbedded sands, silts and clays; only about 50% of this unit is sandy. Local discrepancies from this overall pattern are likely, as indicated by the relatively high conductivity measured at FIRW-76 (see Table 4.5).

Table 4.7. Comparison of average hydraulic conductivity values measured with the single-well steady-state pumping tests, the pressure transient pumping tests and the multiple-well pressure interference tests.

| Unit | Single-Well Steady State Tests K (m/s) | Single-Well Pressure Transient Tests K (m/s) | Multiple-Well Interference Test K (m/s) |
|----------------|---|---|--|
| S ₁ | 1.5×10^{-5} | 2.0×10^{-5} | 4.7×10^{-5} |
| S ₂ | 8.6×10^{-6} | nc | na |

nc - not calculated, too few samples for a meaningful average
na - not available

4.4. Groundwater Quality at the FIR Test Sites

Groundwater samples were collected from all wells drilled for this program. Sampling and analysis methods were described in Section 4.1.3. Results are reported in two sections; Section 4.4.1 discusses major ion and electrical conductivities of the samples and Section 4.4.2. discusses trace element concentrations. Groundwater from all wells at each site is poorer than standards for drinking water supply and surface water quality goals for protection of aquatic life.

4.4.1. Major Ion Composition and Electrical Conductivity of FIR Groundwater

Major ions and electrical conductivities in groundwater from the FIRW, TJA and FIRE sites are given in Table 4.8. As indicated by the electrical conductivities (EC's), groundwater under FIR is brackish. EC's are in the range observed in the TJ drain and shallow groundwater and are indicative of saline conditions (see Chapter 2, CH2M Hill,

1989; USBR, 1987; and Hoffman *et al.*, 1990). The salinity (in mg/L, estimated as $640 \times$ EC (dS/m); Logan, 1961) alone makes this water unfit for beneficial use as a drinking water supply, for livestock watering or irrigation.

Table 4.8. Major ion concentrations in groundwater samples from the FIRW, TJA and FIRE sites. Samples were collected between May and September, 1990.

| Well | EC (dS/m) | Cl (mg/L) | S (mg/L) | Na (mg/L) | Ca (mg/L) | Mg (mg/L) |
|----------|--------------|--------------|-------------|--------------|--------------|--------------|
| FIRW-33A | 15.1 | 5512 | 71 | 3274 | 18 | 33 |
| FIRW-33B | 15.7 | 5660 | 64 | 3217 | 21 | 35 |
| FIRW-76 | 5.8 | 1890 | 84 | 1150 | 13 | 7 |
| FIRW-95 | 4.3 | 1312 | 70 | 818 | 6 | 2 |
| TJA-25 | 21.2 | 7343 | 40 | 4566 | 40 | 73 |
| TJA-30A | 22.5 | 8120 | 52 | 4634 | 33 | 79 |
| TJA-30B | 19.9 | 7140 | 32 | 4201 | 30 | 57 |
| TJA-75 | 9.3 | 3395 | 9 | 1901 | 21 | 14 |
| TJA-100 | 8.4 | na | 1 | 1672 | 15 | 15 |
| FIRE-33 | 8.9 | 2674 | 87 | 1608 | 19 | 47 |
| FIRE-75 | 7.4 | 2608 | 5 | 1496 | 22 | 13 |
| FIRE-100 | 7.2 | 2747 | 23 | 1443 | 45 | 5 |

The ionic composition of groundwaters in the S₁ and S₂ units is dominated by sodium and chloride. Sulfur, Ca and Mg are only present only in minor amounts. Chloride concentrations in the S₁ unit range from 2,674 to 8,120 mg/L and the average value is 6,100 mg/L. Within the S₂ unit, Cl concentrations are somewhat lower and range from 1,312 to 2,747 mg/L. The average value is 2,390 mg/L. The lower Cl concentrations and EC's in the S₂ unit, compared to the S₁ unit, are consistent with the bulk-conductivity data measured with the EM39 conductivity log (see Section 4.2.2). Sulfur concentrations are low in both units. Groundwater in the Carson Desert above the C1 clay has considerably higher concentrations of S (Welch *et al.*, 1990; Lico *et al.*, 1987; USBR, 1987; Hoffman *et al.*, 1990). Cations are dominated by Na in both the S₁ and S₂ units. Concentrations in the S₁ unit range from 1,608 to 4,634 mg/L; the average value is 3583

mg/L. Sodium concentrations in the S₂ unit range from 818 to 1,901 mg/L; the average is 1,413 mg/L

Calcium and Mg are present only in minor amounts in both the S₁ and S₂ units. Groundwater in the S₁ and S₂ unit is distinct from the shallow groundwater above the C₁ clay, insofar as shallow groundwaters also have relatively high concentrations of Ca and Mg (Welch *et al.*, 1990; Lico *et al.*, 1987; USBR, 1987; Hoffman *et al.*, 1990).

4.4.2. Selenium, Arsenic, Boron and Lithium Concentrations in FIR Groundwater

Selenium, As, B, and Li concentrations were measured in all samples collected from the groundwater study sites. Additional trace elements analyses were also performed but are not reported here because concentrations were below detection limits or not believed to present significant environmental risks (based on work of others in the Stillwater Wildlife Management Area, e.g., Hoffman *et al.*, 1990). Samples were analyzed using the procedures described in Section 4.1.3. Table 4.9 lists measured concentrations of these constituents in the groundwater samples. Note that B concentrations are listed in mg/L, while the remaining data are in µg/L.

Selenium concentrations in all samples are low (≤ 2 µg/L). These levels are below even the most stringent values recommended for protection of aquatic life (2 to 5 µg/L; Lemly and Smith, 1987) and far below the State of Nevada's 50 µg/L criterion for propagation of wildlife.

Arsenic concentrations range from 3 to 220 µg/L. These are in the range of values reported for drainage water (see Chapter 2, this report; USBR, 1987; and Hoffman *et al.*, 1990). These concentrations are also typical of groundwaters in the Carson Desert (Glancy, 1986; Welch *et al.*, 1990; and Hoffman *et al.*, 1990) and exceed the drinking water standard of 50 µg/L. The State of Nevada's criterion for protection of aquatic life (40 µg/L) is also exceeded by groundwater from the majority of these wells. Arsenic concen-

trations in groundwater at the TJA and FIRE sites decreases with depth. The average concentration of As in the S₂ unit is 23 µg/L, compared to 196 µg/L in the S₁ unit.

Table 4.9. Trace element concentrations in groundwater samples from the FIRW, TJA and FIRE sites. Samples were collected between May and September, 1990.

| Well | Se (µg/L) | As (µg/L) | B (mg/L) | Li (µg/L) |
|----------|--------------|--------------|-------------|--------------|
| FIRW-33A | 1.1 | 150 | 17 | 160 |
| FIRW-33B | 1.6 | 174 | 15 | 130 |
| FIRW-76 | 0.3 | 147 | 5 | 70 |
| FIRW-95 | 0.3 | 187 | 4 | 50 |
| TJA-25 | 2.1 | 202 | 21 | 240 |
| TJA-30A | 0.8 | 204 | 21 | 270 |
| TJA-30B | 0.6 | 220 | 20 | 270 |
| TJA-75 | 0.2 | 33 | 15 | 110 |
| TJA-100 | 0.8 | 6 | 19 | 170 |
| FIRE-33 | 0.8 | 160 | 16 | 260 |
| FIRE-75 | 0.3 | 20 | 16 | 310 |
| FIRE-100 | 0.2 | 3 | 14 | 1200 |

Boron concentrations are also high, ranging from 4 to 21 mg/L. Again, these are in the range of values reported for drainage water and shallow groundwater in the Carson Desert (Chapter 2, this report; Lico *et al.*, 1987; USBR, 1987; and Hoffman *et al.*, 1990). These values are far in excess of the 0.2 mg/L effect level of B on fish and the irrigation water criterion of 1 mg/L.

Lithium concentrations are reported in the last column of Table 4.9. These values are low in comparison to most values reported for the shallow groundwater system (USBR, 1987; Hoffman *et al.*, 1990). Relevant water quality criterion for lithium are not available. Toxicity studies conducted by Finger *et al.* (1989) suggested a possible correlation between elevated lithium concentrations and mortality of several aquatic indicator organisms.

The combination of high salinity and elevated As and B concentrations in FIR groundwater severely limit, if not preclude, conventional beneficial use of this resource. Although this investigation is not exhaustive, the preponderance of data suggest that good quality water will not be found in the S₁ and S₂ aquifers. Isolated pockets of groundwater marginally suitable for limited beneficial use may be found, such as at the FIRE site

4.5. Implications for Drainage Water Management at FIR

Motivations for the groundwater investigations described here were two-fold. First, additional information on flow of shallow groundwater to the TJ and nearby surface drains was needed. (Section 4.5.1. discusses the volume of groundwater likely to discharge into shallow drains over the coming decades.) Second, the aquifers under FIR may be suitable for reinjection of drainage water. Preliminary evaluation of the feasibility of reinjecting drainage water into the S₁ or S₂ unit is discussed in Section 4.5.2.

4.5.1. Groundwater as a Source of Salts and Trace Elements to Shallow Drainage

The massive nature of the C₁ clay limits the volume of groundwater that is likely to discharge into a shallow drainage system at FIR. Only the shallow groundwater currently above the C₁ unit will make its way into drainage systems tapping the surface-most horizon. With this in mind, the volume of saline groundwater that may eventually be drained can be calculated from available information on the depth to the top of the C₁ unit, the depth to the watertable and an estimate of the porosity of the sand, silt and clay loam above the C₁ clay. Contour maps showing the "depth to barrier" (e.g., the C₁ clay) have been prepared for the USBR by CH2M Hill (1989). Similarly, maps showing depth to the watertable have been prepared. Estimates of porosity based on similar textured sediments can also be used to complete this calculation.

It is worth a moment to discuss the sources of error inherent to this simple approach for estimating the volume of poor quality groundwater tapped by shallow drains. First, it

neglects the volume of salts and trace elements residing in the vadose zone (region between the ground surface and the watertable). As described in Chapter 3, in newly irrigated field, large inventories of salts may still reside in the vadose zone and contribute to poor quality drainage. However, in fields irrigated and drained for longer periods, the vadose zone is at least partially leached and will contribute little to salt and trace element concentrations in shallow drains. This approach also neglects any contribution from lateral flow of groundwater into the FIR. Because of the extremely heterogeneous nature of the surface soils and sediments, this contribution should be relatively small in most areas of the Reservation; in particular, clay lenses act as barriers to lateral migration. In any event, lateral flow is most likely associated with leakage from surface canals and drainage ditches, both of which have higher quality water than the shallow groundwater. For the above reasons, this approach is reasonable for obtaining a rough estimate of the volume of poor quality groundwater above the C₁ unit.

