APPENDIX D

DATA ANALYSIS

TABLE OF CONTENTS

Page

1.0 DATA ANALYSIS D-1
- 1.1 INTRODUCTION TO INVESTIGATION ITEMS FOR HABITAT CHARACTERIZATION D-1
- 1.2 HABITAT CHARACTERIZATION INVESTIGATION TOPICS D-1
  - 1.2.1 Hydrologic Flow Regimes D-1
  - 1.2.2 Concentrations of Major Chemical Constituents in the Arid West D-9
  - 1.2.3 Taxonomic Richness D-38
  - 1.2.4 Toxicity Database Species D-43
  - 1.2.5 Flow-Augmentation Effects D-45
  - 1.2.6 Ambient Background Concentrations D-72
  - 1.2.7 Wastewater Treatment Improvements D-73
  - 1.2.8 Adequacy of Available Habitat Characterization Protocols D-93

LIST OF FIGURES

D-1 Mean Monthly Streamflow Downstream of WWTP Outfalls
D-2 Ratio of Annual Maximum Flow to Annual Mean Flow Downstream of WWTP Outfalls
D-3 Piper Diagram for the Phoenix Study Area
D-4 Piper Diagram for the Nogales Study Area
D-5 Piper Diagram for the Tucson Study Area
D-6 Piper Diagram for the Santa Ana Study Area
D-7 Piper Diagram for the Fountain Creek Study Area
D-8 Piper Diagram for the Las Vegas Study Area
D-9 Piper Diagram for the Santa Fe Study Area
D-10 Piper Diagram Showing Average Cation and Anion Concentrations for 7 of the 10 Study Areas and Eastern Streams
D-11 TDS Box Plots
D-12 Conductivity Box Plots
D-13 Hardness Box Plots
D-14 Alkalinity Box Plots
D-15 Dissolved Oxygen Box Plots
D-16 pH Box Plots
D-17 Ammonia Box Plots
D-18 Loadings for PC1, PC2, and PC3 for Cations and Anions Only Model
D-19 Scores for PC1 vs. PC2, PC1 vs. PC3, and PC2 vs. PC3 for Cations and Anions Only Model
D-20 Loadings for PC1, PC2, and PC3 for Cations/Anions + Key Parameters Model

D-21 Scores for PC1 vs. PC2, PC1 vs. PC3, and PC2 vs. PC3 for Cations/Anions + Key Parameters Model
D-22 Loadings for PC1, PC2, and PC3 for Cations/Anions + Key Parameters + WWTP Parameters Model
D-23 Scores for PC1 vs. PC2, PC1 vs. PC3, and PC2 vs. PC3 for Cations/Anions + Key Parameters + WWTP Parameters Model
D-24 Number of Families from Macroinvertebrate Studies Conducted across Western United States
D-25 Cumulative Frequency Distributions through Time
D-26 Change in Species Richness: Above and Below Wastewater Treatment Effluent Discharge
D-27 Santa Cruz River, near Nogales - Total Number of Macroinvertebrate Taxa (Lawson, 1995 - Taxa Identified to Lowest Practical Level)
D-28 Santa Cruz River, near Nogales - Average Number of Macroinvertebrate Taxa per Month vs. Total Number of Macroinvertebrate Taxa per Year (Lawson 1995)
D-29 Santa Ana River - Seasonality of Macroinvertebrate Taxa: 1981 (CEC) (Taxa Identified to Lowest Practical Level)
D-30 Santa Ana River - Seasonal vs. Annual Macroinvertebrate Taxa Richness: 1991 (CEC) (Taxa Identified to Lowest Practical Level)
D-31 Santa Ana River - Number of Macroinvertebrate Taxa: 1991. 1998 (CEC) (Taxa Identified to Lowest Practical Level)
D-32 Santa Ana River - Longitudinal Variation in Fish Abundance and Species Richness (CEC)
D-33 Santa Ana River - Annual Variation in Fish Abundance and Species Richness (CEC)
D-34 South Platte River - Number of Macroinvertebrate Taxa: Denver Metro Data, 1988-1993 (Taxa Identified to the Lowest Practical Level)
D-35 South Platte River - Annual Variation in Fish Abundance and Species Richness: Denver Metro Data 1988-1998
D-36 Fountain Creek - Number of Macroinvertebrate Taxa 1980 vs. 1989 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)
D-37 Fountain Creek - Number of Macroinvertebrate Taxa: 1998 vs. 1999 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)
D-38 Crow Creek - Number of Macroinvertebrate Taxa: Taxonomic Identification to Lowest Practical Level vs. Family/Order Level (Chadwick Ecological Consultants 1994)
D-39 Crow Creek - Number of Macroinvertebrate Taxa, 1993 (Chadwick Ecological Consultants 1994) vs. Site Reconnaissance 2000 (Taxa Identified at Family/Order Level)

D-40 Crow Creek - Number of fish Species in Crow Creek, August 1991 (Wyoming Game Fish Data reported by Chadwick Ecological Consultants 1994)
D-41 Terrestrial Habitat Values: Sites Where Effluent Comprises Almost 100% of Instream Flow
D-42 Terrestrial Habitat Values: Sites Where Effluent Comprises Majority of Instream Flow1
D-43 Number of Macroinvertebrates Taxa Above and Below Effluent Discharge (Taxa Identified to Family/Order/Class Level)
D-44 Number of Macroinvertebrate Taxa below Effluent Discharge vs. RBP Habitat Assessment Score (Taxa Identified to Family/Order/ Class Level)
D-45 Percentage of Mayflies, Stoneflies, and Caddisflies (EPT Taxa): Above and Below Effluent Discharges
D-46 Santa Fe River - Number of Macroinvertebrate Taxa: 1994 Study (CDM 1995), Site Reconnaissance 2000, and NMWW (1999, 2000) (Taxa Identified at Family/Order Level)
D-47 Santa Fe River - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): 1994 Study (CDM 1995), Site Reconnaissance 2000, and NMWW (1999, 2000)
D-48 Santa Fe River - Fish Abundance in Samples Collected in 1994 (CDM 1995) and NMED (2000)
D-49 Santa Ana River - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): August 1991, August 1998 (CEC) vs. Site Reconnaissance 2000
D-50 Santa Ana River - Number of Mayfly, Stonefly, and Caddisfly Taxa (EPT Taxa): 1991 vs. 1998 (Taxa Identified to Lowest Practical Level)
D-51 Santa Ana River - Changes in Fish Species Richness: 1991-1998
D-52 Santa Ana River - Changes in Fish Abundance: 1991-1998
D-53 South Platte River - Number of Macroinvertebrate Taxa: Denver Metro 1988-1993 Data vs. Site Reconnaissance 2000 (Taxa Identified to Family/Order Level)
D-54 South Platte River - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): Denver Metro 1988-1993 Data vs. Site Reconnaissance 2000
D-55 Fountain Creek - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa) 1980 vs. 1989 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)
D-56 Fountain Creek - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): 1998 vs. 1999 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)
D-57 Fountain Creek, CO - Mean Number of Mayfly, Stonefly, and Caddisfly Taxa (EPT Taxa), Upstream vs. Downstream of Effluent Discharge, 1980-1999 (Taxa Identified to Lowest Practical Level)
D-58 Linear Regression - RBP vs. Project Habitat Assessment Scores

D-59 Linear Regression - RBP vs. Project Habitat Assessment Scores (Carrizo Creek Removed)

LIST OF TABLES

D-1 Comparison of WWTP Effluent Flows to the 1.5-Year Flood
D-2 Station ID Keys for Box Plots
D-3 PCA Results – Percentage of Accumulated Explained Variance by the Principal Components
D-4 Number of Taxa (Family/Order/Class) Collected from each Reconnaissance Site in Spring 2000
D-5 Number of Taxa (Family/Order/Class) from Selected Historical Records Gathered from Habitat Characterization Study Sites
D-6 Comparison of Taxonomic Groups Known to be Present at Habitat Characterization Study Sites and Taxonomic Categories Required for Calculation of Acute Toxicity Criteria
D-7 Terrestrial Habitat Characterization - Standardized Scores for Each Study Area
D-8 Numbers of Potential Species at Each of the Study Areas, Sorted by Number of Species Restricted to the Riparian Habitats, Restricted to Upland Habitats, or Able to Use Both Habitats
D-9 Number of Potential Species at Each of the Non-Arid Areas, Sorted by Numbers of Species Restricted to the Riparian Habitats, Restricted to Upland Habitats, or Able to Use Both Habitats
D-10 National Criteria to Protect Aquatic Life

APPENDIX D

1.0 DATA ANALYSIS

1.1 INTRODUCTION TO INVESTIGATION ITEMS FOR HABITAT CHARACTERIZATION

This Habitat Characterization Study primarily has used historic information for data analysis. The investigation items presented in this section are a reflection of types of research questions that can be examined with the use of historic data. The goals of these investigation items include describing what characteristics of the arid West environments warrant protection and how arid West streams differ from the more typical water bodies considered in the derivation of national water quality criteria.

The research team developed a plan to investigate the physical, chemical, and biological aspects of arid West watercourses. The investigation focused on examining the physical, chemical, and biological aspects of 10 study areas in the arid West to determine what characteristics make the West different from other regions. The 10 study areas were as follows:

- Salt/Gila Rivers near Phoenix, Arizona
- Santa Cruz River near Nogales, Arizona
- Santa Cruz River near Tucson, Arizona
- Santa Ana River near San Bernardino/Riverside, California
- Fountain Creek near Colorado Springs, Colorado
- South Platte River near Denver, Colorado
- Las Vegas Wash near Las Vegas, Nevada
- Santa Fe River near Santa Fe, New Mexico
- Carrizo Creek near Carrizo Springs, Texas
- Crow Creek near Cheyenne, Wyoming

1.2 HABITAT CHARACTERIZATION INVESTIGATION TOPICS

For each of the investigation topics discussed in this section, the following information is presented:

- explanation of what is being measured
- description of how the item was measured
- evaluation or the results of the measurement

1.2.1 Hydrologic Flow Regimes

Since the hydrological function of a landscape is fundamentally dependent upon climate, topography, and geology (Dunne and Leopold 1978), ecosystems that use the landscape must adapt to the dynamics of these three controls. The fluvial dynamics of the existing and created riverine system, including surface water hydrology and hydraulics, sediment transport and

distribution, fluvial geometry and dynamics, groundwater movement and storage, and soil moisture distribution of arid West streams are the controlling factor for creating physical ecological structure in riparian communities.

The riparian environment has been described as an ecotone, a transition from the aquatic ecosystem of the stream into the adjacent terrestrial communities of the upland region (Gregory et al. 1991). As such, the geometry of western riparian zones represents a dynamic balance between the kinetic energy of the stream and the regional, relatively infinite energy source/sink of the uplands (Mitsch and Gosselink 1993). The communities that use this seasonal-to-dial exchange of energy and matter must be appropriately dynamic; riparian communities are patchy because they are dynamic (Sparks 1990; Szaro 1990), adapting to rapid changes in the hydrologic environment. The first-order patchiness generates a linear string of microenvironments, a "ribbon-like" braid of refugia, offering shade, cover, and water within the relatively harsh terrestrial upland.

This complex combination of temporal and spatial variability must be matched against the equal complexities of water quality regulations. This is because the format for expressing aquatic life water quality criteria consists of concentrations, duration of averaging periods, and average frequencies of allowed excursions. The purpose of the frequency of allowed excursions is to provide an appropriate average period of time during which the aquatic community can recover from the effect of an excursion and then function normally for a period of time before the next excursion. The U.S. Environmental Protection Agency (EPA) has indicated, as a general rule, the purpose of the average frequency of allowed excursions would be achieved if the frequency were set at once every three years on average (EPA 1991).

Current understanding of stream ecology suggests stream communities are structured differently by biotic and abiotic controls along some type of environmental gradient, for example, elevation or temperature. A new generation of models has emerged that attempt to characterize the spatial and temporal variability in streams and use these characterizations as templates to match the biota to the environment. The idea is that as one moves from a frequently disturbed, unpredictable environment to an infrequently disturbed, predictable environment, there will be a predictable change in the attributes of the biotic community. Aquatic communities in "harsh" environments are expected to be trophically simple, have relatively low species richness, and be stable and persistent in the face of disturbance, due to the dominance of flood- and/or drought-resistant taxa. Communities in more "benign" environments are expected be trophically complex, have intermediate or high species richness, and decreased persistence and stability in the face of unpredictable disturbances.

For the most part, streams in the arid West fall in the harsher, unpredictable ranges in the disturbance models discussed. Lowland streams in the desert Southwest are generally characterized by low flows in downcut channels. The channels are created by occasional summer flash floods from localized thunderstorms falling on a sparsely vegetated landscape. More prolonged, less severe floods occur during the winter as a result of larger weather patterns. Flooding on the Western Great Plains is similar to summer patterns in the Southwest; with little winter flooding and generally some flooding as a result of spring snowmelt. More coastal regions (Sierra Nevada, cold desert regions) experience little summer precipitation but often have heavy

snows and/or winter rains resulting in high frequency and higher predictability of flooding. In the interior mountain ranges, the spring snowmelt flood is highly predictable, and flood frequency and flow variability are low. Arid regions in the Great Basin are frequently categorized as cold deserts, departing significantly from the Southwest in seasonal weather patterns (Turner 1994).

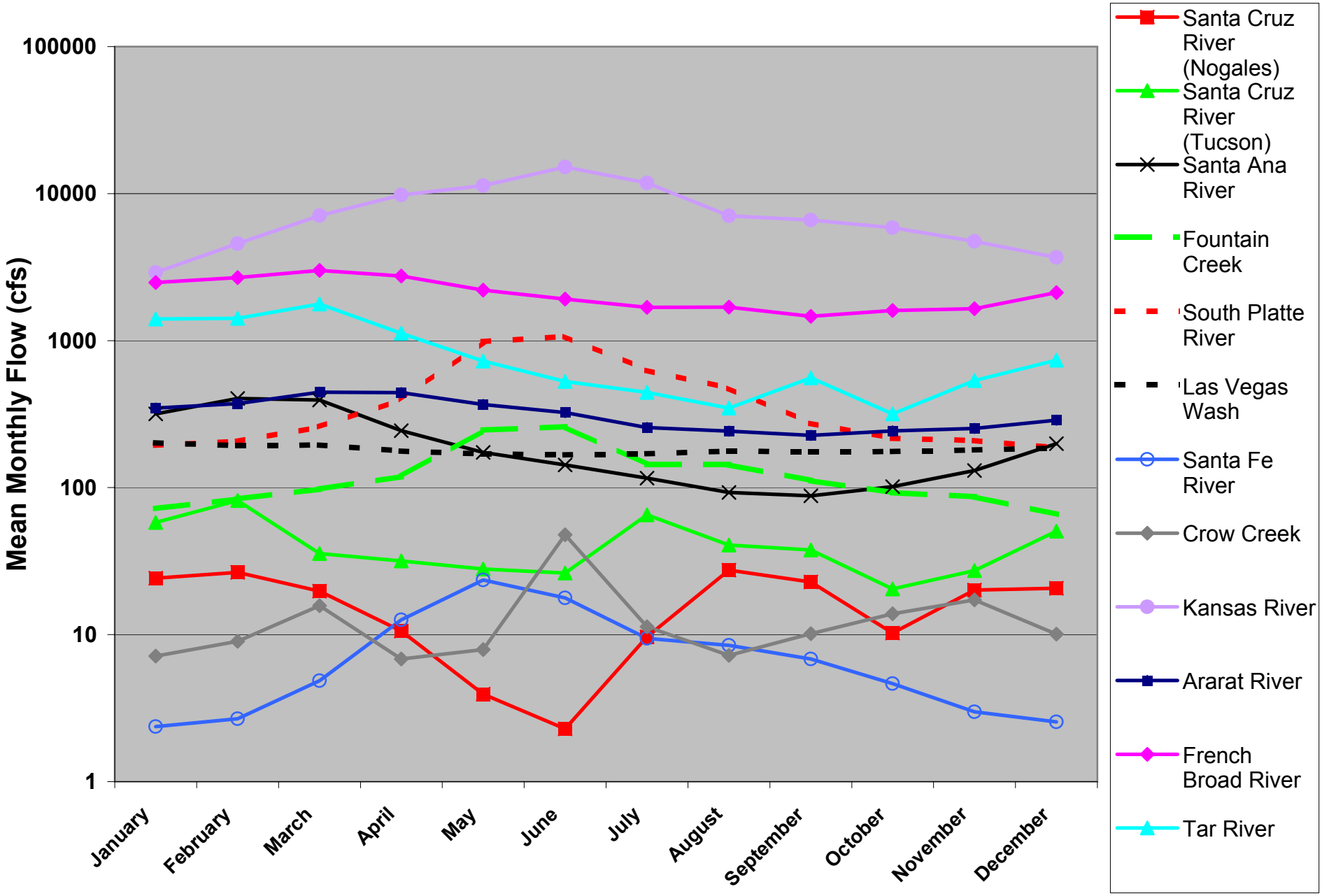
Figure D-1 shows the mean monthly flows downstream of the wastewater treatment plant (WWTP) outfalls for 8 of the 10 sites examined. The two sites not included on **Figure D-1** are the Salt and Gila rivers near Phoenix and Carrizo Creek near Carrizo Springs, Texas. The flow data set for the Salt and Gila rivers has a short period of record that includes very large flooding events, which skews the data set and makes it difficult to compare to other sites. There are no historic flow data for Carrizo Creek. The four non-arid rivers are also included.