Measured depths of the C₁ unit, as reported in CH2M Hill (1989) range from 2.4 to 6.1 m. The average depth is estimated to be mid-way between these two values, or 4.3 m. The depth to the watertable ranges from 1.2 to 3 m, with an average of about 2 m in October, 1985. The porosity (ϕ) of the sediments is estimated to be 35%. The volume (V) of water per unit area, above the C₁ unit and below the watertable is calculated by

$$V \text{ (m}^3\text{/m}^2\text{)} = \phi (D_{C_1} - D_w) \quad [\text{Eq. 4.7}]$$

where D_{C_1} is the depth to the C₁ clay and D_w is the depth to the watertable. Using the average values for these parameters listed above, the volume of groundwater per square meter of ground is 0.8 m³/m². This translates to 2.6 acre-ft/acre. For the estimated 1680 ha (4150 acres) of irrigated land at FIR, the total volume of water is about 1.4×10^7 m³ (11,000 acre-ft). For comparison, this is about 2.5 times the annual flow on the TJ Drain.

While this is an overly simplistic estimate of the volume of salt and trace element laden water tapped by shallow drains, it does provide a frame of reference for assessing drainage needs at FIR. This, together with shallow groundwater quality data collected

previously (USBR, 1987; CH2M Hill, 1989) and being analyzed currently (John Fields, USBR Sacramento, personal communication) can be used to estimate the quantity of salts and trace elements associated with drainage from shallow groundwater.

4.5.2. Potential for Shallow Reinjection of Drainage Water

Both the S₁ and S₂ units may be suitable aquifers for reinjection of drainage waters. Feasibility of this plan hinges on the follow factors:

- Total volume of drainage requiring disposal;
- Injection capacity of each well;
- Compatibility of drainage waters and groundwater in the S₁ or S₂ unit;
- Regulatory requirements regarding protection of beneficial uses of groundwater at FIR and in surrounding areas; and
- Costs associated with construction, maintenance and operation of a reinjection facility.

Each of these factors requires careful analysis before this option can be seriously considered. Nevertheless, sufficient information is available for a preliminary feasibility assessment. Each of the first 4 factors listed above is discussed in turn. Cost estimates for constructing and operating a reinjection facility are premature.

Total drainage volume requiring disposal through reinjection can be estimated from past data regarding flows in the TJ Drain. The USBR (1987) estimated annual flows of about 8×10^6 m³ (6500 acre-ft) in the TJ Drain. As discussed shortly, it is unlikely that this large a volume can be reinjected into either the S₁ or S₂ unit. However, mass balance calculations (USBR, 1987) and visual observations over the last year indicate that the majority of flow in the TJ Drain is derived from sources other than the shallow groundwater system, such as tailwater run-off and canal operational excess. If the subsurface drainage

component can be separated from these other sources, reinjection of the estimated 9×10^5 m³/year (724 acre-ft/year; USBR, 1987) of surface drainage may be feasible

If a buried drain system is installed at FIR, reinjection may also be considered for drainwater disposal. Discharge estimates for a buried drain system have been made by CH2M Hill (1989); estimated rates range from 1.6×10^{-6} to 4.6×10^{-6} m³/(s-m of drain) (1.7×10^{-5} to 5×10^{-5} ft³/s-ft of drain). Assuming 100-m drain spacing (330 ft), and that drains are installed on a total of 4.0×10^6 m² (1000 acres), estimated drainage discharges are about 2.35×10^6 m³/year (1900 acre-ft/year). These discharge estimates have not been reviewed by LBL with respect to newly available data on the soils and shallow groundwater system at FIR. Additional analysis of likely discharge rates from shallow buried drains is required to refine these estimates and evaluate the feasibility of reinjecting discharge from shallow tile drains. As mentioned in CH2M Hill (1989), separation of irrigation return flows from subsurface drainage has several benefits. Another benefit is drainage volume reduction; if subsurface drainage is not separated from return flows it is unlikely that reinjection can be used to dispose of the combined flows.

Reinjection rates for wells screened in the S₁ and S₂ units are estimated to average 1.9×10^{-3} m³/s (50 acre-ft/year per well). This estimate is contingent on the following factors: wells are completed with a casing diameter of at least 0.20 m (8 inches); the gravel pack is installed such that no mechanical plugging in the near-well region occurs; wells are screened over the entire thickness of either the S₁ or S₂ unit; and that mixing between drainage water and native groundwaters does not precipitate minerals or create biological plugging of the well-screen. At present, none of the factors present a major obstacle. Injection testing of wells drilled for the investigation described here will provide additional information on injection capacity of wells screened in the S₁ and S₂ units.

Regulatory requirements for reinjection of agricultural drainage water into the S₁ or S₂ unit must be developed through discussions with local, State and Federal agencies. Regulations regarding injection of waste waters, in general, are site-specific. Specific concerns that need to be addressed for reinjection of drainage water at FIR include the follow-

ing: degradation of local and regional groundwater resources; impairment of beneficial uses of local and regional groundwater; and potential influences of reinjection on nearby farmlands or wetlands, such as rising watertables or degradation of shallow groundwater or surface water quality. Because of poor water quality in the S₁ and S₂ units at FIR, reinjection of drainwater is unlikely to impair beneficial uses of FIR groundwater. However, since the S₁ unit is used for a number of purposes in the City of Fallon and surrounding areas, potential impacts of reinjection on these users will be required. The basic information is now available to provide an assessment of these concerns. If reinjection is pursued as an option for drainage water disposal, discussions with appropriate regulatory agencies and concerned parties should be initiated soon.

5. TRACE ELEMENTS IN CRUSTS AND SURFACE SOILS OF SEVERAL AREAS SURROUNDING THE FALLON INDIAN RESERVATION

In arid, closed-basin environments such as that of the Lahontan Valley and Carson Sink, high salinity and trace element concentrations are not characteristics limited to agricultural drain waters. The inventories of salts and trace elements in surface soils and sediments in non-irrigated lands of the region can also be substantial. The Carson Sink is perhaps the most obvious and extreme example. However, even in surrounding areas, especially those with relatively finer soil textures, local depressions subject to both periodic ponding and evaporation can become enriched with respect to salts and various trace elements. Within the SWMA, salt and trace element loading continues as a result of both human activities and natural processes.

Given the ubiquitous presence of elevated concentrations of salts and the plausibility of elevated concentrations of numerous trace elements in playa crusts and pond sediments of the area, quantification of potentially bioavailable inventories of these constituents is an important step towards understanding problems of trace element accumulation in the food chain. Such information is very useful in beginning to understand the dynamics of trace element cycling in at least two likely scenarios. In one scenario, rainfall or runoff water accumulate in usually dry playas such as the Carson Sink, dissolving surface salts and trace elements into the ponding waters. When such ponding is persistent enough to attract wildlife, information addressing ranges of expected concentrations of various solutes is useful in anticipating the further developments in such systems. This type of information may become even more relevant during the drying of such transient ponds, where evaporation is likely to concentrate certain dissolved trace elements to very high levels. Examples of such waters were described in Section 2.2.2. A second, equally probable scenario is that of a similar wetting and drying cycle taking place over a dry lake bed within the SWMA. In either case, marginal to poor water quality are likely outcomes even with the

application of high quality waters. A brief survey of the literature suggested that information in this area was limited.

Before proceeding to describing the work performed on Carson Sink and SWMA soils, the issue of bioavailability of trace elements warrants at least some discussion. Quantification of the bioavailable portion of the total elemental inventory of individual trace elements in soils and sediments is not a simple matter. It is at best naive to assume that the bioavailable inventory of any particular element is some fixed fraction of the total elemental concentration. This is especially the case with trace elements such as As, B, Mo and Se which may occur in wide varieties of solid phases, surface complexes and (with the exception of B) oxidation states. Furthermore, different species of plants and animals are known to have widely different rates of uptake of and tolerance for different trace elements. Mobility or bioavailability of particular trace elements is further complicated by the fact that the local soil/sediment environment is often not characterized by one particular redox state. Even in some cases where such simplifications are reasonable, oxidation states of certain trace elements may persist in disequilibrium with the measured redox environment for considerable periods of time.

The case of Se in Kesterson Reservoir soils illustrates this point. Presently, the major portion of the Kesterson Se inventory (> 85%) remains in various oxidation states lower than that of Se(VI) which would be the predicted stable form in the aerated vadose conditions which have prevailed for nearly 3 years. Our experience with Se at Kesterson Reservoir has also clearly demonstrated that total elemental analyses for Se in this environment provided a poor index for Se bioavailability. Uptake of Se in the food chain has been shown to be relatively insensitive to total Se concentrations in these soils. This lack of correlation between soil Se content and Se uptake by plants and animals is presently observable in upland environments. Total soil Se contents have also failed miserably as an indicator of Se bioavailability under ponded conditions at Kesterson Reservoir. At a test plot where the seleniferous surface soil was removed, leaving a "clean" residual soil surface, ponding from water table rise occurred. The surface water which formed over this "clean"

(< 3 mg Se/kg) soil persisted for several months at Se concentrations in excess of 1,000 µg/L. Such an example illustrates the questionable merit of the total elemental analyses approach in general.

The work to be described in following sections focuses in one way or another on water-soluble concentrations of various trace elements. It is recognized that the water-soluble fraction of any constituent is not necessarily an ideal indicator of its bioavailable fraction. Nevertheless, the water soluble fraction of any constituent is clearly a reasonable indicator for its partitioning into surface waters. It is further recognized that the “water-soluble” fraction is not often a precise quantity, especially with respect to trace elements. The chemical composition of any water-extract obtained from a soil is potentially subject to influences of particular redox conditions, dissolution and desorption kinetics, ionic strength effects and solubility limits. With these limitations in mind, the water-extract data are useful in providing estimates of readily mobile fractions of various elements.

In the interest of developing an understanding of the processes described above, two simple studies have been performed. The first involved measurements of water soluble inventories of various trace elements in a limited number of Carson Sink surface soils and SWMA pond bottom sediments. These results are described in Sections 5.1 and 5.3. In Section 5.2, the trace element data presented in Section 5.1 are used to provide ranges of possible surface water concentrations for these constituents using a simple mixing model. A second activity which has been completed involves measuring changes in water quality over time in initially “clean” waters ponded over dried SMWA sediments. This experiment is described in Section 5.4. Plans and materials for more definitive experiments are also being developed. These efforts are not aimed at providing extensive areal surveys of trace element inventories or salinity. Such studies have been and are being conducted by other groups. The work presented in this chapter focuses on mobile fractions of various trace elements in sediments and their potential recycling into ponded waters.