The seasonal variation for each stream system is consistent with the previous discussion. Most of the non-arid study areas display much less seasonal variability than the arid study areas, as measured by the standard deviation of the monthly means. The pattern of the curve is also illustrative, showing the distinction between streams that peak in late spring due to snowmelt (for example, Fountain Creek and the Santa Fe River) and streams that receive flow from monsoonal runoff in July and August. The Santa Ana River, on the other hand, shows considerably less of a summer increase, compared to the South Platte River.

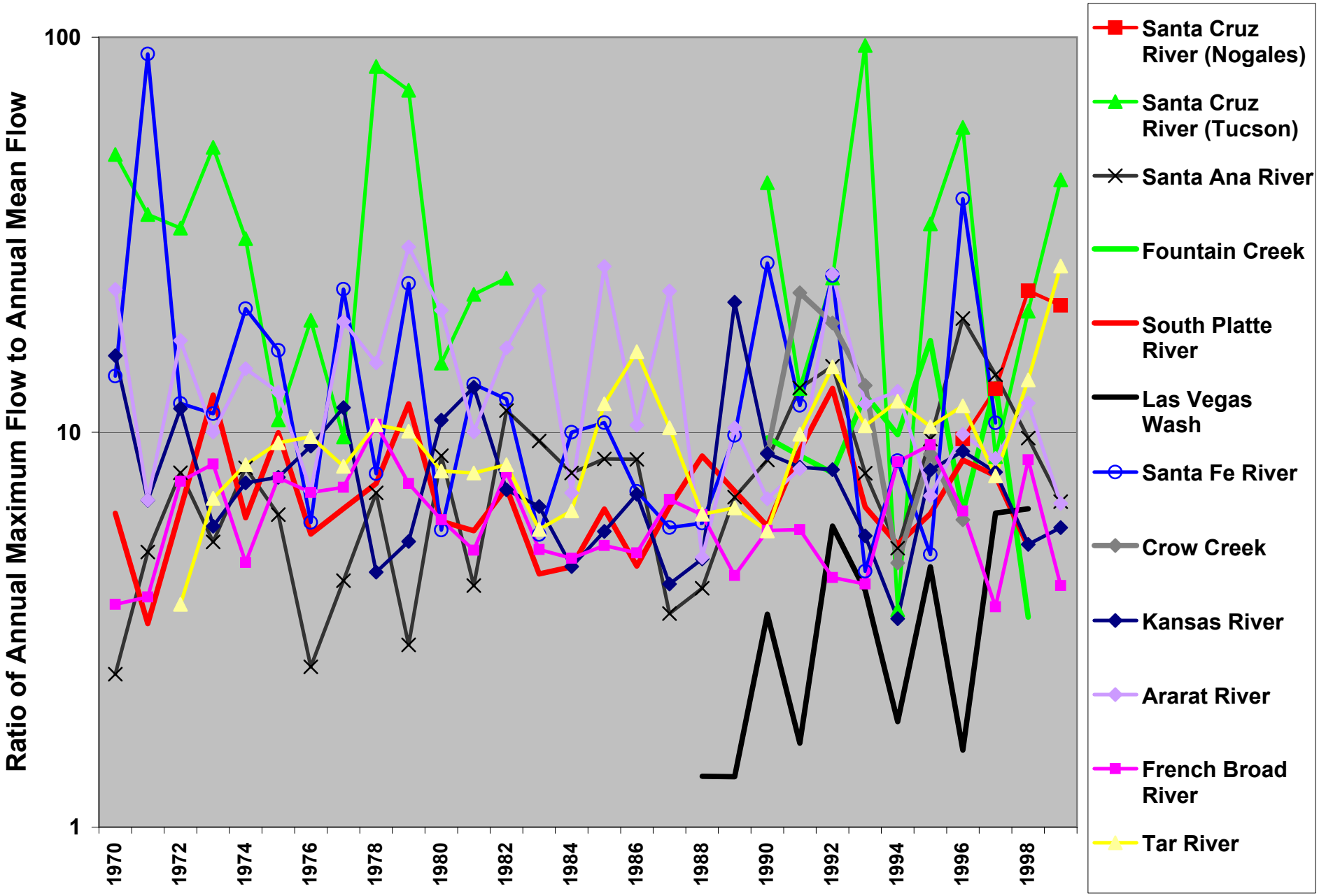
The ratio of the annual maximum flow versus annual mean flow was used to quantify the "flashiness" of these arid West stream systems (**Figure D-2**) and in the stream runoff to rainfall ratio. The four non-arid streams generally convey flows that have low peak to mean discharges, relative to several of the arid West streams. The Ararat River in North Carolina is a headwaters stream and has higher ratios than some of the arid West and the other three non-arid streams.

The dynamics of western streams can be further explored by comparing flows upstream and downstream of the WWTP. Many of the streams show a lower peak-to-mean ratio below the WWTP. There can be many reasons for this, including the operation of flood control, stormwater systems, and diversion structures; however, the steady supply of wastewater would tend to increase this stability. One exception in the data is the Santa Fe River, which becomes more "flashy" below the WWTP. This might be due to the large watershed captured between the plant and the U.S. Geological Survey (USGS) gaging station at La Bajada.

Because the variability of flooding patterns over the entire range of the arid West is great, the vast majority of the fauna has evolved with either possessing morphological, behavioral, or life history adaptations that allow them to recover very quickly from floods. This section summarizes an extensive literature review conducted on the effects of flooding on the aquatic biota in lotic systems in the arid western United States. The goals of the review were to specifically identify the response and subsequent time to recovery of periphyton, aquatic invertebrates, and fish in these arid regions, and to provide a number of ecological concepts important to interpretation of data collected as part of the Arid West Habitat Characterization Study. The complete literature review, with citations, is presented as an appendix titled Overview of the Effects of Flooding on the Aquatic Biota in Lotic Systems in the Arid Western United States (**Appendix G**).



Mean Monthly Streamflow
Downstream of WWTP Outfalls
Figure D-1



Ratio of Annual Maximum Flow to Annual Mean Flow
Downstream of WWTP Outfalls
Figure D-2

Periphyton communities appear to be the most easily disturbed by shear stress and scouring, but studies that have closely monitored recovery seldom report recovery times longer than one month once seasonality factors are taken into account. Periphyton exhibit good dispersal abilities, short generation times, and flexible life history strategies that all allow rapid recolonization from floods. Studies in the arid West indicate that recovery can take as little as five days and up to three months, depending on the season and frequency of flood events.

The life history characteristics of macroinvertebrates in unpredictable flood environments demonstrate very rapid development times and prolonged reproductive periods (year round when temperatures are moderate during winter). Despite varying measures of recovery, the literature suggests that recovery in frequently flooded systems in arid regions can occur in approximately two months. Values in the literature range from 7 to 52 days, depending on flood frequency and season. In more predictable flood regimes, timing of disturbance may result in a prolonged recovery. A discrepancy in the timing of disturbance could include a drought period where typically the stream experiences a flood regime. The recovery from disturbance may take longer because there are fewer pathways (e.g., adults or eggs) for organisms to recolonize the disturbed reach. Reported times of more than four months are uncommon for invertebrate communities, but some individual taxa may take up to two years to recover.

The impact of flooding on arid riparian plant communities has been less extensively studied (Harris 1986; Hupp and Osterkamp 1985; Junk et al. 1989; Sparks et al. 1990; Stromberg et al. 1991). The most complete work on an arid, intermittent to ephemeral stream was conducted by Stromberg and co-workers (1991) on a protected and monitored reach of the Hassayampa River in Arizona, about 37 miles northwest of Phoenix (Stromberg 1997; Stromberg et al. 1997). These workers determined that the flood stage and duration, soil moisture retention, and timing of flood events conspired in a complex manner to select between Fremont cottonwood (*Populus fremontii*), Goodding's willow (*Salix gooddingii*), and salt cedar (*Tamirix chinensis*).

These results have been applied to the management of salt cedar infestation on the upper San Pedro River, about 62 miles southwest of Tucson (Bagstad and Stromberg 2001; Stromberg 1998). Long-term lowering of the elevation of the water table along the San Pedro River has been attributed to overpumping of the regional aquifer. Although salt cedar appears to be primarily tied to depth to groundwater, general plant diversity is a function of floodplain topography and seasonal rainfall patterns. This investigation and the Hassayampa River studies should be extended with a quantitative accounting of the subsurface water balance at the studied sites.

The instream and riparian ecological studies described above are profoundly useful in directing future hydrological research on riparian systems. However, a great deal remains before these studies can be generalized and used for making flow management decisions. A step in the right direction was developed by Richter and co-workers (1996, 1997, and 1998), with the Nature Conservancy, who have initiated a system of hydrologic indicators appropriate for stream ecosystem preservation. Specifically, these indicators determine if management changes have altered the natural hydrologic flow regime sufficient to cause ecosystem impacts.

Again, the Nature Conservancy system is focused on direct instream changes and only indirectly addresses changes in the riparian community. Moreover, no such system has been developed for ephemeral or effluent-dependant streams. Without a full understanding and quantitative accounting of the interacting hyporheic, alluvial, deep-regional, and soil-moisture hydrologic regimes, it is impossible to know the general response of the riparian community to instream flow alteration. These research topics, along with integrated work on the fluvial dynamics and plant ecology, remain a major research gap in understanding the ecology of effluent-dependant waters.