5.1. Soluble Trace Element Contents of Surface Soil Samples From the Carson Sink

Carson Sink surface soils were sampled at five locations. The general locations of the sampling sites are illustrated in Fig. 1.1. Site CS-D1 is located in the southern portion of the Sink, near the southern boundary of the Fallon National Wildlife Refuge. Site CS2 is located on the west side of the sink, 0.1 km east of Hwy 95 and 8.8 km south of Parran Station. Sites CS7, CS8 and CS9 are in the northern portion of the Sink. Site CS7 is 0.1 km south of the access road along the north rim of the Sink, 12.5 km southwest of the turnoff to Wildhorse Pass (West Humboldt Range). Site CS8 is 1.5 km south of CS7 and CS9 is 2.5 km south of CS7. Currently, only water-extracts and phosphate extracts have been performed on these samples. (Water-extraction and phosphate-extraction procedures are outlined in Appendix B.) The motivation for performing the phosphate extractions is also provided in Appendix B. Briefly, under certain conditions, phosphate-extracts provide a more effective means for bringing arsenate, borate, molybdate and selenite into solution. The difference between phosphate-extracted and water-extracted inventories of these anions is *assumed* by some to represent the adsorbed inventory of these constituents. The data presented in Fig. 5.1(a and b) summarize water-extracted and phosphate-extracted As, B, Mo and Se concentrations from the surface 0.10 m of playa soils at each site, except for sites CS7 and CS8 with represented 0-0.01 m surface crusts. Water soluble As concentrations range from 1 to 18 mg/(kg soil). The concentrations of As in the water extracts indicate that surface water As concentrations in excess of the 40 $\mu\text{g/L}$ criterion could occur during flooding of the playa. This projection will be discussed at the end of this section.

The water-extracted As is typically only a small fraction of the total soil As. Likewise, water-extracted B, Mo and Se are also commonly small fractions of their respective total elemental inventories. The water-extractable inventories do provide indications of soil environments where these trace element concentrations may be readily mobile, and consequently readily bioavailable. In the present work, water-extracted trace element concentrations are useful in screening for potentially problematic soils.

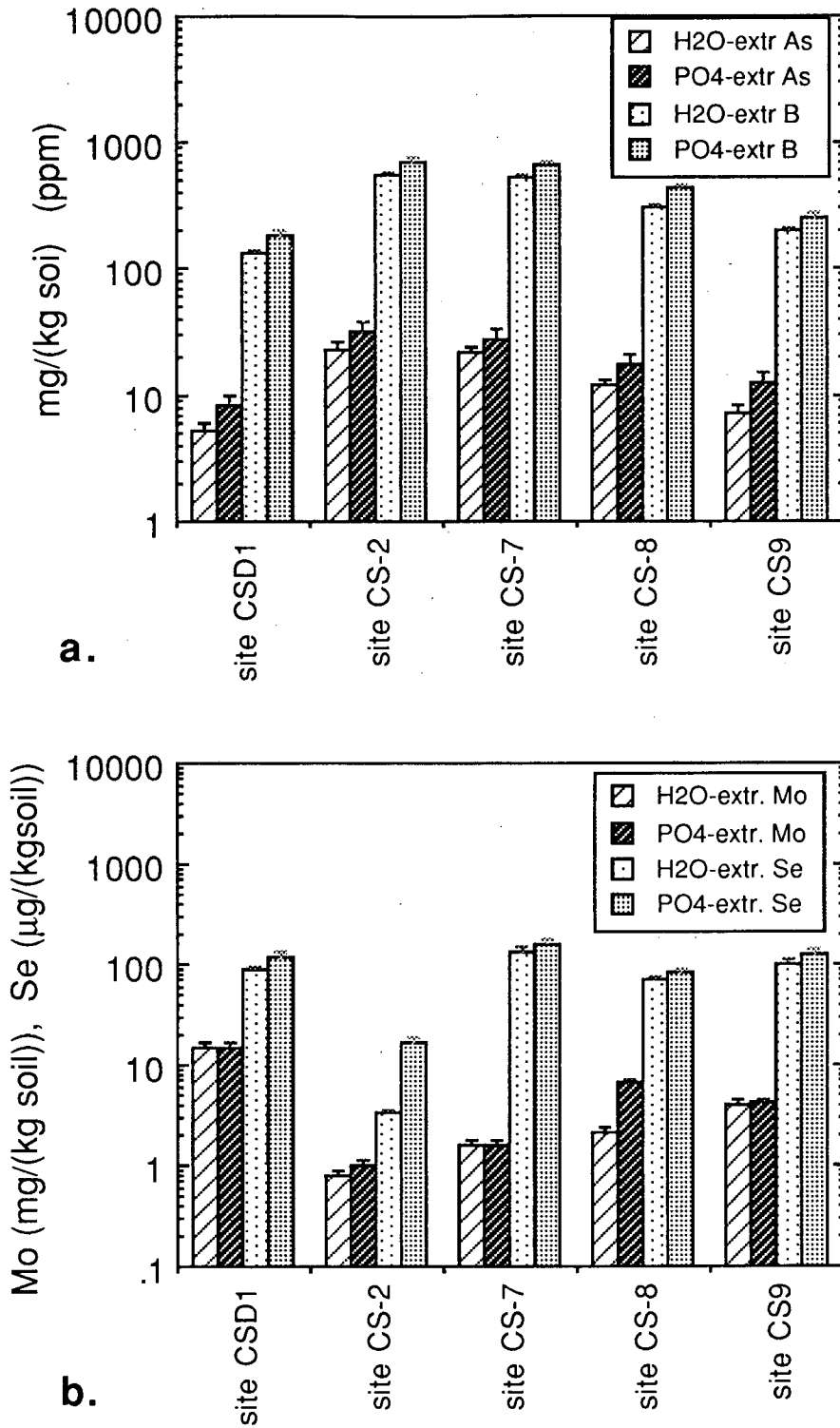


Figure 5.1. Water-extracted and phosphate extracted concentrations of (a) arsenic and boron and (b) Molybdenum and selenium in the surface-most 0.10 m soils (for sites CSD1, CS2 and CS9) and the uppermost 0.01 m of salt crusts (for sites CS7 and CS8). Note that selenium concentrations are in units of $\mu\text{g}/(\text{kg soil})$.

Boron concentrations in all Carson Sink soil water extracts are high (80–400 mg/kg), and even exceed typical concentrations for *total* B in soils (e.g. 2–100 mg/kg, Adriano, 1986). Boron concentrations in the extract solutions were all at concentrations which greatly exceeded both the effect level for vertebrate embryos (0.200 mg/L), and the irrigation water quality criterion (1.00 mg/L), indicating that ponded waters in equilibrium with these playa soils would contain toxic B concentrations. The very high B concentrations in the Carson Sink surface waters reported in Section 2.2.2. lend support to this interpretation. Boron inventories released with phosphate extracts in many samples (including the Carson Sink soils shown here, SWMA sediments and FIR soils) were commonly 40% to 80% greater than B inventories obtained with water extracts. Hot-water extractable boron contents (Bingham, 1982) of these soils were within this same range relative to water-extracted B.

Water-extracted Mo from Sink soils ranged from 1.5 up to 14 mg/(kg soil). These water-extractable Mo concentrations are quite high and are in the range of typical *total* soil Mo concentrations (< 3–15 mg/kg, Adriano, 1986). Although water-extractable Mo is commonly poorly correlated to plant-available Mo, it was noted in the study by Vlek and Lindsay (1977) that in alkaline, high Mo soils, water-extracted Mo is a satisfactory indicator of available Mo. Extractions of these soils and crusts with phosphate solutions usually did not yield higher concentrations of Mo than water-extractions.

Water-extracted Se from Carson Sink surface soils ranged from 3 to 130 $\mu\text{g}/\text{kg}$. The extracted Se was primarily selenate (SeO_4^{2-}). Phosphate-extracted Sink soils did not release significant quantities of selenite (IV) (SeO_3^{2-}), as expected from the high alkalinity of these soils (pH 8.8–10.0) (e.g. Neal *et al.*, 1987; Bar-Yosef and Meek, 1987). It is emphasized that water-extracted Se is a poor indicator of total soil Se, yet is potentially useful as an indicator of readily mobile, bioavailable Se. However, due to strong redox sensitivity of the water-soluble Se inventory, such data require cautious interpretations in wetland environments.

It is of interest to note the possibility that some of the Se in the north Carson Sink playa soils may have its origin in both runoff and erosion from the West Humboldt Range, rather than from the Carson River. Evaporite crust samples collected on the south slope of the Mopung Hills (part of the W. Humboldt Range) yielded water soluble Se concentrations of 45 to 105 $\mu\text{g}/(\text{kg soil})$, and similar samples collected along the southern length of Wildhorse Pass contained from 3 up to 1040 $\mu\text{g}/\text{kg}$ water-soluble Se.

Depth profiles for extractable B, As, Se and Mo from the CSD1 site are presented in Figs. 5.2a, b, c and d respectively. Both water-extracted and phosphate-extracted concentrations are provided. These figures illustrate that extractable concentrations of these trace elements are very high even below the surface crust. In several other Carson Sink profiles, the extractable concentrations of these trace elements increased with depth.

5.2. Estimates of Trace Element Concentrations During Ponding Over Some Carson Sink Sediments

The need for a method of estimating trace element concentrations in waters ponded over playas is apparent in view of the high soluble inventories of various trace elements measured in surface crusts and soils, the periodic accumulation of surface waters and the recent large-scale die-offs of wildlife in the Carson Sink. In this section, a simple model is aimed at developing a sense for the magnitudes of such concentrations. It is emphasized that the model is only presented as a means of very roughly estimating potential concentrations of trace elements in ponding waters. No specific mechanisms are employed in this model, hence assumptions used are minimal. Only the initial soluble trace element inventory (in the surface soil), depth of ponding and fraction of the soluble inventory which mixes into the ponded water need to be specified.

It will be simpler to illustrate a specific example, from which the general feature of the model can be deduce. For this, the case of boron (B) mixing into ponding waters will be used. Consider a unit volume of surface soil. For this example, a bulk surface area (A) of 1.00 m^2 and a thickness (denoted by d_s) of 0.10 m will be used. The surface soil has some

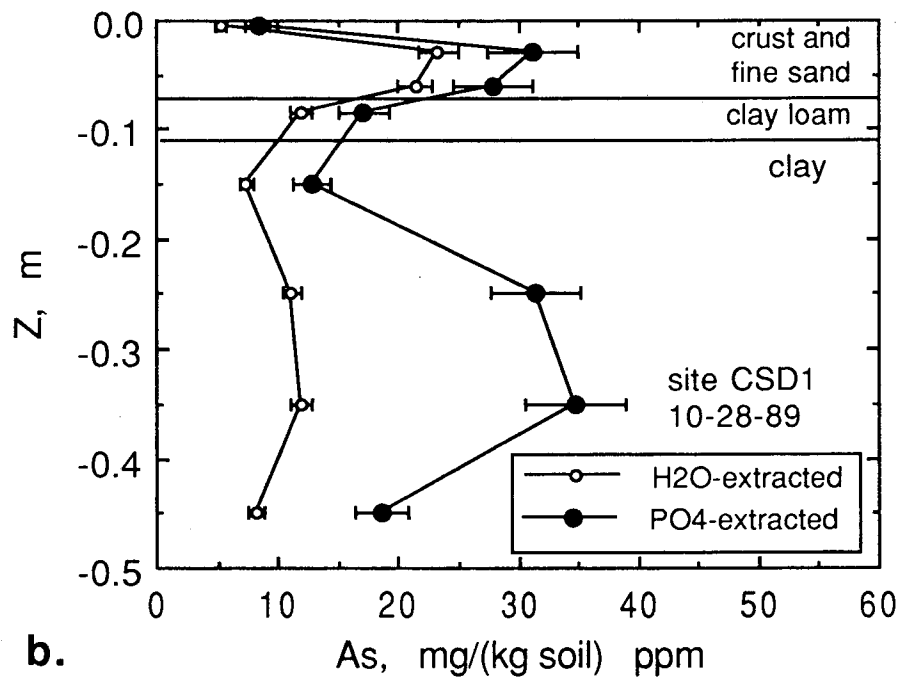
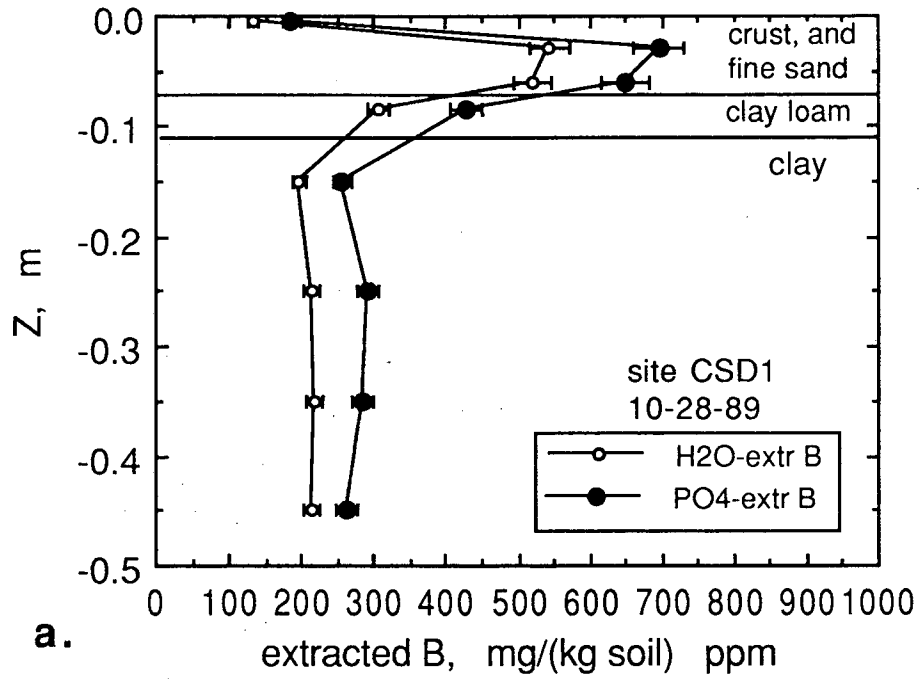


Figure 5.2. (a) Water-extracted and phosphate-extracted concentration profiles of arsenic and (b) Boron in Carson Sink soil sampling site CSD1.

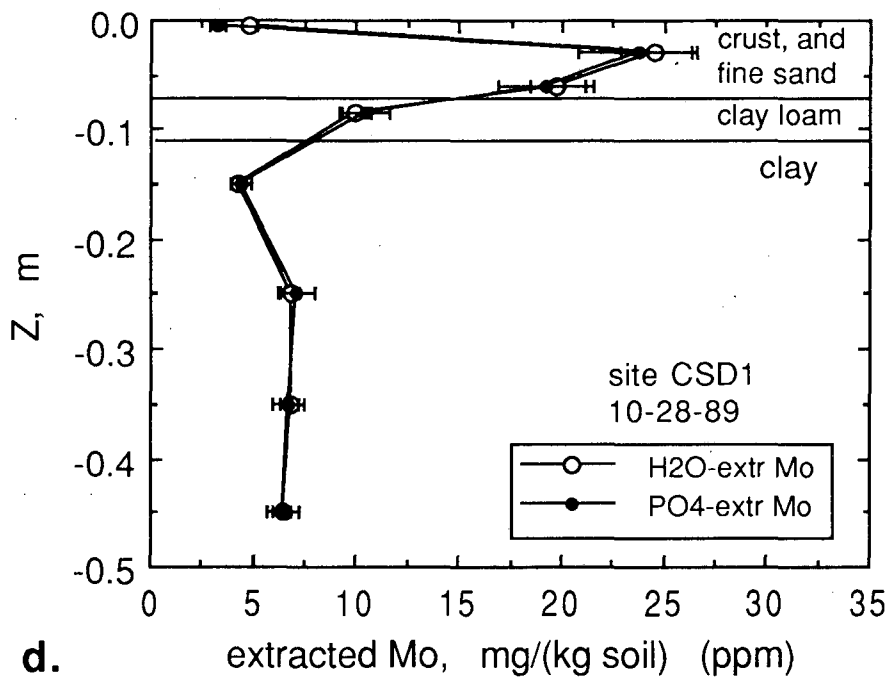
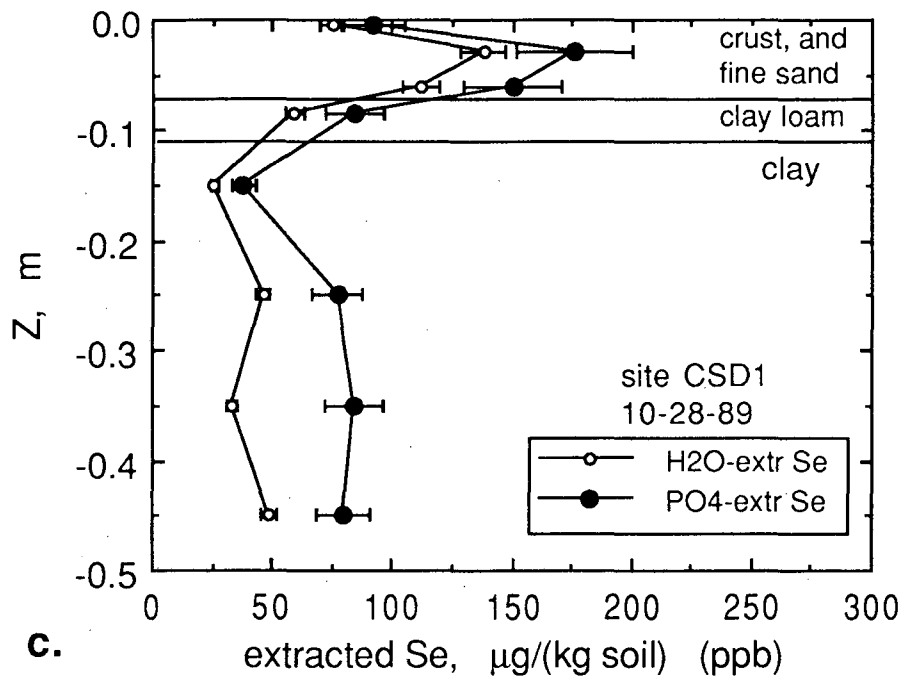


Figure 5.2. (c) Water-extracted and phosphate-extracted concentration profiles of selenium and (d) Molybdenum in Caron Sink soil sampling site CSD1.

bulk density (ρ_b) which will be about $1,300 \text{ kg/m}^3$ (mass of soil per bulk volume). The water-soluble concentration of B in this volume of soil is assumed to equal its water-extracted concentration (denoted by $C_s(B)$, the mass of B per mass of soil). The total mass of the “soluble” fraction of B (denoted by $m_o(B)$) is given by the product of this concentration times the bulk volume times the bulk density, or

$$m_o(B) = C_s(B)A d_s \rho_b \quad \text{Eq. (5.1)}$$

Only a fraction (denoted as r) of the total $m_o(B)$ mixes into the ponded water. Thus the mass of B dissolved in the pond water (denoted by $m_f(B)$) is given by

$$m_f(B) = r C_s(B) A d_s \rho_b \quad \text{Eq. (5.2)}$$

The mass of the ponded water (denoted by m_w) per unit area A is given by $m_w = \rho_w A d_w$, where ρ_w is the density of the ponded water and d_w is the depth of ponding. Since only estimates of water quality are sought, a value of $1,000 \text{ kg/m}^3$ may be used for ρ_w . The resulting concentration of B in the ponded waters (denoted by $C_w(B)$) is then given by $m_f(B)/m_w$ when expressed per mass of ponded water, or

$$C_w(B) = \frac{r C_s(B) \rho_b d_s}{\rho_w d_w} \quad \text{Eq. (5.3)}$$

which when reasonable values for the particular variables are substituted simplifies to

$$C_w(B) = \frac{(0.13m)r C_s(B)}{d_w} \quad \text{Eq. (5.4)}$$

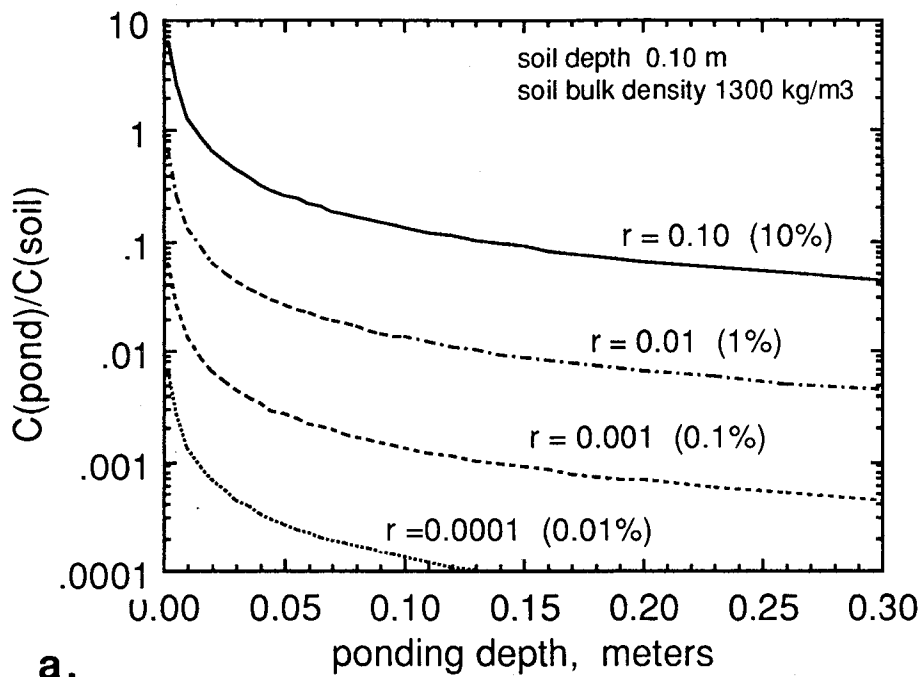
Thus, the concentration of B in the ponded waters is directly proportional to the initial soluble inventory times the mixing ratio r and inversely proportional to the ponding depth. The above relation may be rearranged to express the result as a ratio of the concentration in the ponded waters to that of the initial soil concentration, or