In addition to these direct impacts on biota, flood hydrology can generate secondary impacts on the riparian and stream ecosystem by simply moving the active channel. The position of a thalweg within an ephemeral floodplain is a random function of individual flood events. Over short time periods, the active channel can be expected to change location dramatically. Despite this, the channel currently in use will be reflective of the most frequent flood event.

For a perennial stream, the location of the active channel will be less variable and the cross-sectional area of an active channel also will be determined by the most frequent flood. For effluent-dependent streams, which are modified from ephemeral to perennial, extant fluvial theory is insufficient to predict the behavior of the channel over time sufficient to evaluate the impact of this change on the habitat. Geomorphic data from the 10 study areas suggest that most of the channels are entrenched from the WWTP outfall downstream. This may be the result of discharging sediment-free effluent into a system that is capable of transporting much higher concentrations of suspended material. For example, the Santa Fe River receives clear effluent at the WWTP outfall that immediately is conveyed in an incised channel cut below the upstream grade of the formerly ephemeral stream. Within a 1.5 mile downstream, the water becomes turbid. Despite this observation, incision also may be attributable to a strong disparity between the effluent-dependent discharge and the upstream, channel-forming discharge originally established by the stream.

The shape and size of the ephemeral upstream channel of the Santa Fe River and all other reaches studied is maintained by the channel-filling floods that occur most frequently. In other words, if these stream systems have been allowed to come to a temporary steady state with respect to sediment removal, transport, and deposition, the cross-sectional geometry of the active channel reflects this moment in time. A particular reach should reflect a balance between the physical character of the channel substrate, banks, bars, and other sources of sediment and the tractive energy imposed upon them by the stream flow. This principle, developed by a number of workers in fluvial geomorphology and river engineering (Andrews 1980; Emmet 1975; Leopold et al. 1963; Schumm 1956), has led to the designation of a “channel maintaining flow, or flood.” The discharge corresponding to this event is sometimes referred to as bankfull discharge.

The determination of the channel geometry that is maintained by the most frequent flood events can be determined from geomorphic field evidence. This detailed analysis was not performed at the 10 study areas. Instead, a relationship was used from Dunne and Leopold (1975) who demonstrated, based on a variety of streams across North America, that the channel-forming or bankfull discharge generally corresponds to the 1.5-year recurrence interval event for a particular

watershed. Moody (2000) recently determined that this is a good estimate for the arid West, although some minor variation (1.1 to 1.8 years) occurs from stream to stream.

However, it is unclear if the concept of bankfull discharge has any meaning in an ephemeral flow. The correlation between Leopold's 1.5-year storm and the shape of measured streams in the arid West may only reflect the high likelihood of the channel experiencing the 1.5-year flood at any particular point in time. If this is the case, the estimate may still be of value in measuring the channel that a perennial flow would be expected to form once it had been converted into an effluent-dependant stream. In other words, it might be useful to compare the "perennial" flow emanating from the WWTP to the flow of the 1.5-year event. Will the stream channel be over or undersized by the addition of effluent?

Table D-1 is a compilation of 1.5-year discharges extracted from partial volume or annual peak discharge records for USGS stream gaging stations at each of the nine study areas that had records. These values are set against the 1999 average daily discharge for each of the WWTPs occurring along the reference reaches. These data indicate that the effluent-dependant stream is undersized to the 1.5-year discharge for six of the reaches, about equal for two (Crow Creek and Santa Ana) and oversized in only one case (Las Vegas Wash). No flow data are available for Carrizo Creek but inspection of the area suggests that the creek is flowing very close to bankfull discharge with no contribution from the upstream watershed. As such, the Carrizo Springs WWTP effluent discharge would be considered approximately equal to the 1.5-year flood.

Table D-1
Comparison of WWTP Effluent Flows to the 1.5-Year Flood

Stream	1.5 year flood m^3/sec	Total effluent m^3/sec	Individual WWTP flows m^3/sec	
Crow Creek	1.13	0.437	0.16	0.27
Fountain Creek	80.6	1.95	1.95	
Las Vegas Wash	3.00	5.87	3.14	0.38
Salt	265	4.05		
Santa Ana	10.8	3.24	1.88	1.36
Santa Cruz-Nogales	50.5	0.479	0.48	
Santa Cruz-Tucson	47.5	2.06	0.99	1.07
Santa Fe	19.3	0.241	0.24	
South Platte	86.9	6.48	6.48	

Based only upon these relations, it seems likely that for the greatly undersized effluent discharges there will be little impact on the geomorphic stability of the reach. The channel maintaining flood appears to be still capable of defining the basic cross-sectional geometry of the reach, all else equal. For the reaches in which the effluent-dependant stream is approximately equal or oversized with respect to the 1.5-year flood, it is less clear what the expected fluvial response will be. It is possible that the effluent-dependant stream will become the channel-maintaining flow and that the runoff from the 1.5-year event will now encroach upon the floodplain. These frequent floods also may cut new channels into the floodplain. Longer return floods might also increase in depth and velocity.

It is not clear how the change in the geomorphic stability will impact the recovery time for biota in the effluent-dependent streams. If the introduced effluent forces a change in channel dimensions, a period of adjustment may be initiated while the community responds to the new hydrologic regime. Possibly the more disruptive condition will be one in which the effluent discharge is approximately equal to the channel-forming discharge. Such systems might oscillate in behavior as random variations in flow cause constant changes in the size, shape, and/or location of the active flow. This could be analogous to the condition in which the system is under continuous disturbance.

Studies summarized by EPA (1991) indicate that rather than requiring more time to recover from the effects of additional anthropogenic disturbances, lotic communities with high natural background disturbance frequencies are actually predisposed to recover more rapidly because only species that are able to recolonize and reproduce quickly, or perhaps to avoid disturbances, can persist there. This does not imply that they are more resistant to novel anthropogenic disturbances with which they have had no previous evolutionary experience; it only implies that they are predisposed to recover quickly once the disturbance is gone. The question then is how frequently aquatic communities can experience these additional disturbances (excursions of criteria) without being unacceptably affected. The results of the literature search described in this section highlight the recovery periods of western lotic systems and indicate generally that the time frame for recovery is considerably less than three years. Potential areas for further analysis include pairing the excursion of criteria and the recovery periods from natural disturbances in establishing an average frequency of allowed excursions for the western United States.

In conclusion, our examination of this part of the investigation topics seems to have generated more questions than answers and significant data gaps remain. It seems clear that the question of disturbance in both lotic and riparian communities needs to be more mechanistically tied to flood flow hydrology if general predictive models are required. It also seems clear that current water quality standards criteria are based on recovery times that do have wide support from arid riparian ecology. Western systems appear to be more tolerant and more species-specific in their response to flooding. In short, ecological communities seem to have adapted to hydrologic events that their human counterparts would label disastrous.

1.2.2 Concentrations of Major Chemical Constituents in the Arid West

The chemical composition and ionic strength of an aqueous solution can have a significant effect on the organisms inhabiting that solution. This fact has been understood for a long time. In general, ionic strength effects can act to structure or limit an aquatic community, in terms of species composition, abundance, and other characteristics. The most basic example of this is why freshwater fish cannot survive in salt water and, conversely, why most marine fish cannot survive in fresh water. Ionic strength effects are also responsible for the lack of fish in the Great Salt Lake of Utah (Sigler and Sigler 1987). In fact, in an experiment often credited as the first toxicity test, Aristotle transferred freshwater fish into seawater to observe the effect (Cairns 1986). This first toxicity test was, in essence, a measure of the effects of ionic strength on the exposed fish. A literature search was conducted on the effects of ionic strength on aquatic organisms. Results of this search are summarized in the following paragraphs.

A number of studies exist for aquatic organisms, primarily those used in whole effluent toxicity testing, showing that an excess of major ions in solution can create a toxic situation. These excessive concentrations most likely overwhelm the osmotic capacities of the organisms, resulting in ionic strength toxicity. Notably, many of these studies are from sites or locales in the arid western United States, reflecting the regional influence of water quality in this part of the United States. These studies are summarized in **Appendix I**.

Ionic strength effects are generally evaluated using a variety of water quality measurements, most commonly conductivity, total dissolved solids (TDS), salinity, and sometimes alkalinity or hardness. Based on this preliminary review, atypical ratios of major ions or elevated concentrations of major ions in solution may be toxic to aquatic organisms. The effects of these solutions may be straight toxicity to sensitive species or a structuring effect on the aquatic community. Recently, models have been produced to effectively predict toxicity due to major ion effects, which are, in essence, ionic strength effects.