$$\frac{C_w(B)}{C_s(B)} = \frac{(0.13m)r}{d_w} \quad \text{Eq. (5.5)}$$

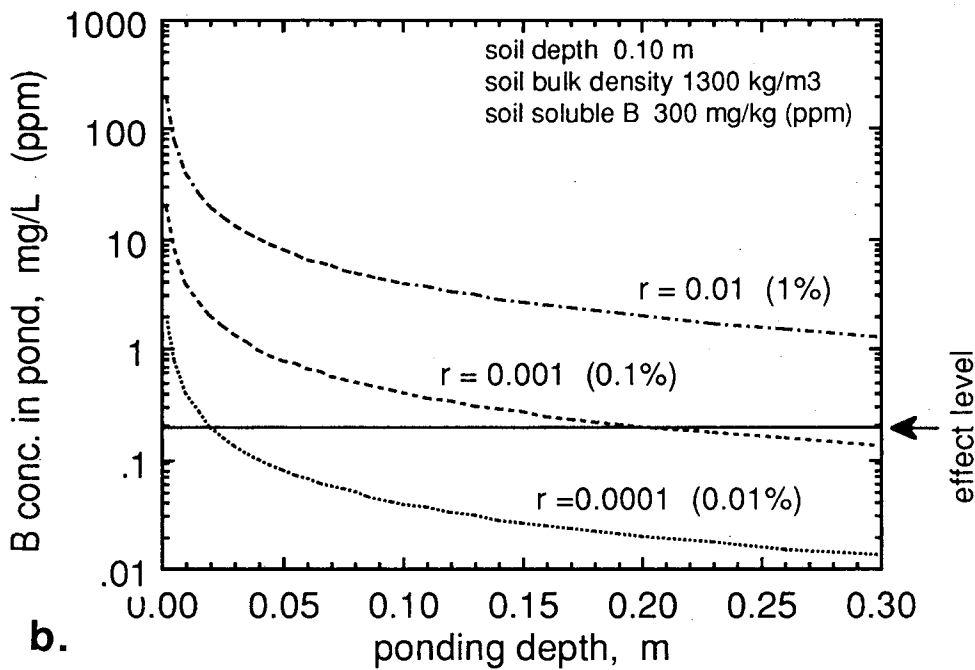
The right side of Equation 5.5 actually is an expression for the ratio of concentrations in the ponded waters versus the initial soil concentration for any of the trace elements of interest, although different values of r would probably be needed for the different components. These rather simple results clearly hinge on an appropriate selection of r , the fraction of the initial inventory which mixes into the ponded waters.

One may still use the above result with a range of possible r values in order to gain an appreciation for possible magnitudes of B concentrations in ponded waters. This approach was taken in combination with the measured trace element concentrations presented in section 5.1. In Fig. 5.3a, the previous Equation 5.5 for the ratio of pond to soil concentrations is plotted as a function of ponding depth for 4 different values of r . Again, the linear dependence of the resulting pond concentration on r is clear. Equally important is the consequence of the inverse dependence of the pond concentration on the depth of ponding. This inverse relation allows for large decreases (dilutions) with small increases in ponding in very shallow systems, or conversely large increases in concentrations with decreased depths at very shallow ponding depths. On the other hand, at large ponding depths, concentrations are relatively insensitive to fluctuations in depths. The inverse relation between concentrations and ponding depth also demonstrate that measurements of relatively stable concentrations during periods of deeper ponding provide no assurance that even roughly similar concentrations will prevail as pond levels recede. This latter feature would be important when evaporation is more important than seepage as the mechanism for pond level recession. These apparently trivial results become more significant when the measured concentrations of soluble inventories of trace elements in Carson Sink soils are entered into the equations.

The application of the model to the case of B in Carson Sink soils, using Equation 5.4, is illustrated in Fig. 5.3b. A concentration of 300 mg/(kg soil) was used as a reasonable $C_s(B)$ based on the data presented in Section 5.1. The curves represent pond water B concentrations as a function of pond depth assuming only 0.01%, 0.1%, and 1% of the initial soluble B mixes into the pond waters. Included in the graph is a line denoting the ef-



a.

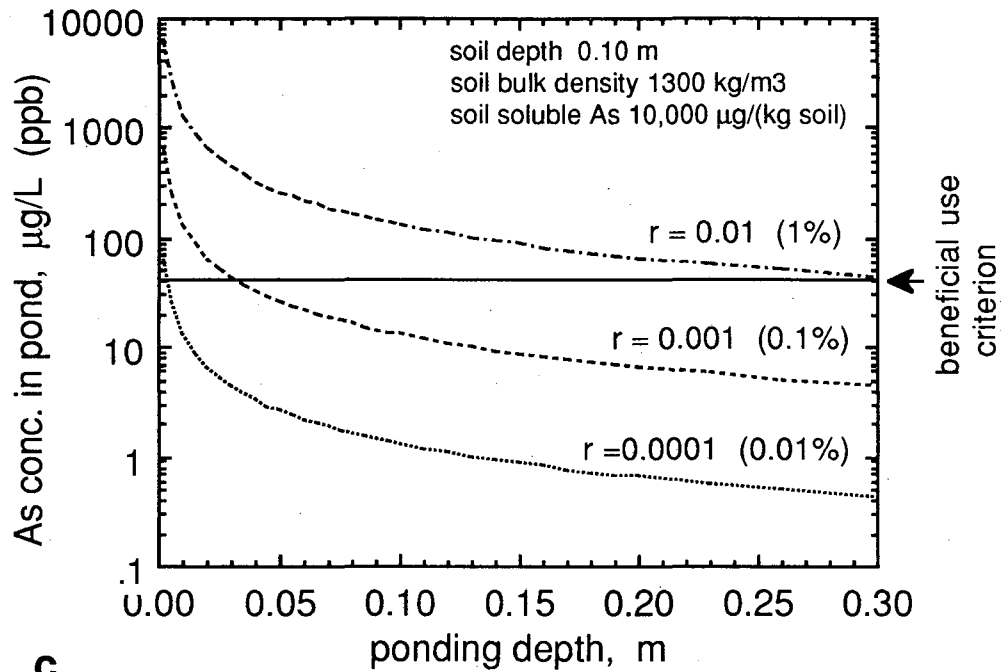


b.

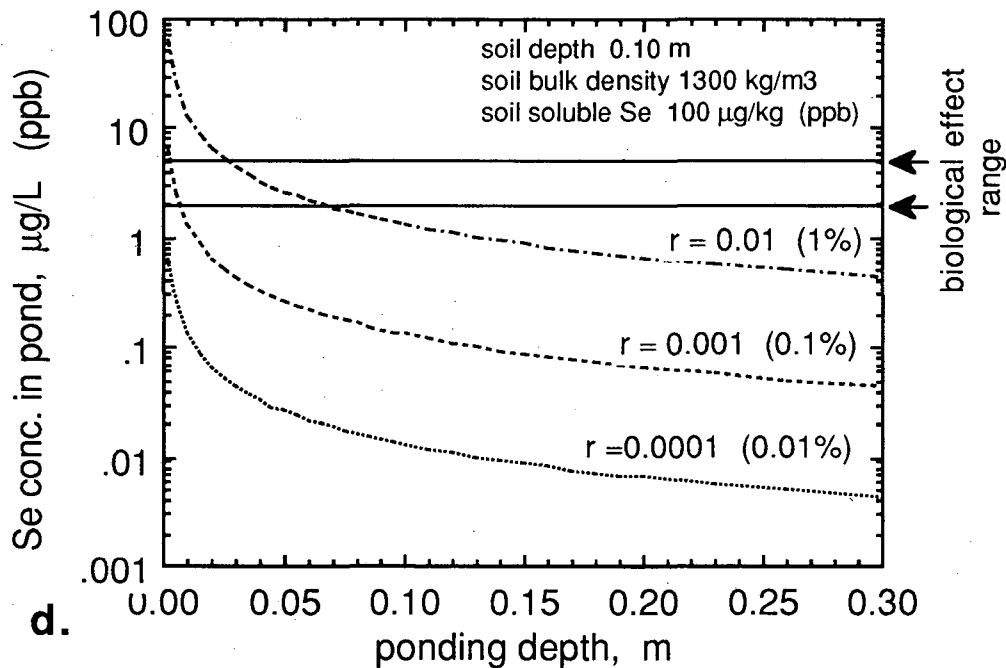
Figure 5.3. (a) Estimates of ratios of solute concentrations in ponded waters to their initial soluble concentration in surface soils, as a function of ponding depth for mixing ratios of 0.0001, 0.001, 0.01 and 0.1. (b) Estimates of boron concentrations in waters ponded over the Carson Sink for mixing ratios of 0.0001, 0.001 and 0.01.

fect level for B. The fact that even the curve with only 0.1% mixing remains substantially above the reference line suggests that B concentrations in such ponding events will be high. Similar graphs for As and Se in hypothetical ponding events over the Carson Sink are shown in Figs. 5.3c and d. In Fig. 5.3c, As concentrations are projected to exceed effect levels over significant ranges of ponding depths when r is greater than about 0.001 (0.1%). Selenium concentrations in ponding waters do not appear likely to exceed effect levels.

The previous calculations have been done to illustrate the fact that surface water quality in the Carson Sink will not satisfy various water quality goals when even only very small fractions of the surface sediment inventory of trace elements partition into ponding waters. It is recognized that the calculations include simplifications which will not be appropriate to particular cases. However, field data relevant to such ponding events indicate that results from the previous calculations are reasonable. Reported values of trace element concentrations in waters ponded in the Carson Sink during the 1986–87 flooding suggest that partitioning of some sediment trace elements at similar, if not greater, mixing ratios occurred. Rowe and Hoffman (1987) have reported B and As concentrations in Carson Sink surface waters of 40 mg/L and 800 μ g/L respectively for this event. In the very different setting of small ephemeral pools formed from rainfall ponding on soils at Kesterson Reservoir, California during the 1988–89 and 1989–90 wet seasons, further indications of solute transfer from soils to surface waters can be obtained. Data collected from these ephemeral pools indicated that approximately 1% of the soluble Se inventory in the surface 0.15 m of soils mixed into ponded waters (LBL, 1991). In cases where the ponded water results from water table rise, or when ponding water results from accumulation of runoff which has accumulated solutes during its traverse, even higher final solute concentrations can be expected. Such cases have also been documented at Kesterson Reservoir (e.g., LBL, 1991).



c.



d.