As a measure of how chemical composition and ionic strength may affect arid West streams, the project team compared the water chemistries from the 10 study areas (arid sites) to other freshwater systems elsewhere in the country using three methods. The first method compared arid stream chemistries with standard whole effluent toxicity (WET) testing water chemistry. The second method compared the arid stream chemistries to toxicity database water chemistries used in deriving aquatic life criteria for ammonia, cadmium, copper, and zinc. The third method compared the arid stream chemistries with the water chemistries from selected non-arid streams. Selection of these streams was discussed in **Appendix B**. In addition, the relative impact of WWTP discharge on the arid versus non-arid streams was evaluated. When a non-arid stream is incorporated into the data analysis, the comparisons discussed in the remainder of this section will be used to assess whether differences exist between arid and non-arid water chemistries. Conclusions regarding the differences in chemical composition are provided in **Section 1.2.2.4**.

1.2.2.1 Comparisons with WET Test Composition

Freshwater WET tests are usually conducted using the zooplankter *Ceriodaphnia dubia* or *Pimephales promelas* (fathead minnows) using moderately hard synthetic (MHS) freshwater as a control and diluent. MHS water is obtained by adding the following salts to 20 liters (L) of deionized water, as follows:

Salt	Amount/20 L
$MgSO_4$	1.20 g
$NaHCO_3$	1.92 g
KCl	0.08 g
$CaSO_4 \bullet 2H_2O$	1.20 g

The WET chemistry was added to Piper diagrams for comparison with the arid study area stream data. The Piper diagrams characterize the relative percentages of the major cation (Ca, Mg, and Na+K) and major anion (Cl, SO_4, and CO_3+HCO_3) compositions in terms of percent millequivalents per liters (% meq/L). These diagrams are useful for evaluating differences in major cation and anion compositions. **Appendix B** contains a description on how to interpret Piper diagrams.

The Piper diagrams for the seven arid study areas with major cation and anion data are shown on **Figures D-3 through D-9**. In addition, a Piper diagram that combines all seven arid study areas is shown on **Figure D-10**.

The relative proportions of major cations and anions differ between the WET composition and each of the arid streams. As shown on **Figure D-10**, the WET composition contains a higher percentage of Mg (35% meq/L) than the arid streams (7 to 25% meq/L) and corresponding lower percentages of Ca (25% meq/L) and Na+K (40% meq/L) than the arid streams (25 to 42 and 43 to 63% meq/L, respectively). The WET composition also contains a lower percentage of Cl (2% meq/L) than the arid streams (15 to 70% meq/L), and corresponding higher percentage of SO_4 (60% meq/L) relative to the arid streams (15 to 57% meq/L). The percentage of CO_3+HCO_3 for the WET composition (40% meq/L) falls approximately in the middle of the range for the arid streams (10 to 72% meq/L). The WET composition is also lower than the arid streams with regard to TDS concentrations, as shown by the size of the calculated TDS circles on the Piper diagram displayed on **Figure D-10**. The radius of the circle surrounding the average cation and anion concentration data point is an indication of the TDS concentration.

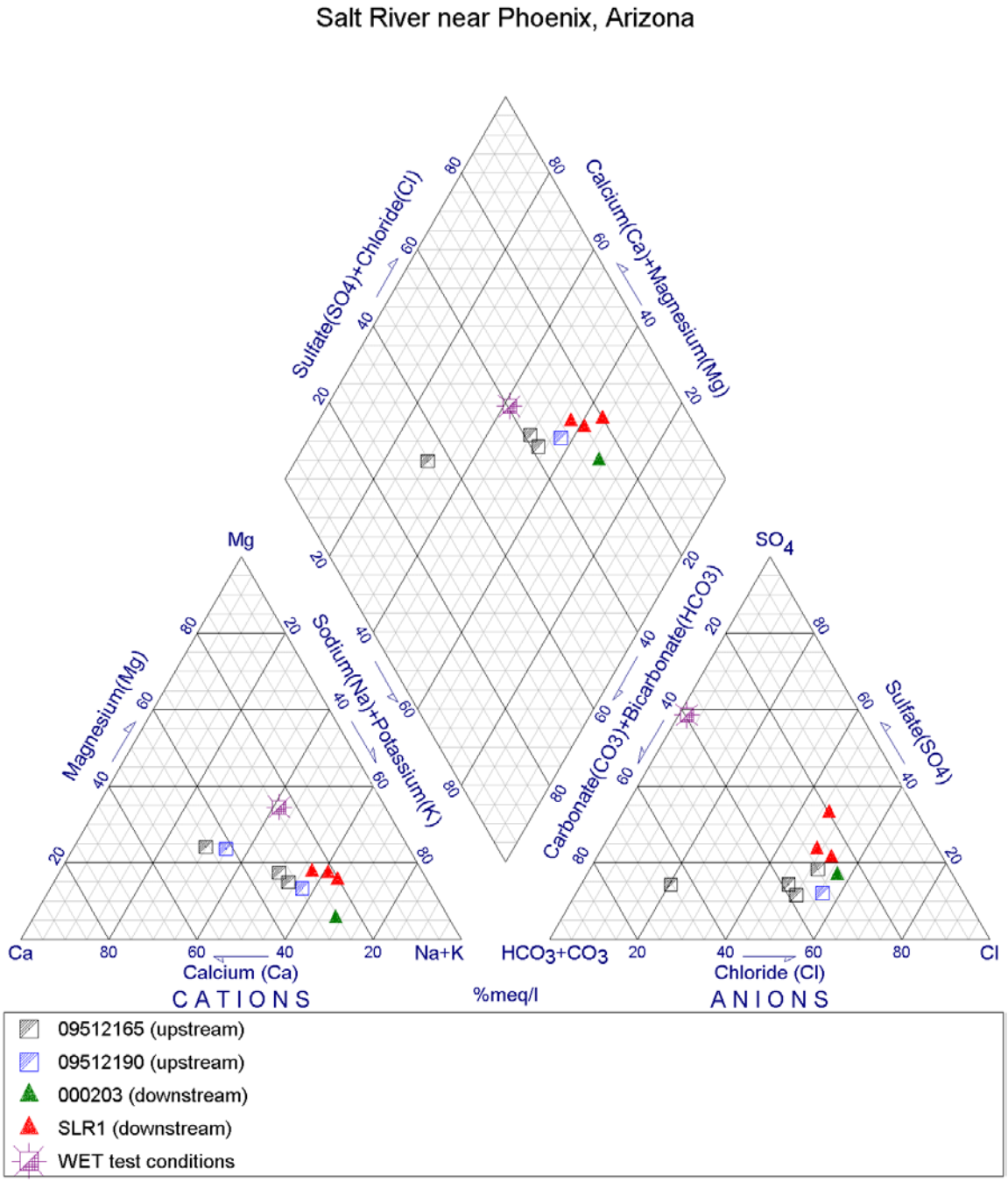
Figure D-10 also shows the composition of two non-arid streams. These data are evaluated in **Section 1.2.2.3**.

1.2.2.2 Comparisons with Toxicity Test Conditions

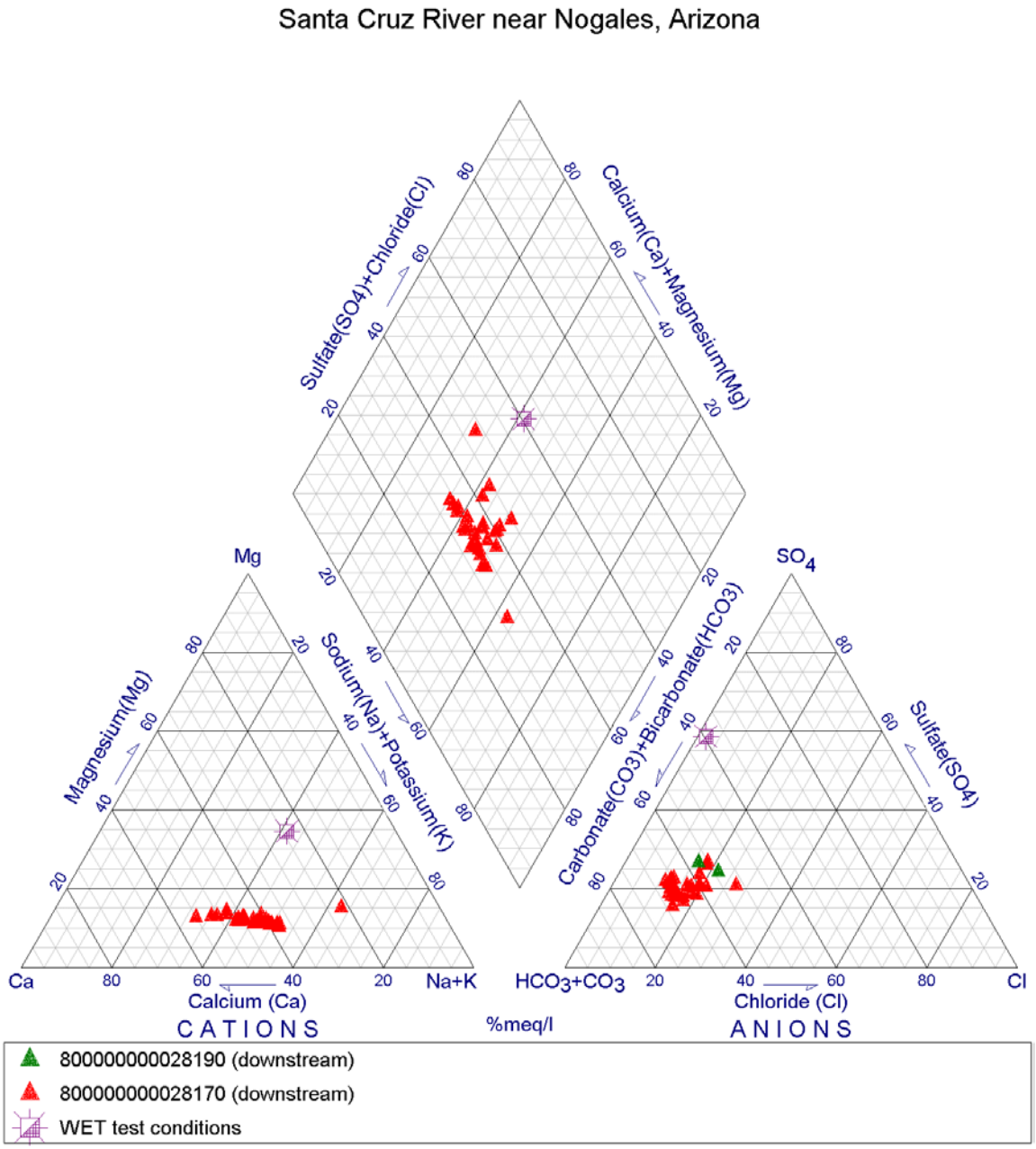
To summarize the data needed for characterization of water quality conditions underlying the EPA water quality criteria, original references from the criteria documents were reviewed. This included references from the cadmium, copper, zinc, and ammonia criteria documents, as well as the 1995 updates to these documents. The list of these documents is contained in **Appendix K**.