Figure 5.3. (c) Estimates of arsenic concentrations in waters ponded over the Carson Sink for mixing ratios of 0.0001, 0.001 and 0.01. (d) Estimates of selenium concentrations in waters ponded over the Carson Sink for mixing ratios of 0.0001, 0.001 and 0.01.

5.3. Extractable Trace Element Contents of Some SWMA Sediments

As in the case of the Carson Sink playa soils, information on present inventories of available trace element contents in surface sediments within the SWMA is useful in understanding the potential extent of sediment trace element inventory recycling into pond waters. Such an understanding would be useful in surface water management decisions. The next two sections represent initial phases of a study aimed at providing such information. As a first step, water-soluble inventories of a set of SWMA sediments were estimated by performing water-extracts. This information is presented in this section. In Section 5.4, results from the first set of sediment rewetting experiments will be presented.

Sediment samples were collected from several locations within the SWMA on May 12, 1990 and on June 13, 1990. Sampling procedures are outlined in Appendix A. Crust samples, shovel samples, auger samples and core samples have all been collected. To date, only the work with the crust and shovel samples have been completed. Samples were collected in one of several ditches supplying South Lead Lake, near South Lead Lake Landing, in Pintail Bay near Pintail Bay Landing, in East Goose Lake and in the Hunter Drain Sump.

The South Lead Lake ditch samples were obtained in a ditch along the south side of Center Rd., 1.00 km west of South Lead Lake Landing. (This is not to be confused with the Lead Bypass Canal which runs along the north side of Center Rd.) Water in this ditch appears to be supplied by a portion of the combined TJ and Paiute Drain waters. At the time of sampling, both flow rates and water levels were low. The maximum channel water depth was about 0.05 m. Sediment and core samples were collected along the shoreline at this site. All sediments described in this section were sampled over the surface 0.10 m depth unless otherwise specified. At the very least, duplicate samples of each type were collected at each site and later analyzed separately.

The South Lead Lake sediment samples were collected 95 m W-SW of South Lead Lake Landing, about 10 m offshore at the edge of a stand of cattails. Shovel and core samples were collected from beneath 0.25 m of ponded water.

Pintail Bay was dry at the time of sampling. Samples were collected at a site located about 80 m N of the south edge of the dry lake (near Pintail Bay Landing). The lake bed was sparsely revegetated with *Kochia*, *Cressa* and *Distichlis*. A superficial, discontinuous salt crust covered the sample site. The sediment was sampled in 3 layers spanning the 0–0.01 m (crust), 0.01–0.05 m and 0.05–0.10 m depths.

East Goose Lake was dry and unvegetated at the time of sampling. This site was the only one sampled on June 13, 1990. Samples were collected in an area 0.80 km N-NW of the intersection of Nutgrass Rd. and Navy Cabin Rd., 100 m S-SW of the dry shoreline. Sediments were sampled at approximately 0–0.01 m, 0.01–0.03 m and 0.03–0.10 m depth intervals.

The Hunter Drain Sump had a ponded area estimated at 10 hectare (25 acres) at the time of sampling. Sediment samples were collected along the south shore, at the shoreline.

Water-extracted concentrations of As, B, Li, Mo, selenite (“Se⁴”) and soluble Se from these sediments are presented in Table 5.1. The lithium data have been included in these analyses in response to the interest in high concentrations of this cation noted in previous studies. Extracted As contents of these SWMA surface sediments are generally lower than those of the Carson Sink extracts by rough a factor of 10. Water-extracted B contents in some SWMA pond crusts are similar to values obtained in the Carson Sink. However, when averaged over the full 0.10 m depth intervals, SWMA sediments are generally significantly lower in water-extracted B than the Carson Sink samples. Molybdenum concentrations in crusts and in most of the East Goose Lake samples were higher than concentrations obtained from the Carson Sink samples. Water-extracted Se was highly variable in the SWMA samples. Crust samples were relatively high, with the Pintail Bay samples having water-extracted Se concentrations of 257 and 501 µg/(kg soil) (ppb soil). The

presence of significant concentrations of Se(IV) in water extracts of these crust may indicate that Se accumulation in SWMA pond bottom sediments by reductive immobilization may have occurred in a manner similar to that observed at Kesterson Reservoir. While these Se concentrations are relatively low in comparison with sediment samples from Kesterson Reservoir, they may still provide sediment pore water and pond water environments which exceed effect levels. The data in Table 5.1 indicate that water-extractable Se contents in some SWMA sediments may average about 50 $\mu\text{g}/(\text{kg soil})$. Under conditions where such sediments are rewetted to saturation, pore water Se concentrations which exceed 200 $\mu\text{g}/\text{L}$ (ppb) may arise. The fact that such possibilities are not discernable in total sediment Se analyses again raises questions concerning the utility of such screening techniques.

Comparisons of relative concentrations of these trace elements between various ponds based on the data in Table 5.1 is not warranted for at least two reasons. First, these data represent a very limited set of point samples taken out of an extensive area which is expected to exhibit considerable spatial variability with respect to sediment chemical composition within any individual pond. A second reason to discount any attempts to make site to site comparisons with this data set involves the fact that samples were collected under widely varying conditions (submerged under water to dry playas). The extent to which these varying field conditions influence water-extractable inventories of the various trace elements is still poorly understood despite the vast literature on equilibrium solubility relations. Thus, while it is to be expected that the lakes further downstream (such as East Goose Lake and Pintail Bay) will on average contain higher trace element concentrations than upstream sites (such as South Lead Lake) because of evaporative concentration of solutes as water is routed through the system, such a conclusion can not be made with these data.

In the experiment described in the following section, some of the sediments characterized here were subjected to rewetting so that the ponded waters could be analyzed.

Table 5.1. Water-extracted concentrations of arsenic, boron, lithium, molybdenum, selenite (Se⁴), and total water-extractable selenium in several SWMA sediments. Note that the arsenic and selenium concentrations are in units of µg/(kg soil) or ppb. Details of locations are provided in the text.

| Site Name | As µg/kg | B mg/kg | Li mg/kg | Mo mg/kg | Se ⁴ µg/kg | Se µg/kg |
|---|-------------|------------|-------------|-------------|--------------------------|-------------|
| South Lead Lake "ditch," sed.-1 | 284 | 5.4 | 3.90 | 1.5 | 2.5 | 6.5 |
| South Lead Lake "ditch," sed.-2 | 235 | 4.8 | 0.50 | 1.0 | 2.5 | 5.5 |
| South Lead Lake Landing sediment 1 | 402 | 5.7 | 1.05 | 1.9 | 2.0 | 4.3 |
| South Lead Lake Landing sediment 2 | 546 | 5.9 | 1.00 | 2.1 | 1.6 | 4.1 |
| East Goose Lake crust | 1150 | 180 | 1.95 | 30.5 | 12.5 | 128 |
| East Goose Lake crust | 995 | 139 | 1.40 | 22.6 | 6.0 | 61.0 |
| East Goose Lake crust | 990 | 121 | 1.65 | 29.8 | 6.0 | 68.0 |
| E Goose Lake sediment, 0.01-.03 m, 1 | 1450 | 172 | 1.55 | 21.7 | 6.5 | 72.0 |
| E Goose Lake sediment, 0.01-.03 m, 2 | 1270 | 128 | 1.35 | 22.2 | 5.5 | 57.0 |
| E Goose Lake sediment, 0.03-.10 m, 1 | 1590 | 144 | 1.15 | 10.4 | 2.0 | 30.0 |
| E Goose Lake sediment, 0.03-.10 m, 2 | 1610 | 128 | 1.25 | 15.1 | 2.5 | 29.0 |
| Pintail Bay Landing, crust 1 | 407 | 198 | 9.95 | 30.0 | 27.8 | 501 |
| Pintail Bay Landing, crust 2 | 405 | 180 | 7.90 | 21.7 | 20.8 | 257 |
| Pintail Bay Landing sed., 0.01-.05 m, 1 | 838 | 34.9 | 2.15 | 2.6 | 5.3 | 37.0 |
| Pintail Bay Landing sed., 0.01-.05 m, 2 | 870 | 33.8 | 2.10 | 2.6 | 4.7 | 37.5 |
| Pintail Bay Landing sed., 0.05-.10 m, 1 | 1020 | 19.5 | 0.80 | 1.0 | 3.7 | 19.1 |
| Pintail Bay Landing sed., 0.05-.10 m, 2 | 1080 | 18.5 | 0.80 | 0.9 | 3.3 | 19.9 |
| Hunter Drain Sump sed., 0-0.10 m, 1 | 1620 | 72.1 | 3.00 | 5.0 | 14.2 | 26.0 |
| Hunter Drain Sump sed., 0-0.10 m, 2 | 1650 | 71.7 | 2.95 | 4.3 | 12.2 | 22.3 |

5.4. A Laboratory Study of Water Quality Changes Following Freshwater Ponding on SWMA Sediments

With the understanding that concentrations of certain trace elements occur at elevated levels in surface sediments within the SWMA, it is important to determine the extent to which currently existing inventories of these trace elements may be recycled into ponded

waters. A number of different laboratory and field experimental procedures may be used to address various aspects of this issue. The particular process of interest in the experiment described here is relevant to situations where dried sediments are rewet. Such conditions occur during reflooding of dry lake beds and also occur along shorelines where pond water levels are raised. The experimental design was kept very simple so that other aspects of this work could continue to proceed.

Of the sediment samples described in Section 5.3, all but the East Goose Lake sediments and the Pintail Bay crusts were also used in the rewetting experiment. The subsurface samples from Pintail Bay were treated individually. After determining the moisture contents of the samples, enough soil from each sample was weighed into a beaker such that the equivalent of 100.00 g of dry soil mass was obtained. The sediments formed pads with thicknesses of about 40 mm. For each sediment, 4 separate replicate subsamples were prepared. These samples were allowed to air dry at room temperature for a period of 65 days. Shrinkage and cracking which occurred during drying of the initially wetter sediments produced large aggregates of nonuniform dimensions. It was anticipated that when waters were ponded over such large, irregular aggregates, large variations in pond water quality may result just from differences in efficiencies of water penetration into aggregates and differences in rates at which trace elements are transported into the ponding waters. Yet such heterogeneous aggregate structures clearly occur in the field during drying of ponds. Therefore, two replicates from each sediment were individually crushed, sieved passed through a 4.5 mm sieve and returned to their original containers. Thus, for a given sample material, duplicates of strongly aggregated dried sediments and duplicates of rehomogenized dried sediments were obtained. In this manner, effects of aggregate structure may be inferred from any systematic differences between aggregated versus crushed samples. After weighing, residual water contents were calculated for each sample. Enough distilled water was then added to each sample to bring the water:soil mass ratio up to 2:1. Initial ponding depths of about 50 mm resulted. Small volumes of the ponded water (ranging from 2 to 10 mL) were sampled from each container after 1, 7, 12 and 22 days following the re-ponding. These samples were diluted to provide sufficient volumes for various

analyses. In this section, only the salinity (EC), B and As data will be reviewed. (Other data are still being processed.)

The EC measurements were performed on the diluted samples, temperature corrected and linearly normalized to the original sample volume. Over the EC ranges encountered in this experiment, such a procedure overestimates the actual undiluted sample EC, but on the other hand such extrapolations are more closely linearly related to the TDS. These renormalized EC data are shown in Figs. 5.4a and b. The EC data shown in these figures exhibit the expected behavior in two manners. First, for a particular sediment, the increasing EC with time is expected as a result of movement of salts into the ponding waters and evaporative concentration of these salts. A second predictable pattern in the EC data concerns the relative magnitudes of salinity in the different sediments. The water extracts of these samples described previously were also measured for EC and were ranked in the same order as shown in Figs. 5.4a and b. The Hunter Drain Sump sediments were the most saline, followed by the Pintail Bay sediments and South Lead Lake and South Lead Lake Ditch sediments. It is also interesting to note the relatively high degree of reproducibility of these results, especially between aggregated and crushed subsamples. This indicates that, for this particular experiment, diffusion of major ions out of the aggregates was not a significant limiting step for ion transport into the pond waters. The initial aggregate dimensions which ranged up to about 30 mm, would have introduced a lag time in the EC increases for the aggregated samples. The expected lag time is associated with diffusion-limited ion transport out from the aggregates. Lack of measurable differences may have resulted from the breakdown of aggregates during rewetting.

Boron concentrations in ponded waters are presented in Fig. 5.5a. Trends in the pond water B concentrations parallel those for salinity. This is true both in terms of temporal increases within a given sediment and in terms of correlations between B contents in water extracts (Table 5.1) and B concentrations in the ponded waters. Again, reproducibility of the data suggests that intra-aggregate diffusion was not a limiting step in the movement of B into the ponded waters. The most important result to note is that effect levels for

B in surface waters were greatly exceeded in all cases, despite ponding with initially clean waters. Prevention of leaching out through the bottoms of the beakers certainly contributes to keeping B and other solutes in closer proximity to the ponded waters and is a limitation of this particular experiment. However, the fact that the effect level was exceeded by over 100-fold in certain cases indicates that during reponding of many areas, downwards leaching will not be sufficient to overcome diffusion of B and other trace elements back into ponding waters. This is especially true over low permeability surfaces which are characteristic of both lakes and playa bottoms.

Arsenic concentrations in waters ponded over SWMA sediment samples are shown in Fig. 5.5b. The patterns described previously for salinity and B are largely repeated in the case of As movement into the ponded waters. The one significant exception to the pattern is the higher As concentrations in waters ponded over South Lead Lake ditch sediments relative to the concentrations in waters ponded over South Lead Lake Landing sediments. In the water extracts, this order is reversed. Again, the important point which is demonstrated in these data is that high concentrations of As can result simply from recycling of a portion of the surface sediment inventory back into the ponded waters.

Considerably more data have been obtained from this and related experiments. Much more work can be done in this area. The combination of laboratory experiments with sediment data and field water quality data will greatly improve our understanding of trace element recycling between sediments and surface waters. Such information is needed in order to understand the implications of surface water management decisions in this environment.

5.5. Summary of LBL Studies on Carson Sink and SWMA Crusts and Surface Soils

Information on salinity and trace element concentrations in crusts and surface soils from the Carson Sink and the SWMA has been presented from several different approaches in the previous sections. Inventories of water-extracted and phosphate-extracted As, B, Se and Mo from several Carson Sink soils have been discussed. Estimates of trace element

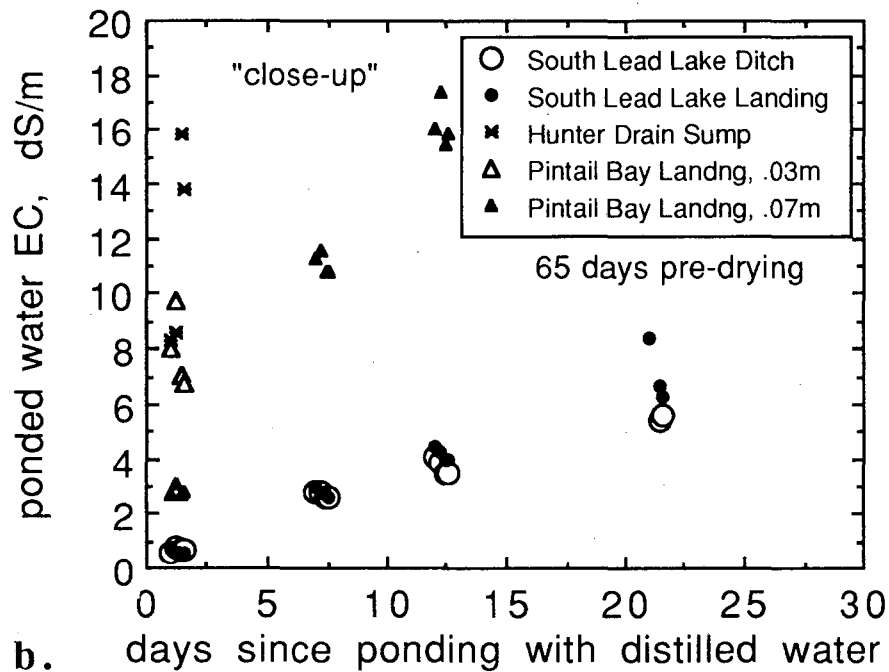
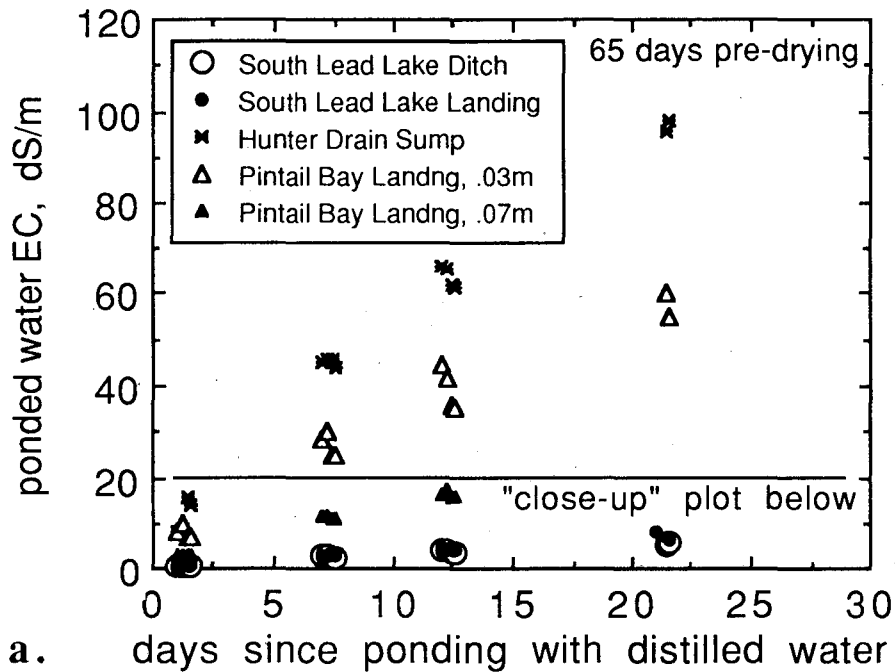


Figure 5.4. (a) Electrical conductivities (EC) of waters ponded over SWMA sediments. The plotted data are linearly normalized to the original sample volume from measurements made on diluted samples. (b) Close-up view of lower EC data.

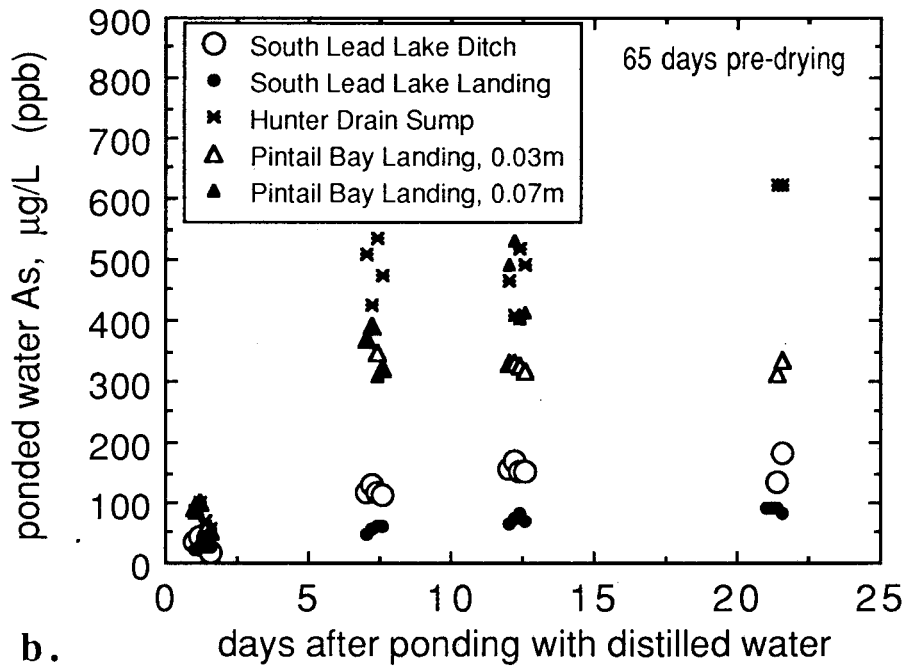
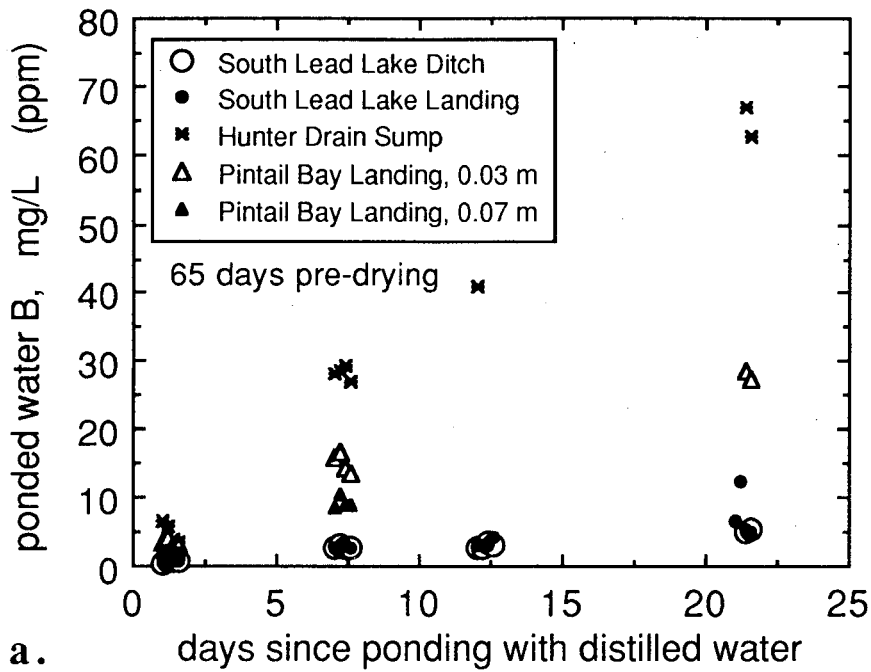


Figure 5.5. (a) Boron concentrations in waters ponded over SWMA sediments. (b) Arsenic concentrations in waters ponded over SWMA sediments.

recycling from Carson Sink soils into ponding waters have been outlined. Data on water-extracted As, B, Li, Mo and Se from SWMA lake sediments have been presented. Laboratory experiment data on surface water quality resulting from ponding distilled water over dried SWMA lake sediment samples were also presented.