The water chemistries summarized from these documents were compared to the water chemistries of 9 of the 10 study areas (arid streams) using box plots showing the distributions of TDS, conductivity, hardness, alkalinity, dissolved oxygen (DO), pH, and ammonia measured upstream and downstream of the WWTP. **Table D-2** provides a key linking the numbering system on the box plots to the station ID. These box plots are shown on **Figures D-11 through D-17**. The components of the box plots are provided on **Figure D-1**. The toxicity test data are shown on these figures as Station 68. Note that toxicity test data were not available for TDS (**Figure D-11**) and ammonia (**Figure D-17**).

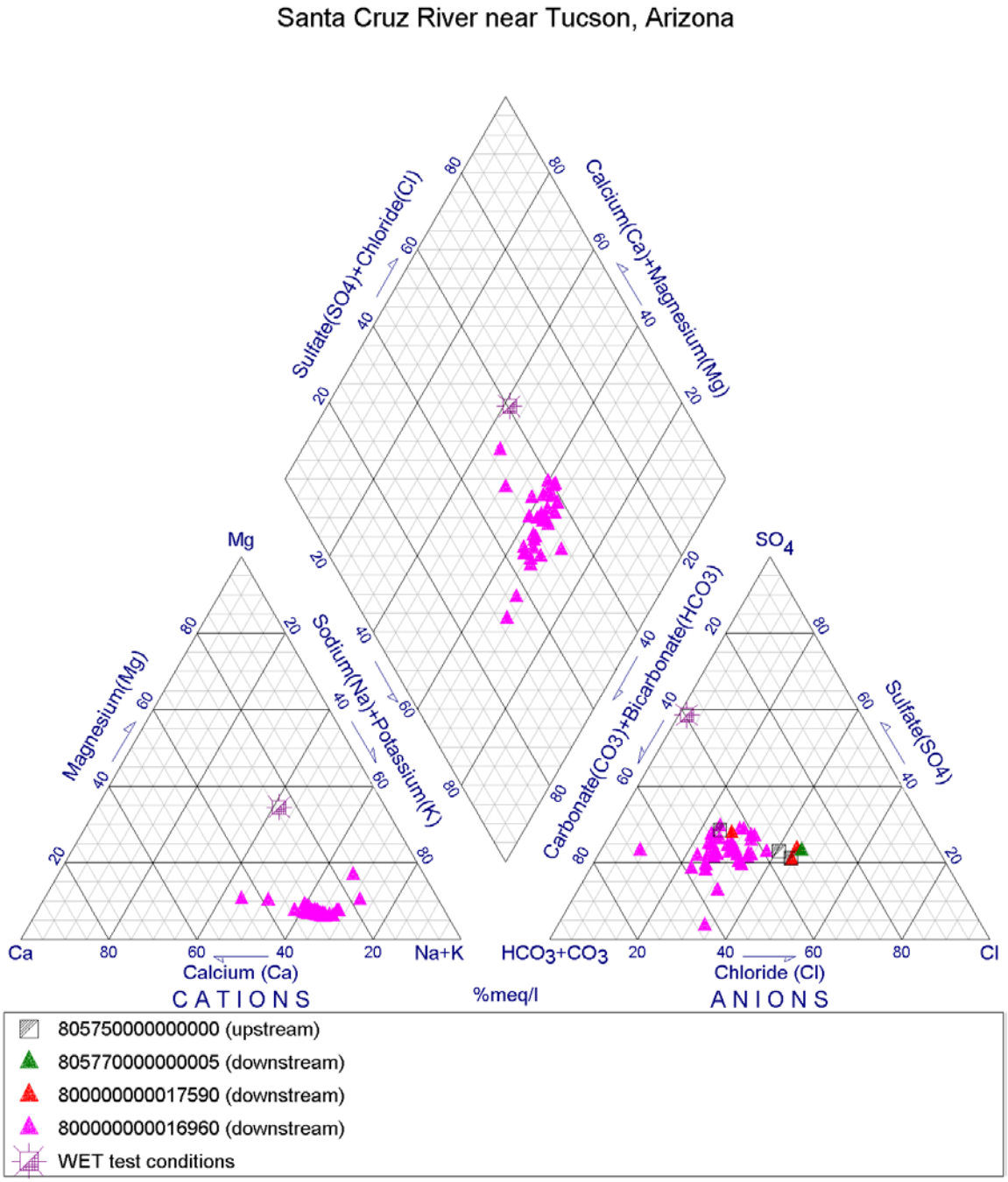
As shown on **Figures D-12, D-13, and D-14**, conductivity, hardness, and alkalinity are generally lower in the toxicity testing dilution waters relative to the arid streams. Hardness (**Figure D-13**) typically ranged between about 50 to 200 milligrams per liter (mg/L) for the toxicity tests, compared with a range of about 100 to 500 mg/L for most of the arid streams. Hardness was much higher in the Las Vegas study area (600 to 900 mg/L). Alkalinity (**Figure D-14**) typically ranged between about 25 to 175 mg/L for the toxicity tests, compared with a range of about 50 to 300 mg/L for most of the arid streams. Conductivity (**Figure D-12**) typically ranged between about 0 to 500 micro mhos per centimeter (µmhos/cm) for the toxicity tests, compared with a range of about 500 to 1,200 µmhos/cm for most of the arid streams. Conductivity was much higher in the Las Vegas study area (2,000 to 3,000 µmhos/cm). As shown on **Figure D-15**, DO