These various ways of investigating Carson Sink playa soils and SWMA sediments indicate that very high concentrations of soluble salts and some trace elements are characteristic of these environments. Water-extracted As concentrations ranged from 235 to 1,650 $\mu\text{g}/\text{kg}$ in SWMA sediments and from 5,000 to 22,000 $\mu\text{g}/\text{kg}$ in Carson Sink soils. Water-extracted B concentrations ranged from 5 to 198 mg/kg in SWMA sediments and from 120 to 500 mg/kg in Carson Sink soils. Although only a small number of samples were analyzed, these high concentrations are to be expected in this setting. These samples were obtained from areas near and within the bottom of a very large closed basin.

Through simple calculations, it has been shown that surface water quality goals can be exceeded even when only a very small fraction (less than 1%) of the soluble inventory of B, As and Se in Carson Sink surface sediments becomes dissolved in ponded waters. Laboratory experiments involving ponding of distilled water over some SWMA sediments demonstrated that high salinities (EC values ranging from 5 to 100 dS/m), high B concentrations (5 to 67 mg/L) and high As (90 to 620 $\mu\text{g}/\text{L}$) concentrations in ponded waters can occur simply through recycling from the sediments.

It is important to recognize that the high salt and trace element inventories in SWMA sediments and Carson Sink soils are primarily a consequence of the regional geochemical characteristics and regional drainage and evaporation patterns which have operated since the recession of Lake Lahontan. The relatively very recent introduction of agricultural drainage waters from farms throughout the Lahontan Valley into the SWMA wetlands have significantly compromised surface water quality (as shown in Chapter 2), yet have contributed to a much lower relative loading of the already high sediment inventories. Thus, even with improvements in inflow water quality, careful water management within the wetlands will

be required to minimize recycling of salts and trace elements from sediments back into surface waters. While such management is impractical within the Carson Sink (which has no discharge area), it should be recognized that the enormous inventory of salts and trace elements in the sink are largely a product of natural drainage and evaporation over thousands of years rather than a result of recent agricultural drainage.

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APPENDIX A: SOIL AND SEDIMENT SAMPLING PROCEDURES

Soil and sediment samples were obtained using several methods. Soil profile sampling was performed using push tube sampling and auger sampling. The push tube samples provided relatively undisturbed soil cores encased in clear plastic sleeves. The steel push tube sampler provides 38 mm diameter cores up to 1.27 m in length. The sampling tube is driven into the soil hydraulically with a trailer-mounted sampler-drill.

Most soil samples included in the present report were obtained by hand augering at 0.10 m intervals. The 0.05 m diameter bucket auger was thoroughly cleaned between sampling increments. Field observations on auger samples included texture estimation, noting the presence of roots, mottling, gleying, evaporites and testing for carbonates. Individual soil samples were stored in Ziploc brand freezer bags. Within most sites, soil cores were taken in up to 3 profiles, laterally spaced at 1 to 2 m. Auger sampling in duplicate to triplicate was performed at most sites down to at least 1.0 m. Samples collected at common depth intervals within a given site were not composited. Later individual analyses of these replicate samples was useful in checking for the possibility of cross-contamination across depth intervals. (No evidence of this problem was detected.) All auger holes were backfilled with compacted potting soil, with the exception of the profiles sampled within non-FIR lands and on the JA field in the FIR. At non-FIR sites, holes were backfilled with local soil. In the case of the earlier JA field samples, holes were backfilled with soil recovered from well drilling, 15 m east of the JA-E-s2 augering site. The auger samples collected in the JA field were exceptions in that no replicate samples were collected. Potting soil was selected as the backfill material at all other FIR sampling sites since it was both very different from all local soils and was readily available. The former property was desired because of our interest in resampling these sites at some future date in order to determine rates of soil profile leaching. By using the compacted potting soil backfill, any accidental resampling of backfill material could be easily detected. Recognizing the greater susceptibility of the backfill material to both wind and water erosion, the surface 0.10 m inter-

val of backfill was comprised of local surface soil. The depth of sampling is generally limited by the presence of the shallow water table. At or immediately below the water table, it became difficult to retrieve unmixed samples, especially in sandy soils. During the auger-sampling within the FIR (December 1989 through March 1990), the water table imposed a sampling depth limit ranging from 1.50 to 3.30 m.

At sampling sites which exhibited a distinct salt encrusted surface, the crust was sampled separately. Crust samples were collected at various playa sites, drain banks, lake shorelines, evaporite deposits and seeps along the West Humboldt and Stillwater Ranges. Soil crust sampling is performed on a specified projected surface area (usually either 0.010 or 0.040 m²), and a nominal depth.

Sediments in the SWMA were collected as both shallow cores and shallow shovel samples. Both types of samples were limited to the surface 0.10 m. Core samples were obtained by pushing a 73 mm I.D. (3 mm wall), 0.10 m long stainless steel ring into the sediment, excavating the core and surrounding sediment with a shovel, trimming the exterior sediment away from the ring wall, trimming the core bottom flush with the ring bottom and sealing the bottom. Both sediment core and shovel samples were kept refrigerated in an ice chest between the time of sample collection and return to the laboratory.

APPENDIX B: SOIL AND SEDIMENT EXTRACTIONS

Soil samples are maintained as close as possible to the field-sampled moisture contents up to the time of extraction. Individual samples are quickly chopped to pass a 4.75 mm sieve, homogenized and returned into the Ziploc brand bag. Subsamples are used to obtain moisture contents of the soils (105°C, 24 hours).

Water-extracts are obtained by mixing 1:5 ratios of soil:water in a reciprocating shaker (3 cycles/second) for 1.0 hours. Distilled water is added to the soil to achieve the 1:5 mass ratio, taking into account the initial moisture content of the soil. Commonly 30.0 g of equivalent dry soil are used in each water-extract. The suspensions are agitated in 250 ml screw cap plastic bottles. After 1.0 h of shaking, the bottles are placed directly into a centrifuge. Centrifugation at ≈ 1000 rcf for 5 to 10 minutes is supplemented by higher rates and longer times as necessary. The supernatant solution is filtered through a 0.45 μm filter. A wide range of soils were also used to compare trace element compositions in extracts obtained using either distilled water, Lahontan Reservoir water, or simulated S-Line Canal water. The motivation for and results of this latter study are described in Section 2.2.1 and at the end of this section.

The more conventional practice of air-drying samples, followed by crushing to pass a 2 mm sieve prior to extraction was not used since this approach is believed to introduce severe changes in the redox status of some samples. In particular, oxidation of Se in originally reducing soils may result in water-extractable Se concentrations that are considerably higher than the those of the original sample. While most soil samples involve in this study to date are not macroscopically reducing, the ubiquitous presence of anaerobic microsites in aggregated soils may still result in the aforementioned artifact. In profiles where such redox-sensitive changes are demonstrated to be unimportant, use of more conventional sample processing will be reconsidered.

Phosphate Extractions

In some soils, an additional extraction is performed in a 1 mM Na₂HPO₄ suspension in a 1:20 soil:extractant mass ratio. The suspension is agitated as previously described, by for a period of 24 h. Centrifugation and filtration procedures are identical to that of the water-extract. The phosphate extraction is used as a method to estimate specifically-adsorbed oxyanion (e.g. selenite, arsenate, molybdate and borate) contents. It is recognized that the procedure yields results that are operationally defined (e.g. 1 mM phosphate extractable at 1:20, 24 h) rather than rigorously mechanistic (e.g. ligand-exchanged or solid phase dissolution). However, nearly all extraction techniques (including certain (simple) water-extractions) suffer from this limitation.

Hot-water Boron Extractions

The hot-water extraction for available B (Bingham, 1982) was also tested on a range of soil samples. A comparison of these results with corresponding B concentrations obtained with ordinary water extracts indicated that hot-water extracts for B are fairly linearly related to B contents obtained in ordinary water extracts, with the former procedure yielding about 51% more B. From a comparison of 24 samples, the correlation is approximated by the linear regression (with an $r^2 = 0.893$)

$$C_B(\text{hot H}_2\text{O}) = (0.86 \text{ mg/kg}) + 1.51C_B(1:5 \text{ cold H}_2\text{O extract}) .$$

This result for B indicates that B concentrations obtained in phosphate extracts are generally similar to those obtained in hot-water extracts.

Total elemental analyses for As and Se using X-ray fluorescence spectroscopy has been performed on a subset of the samples (Giauque *et al.*, 1979).

Soil Extractions Using Lahontan Reservoir Water and a Simulated S-Line Canal Water

In order to determine if the particular chemical composition of the irrigation waters applied over Lahontan Valley farmlands, including those of the FIR, had any significantly different effects than the distilled water used for most of the work on extractable trace element compositions of local soils, comparative extractions on a wide range of FIR soils were performed with distilled water, a simulated S-Line Canal water (using averaged major ion composition data from USBR and USGS reports), and a sample of Lahontan Reservoir water (collected on 11-01-90).

Duplicate 1:5 soil-water extracts were performed on 10 samples as described earlier in this appendix. After subtracting background trace element concentrations from the Lahontan Reservoir water (Table 2.2), the resulting analyses for As, B and Se were all in agreement with each other within uncertainties of the individual analyses.

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