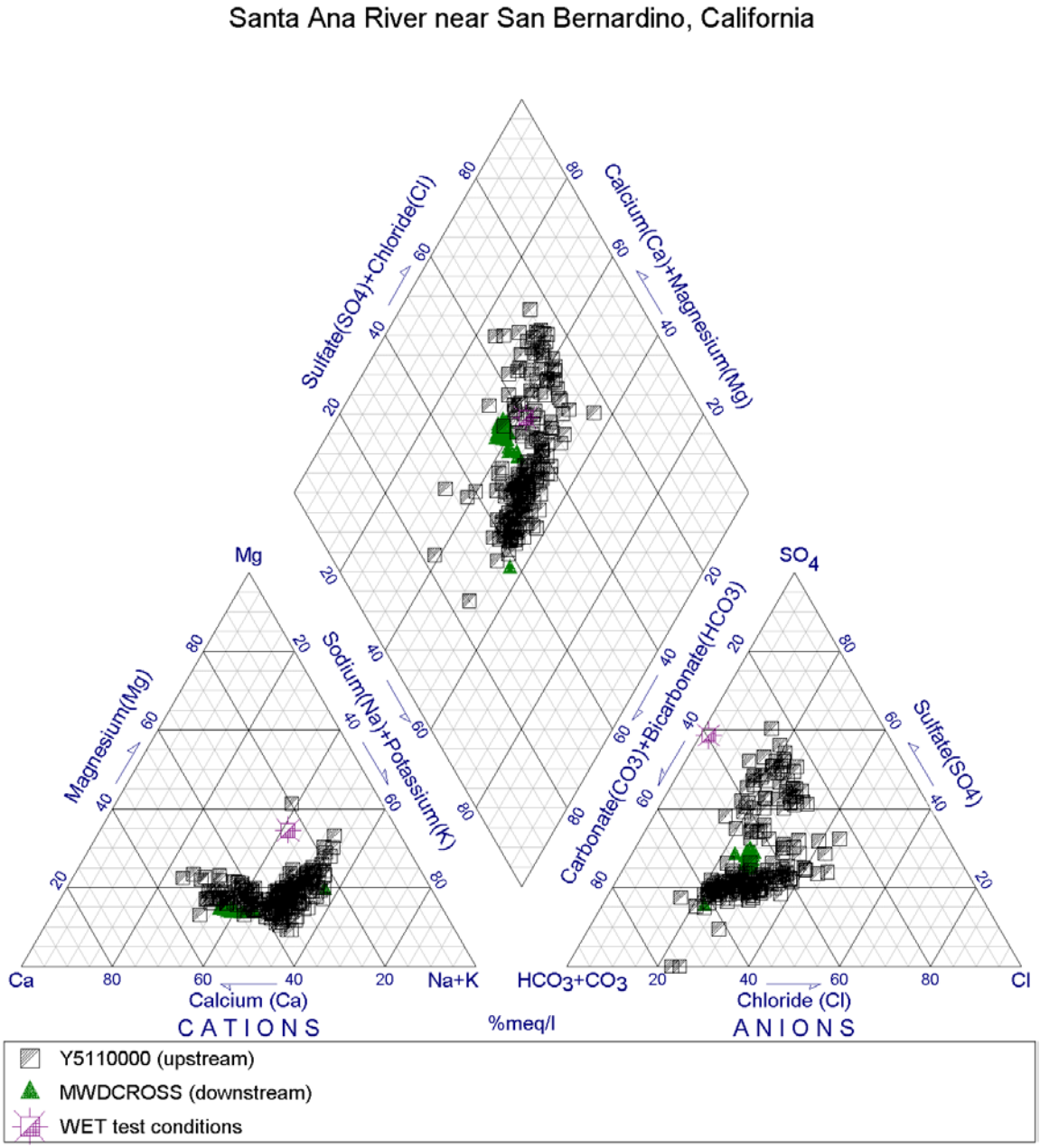
Piper Diagram for the Phoenix Study Area
Figure D-3



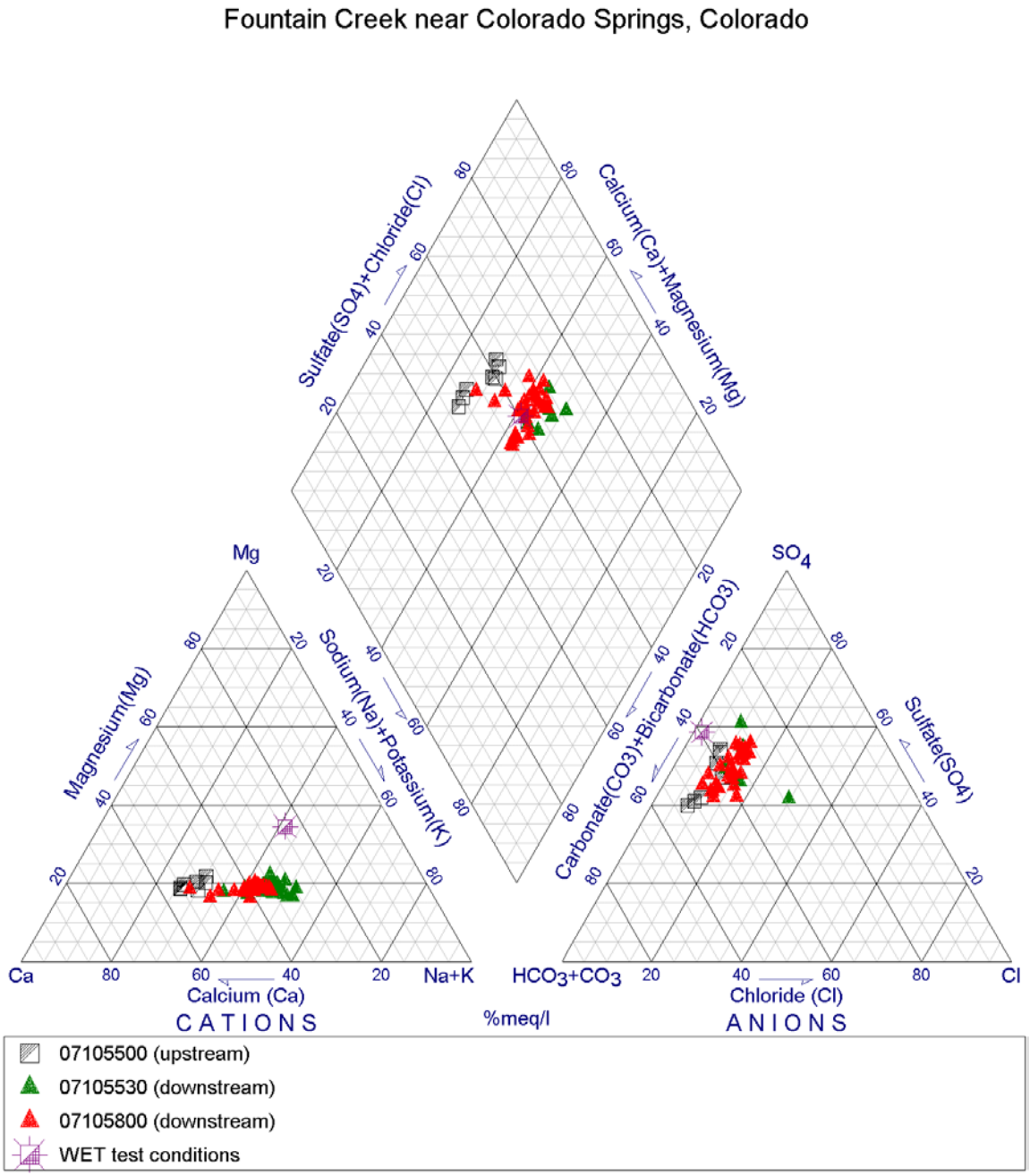
Piper Diagram for the Nogales Study Area
Figure D-4



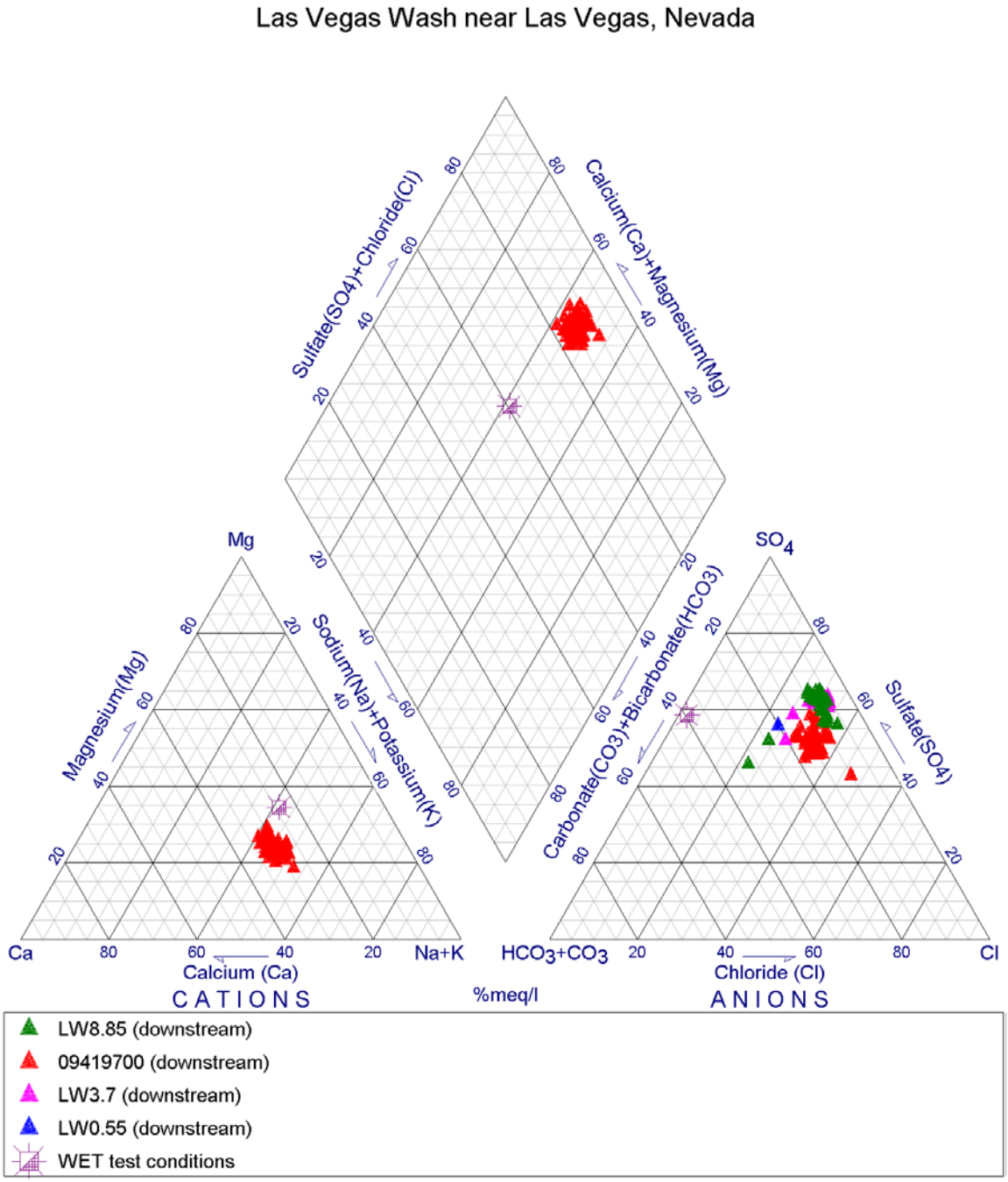
Piper Diagram for the Tucson Study Area
Figure D-5



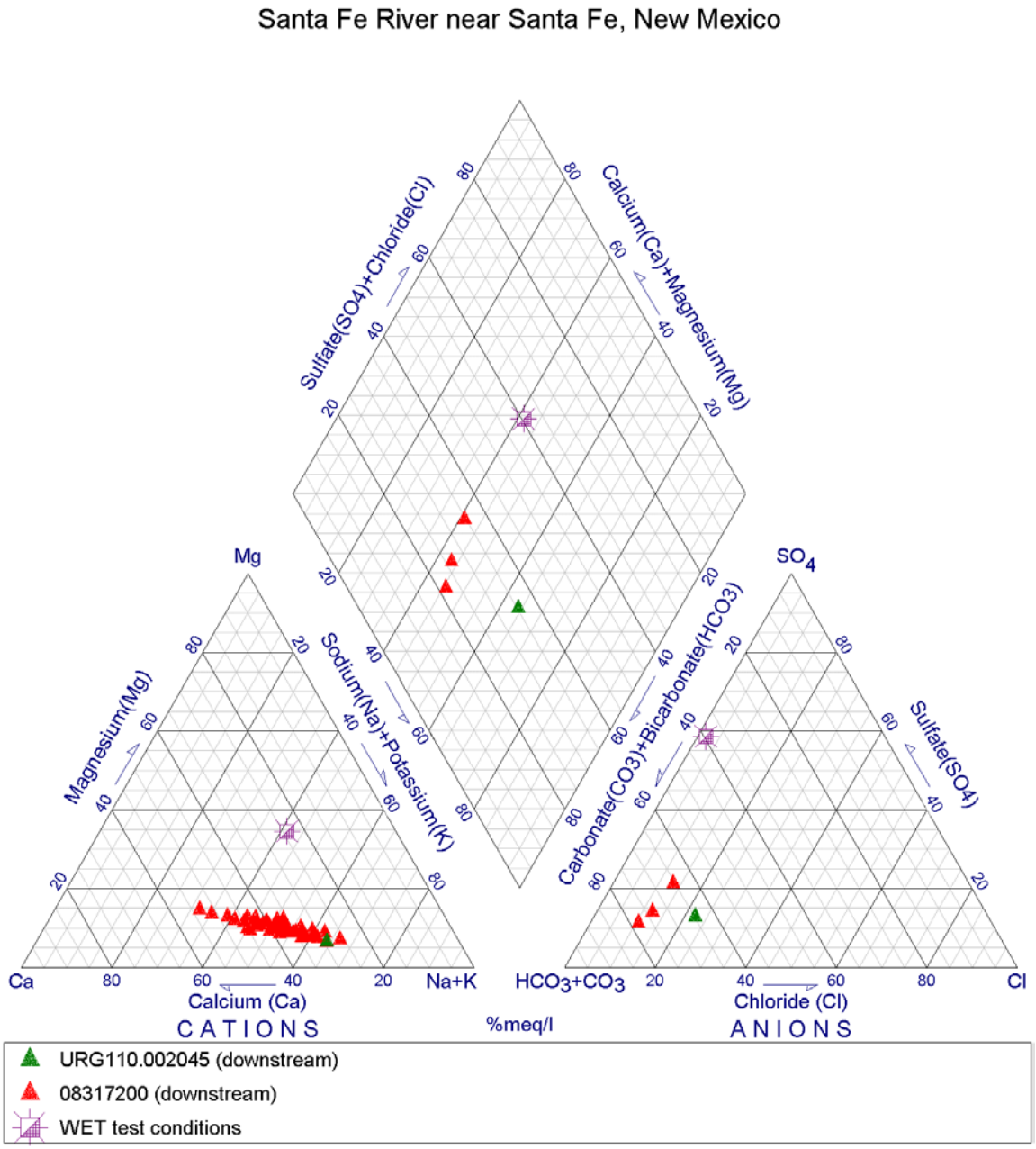
Piper Diagram for the Santa Ana Study Area
Figure D-6



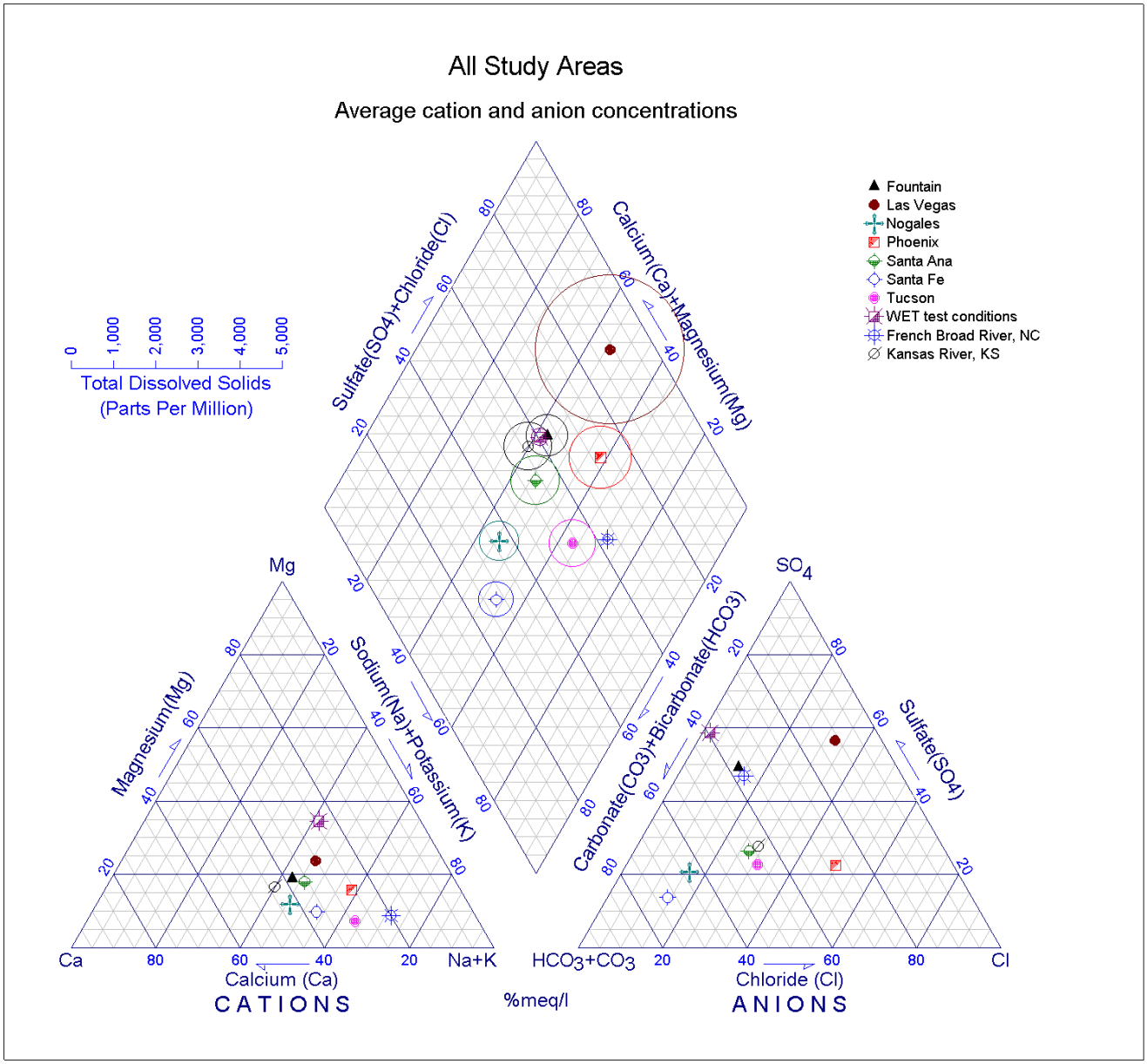
Piper Diagram for the Fountain Creek Study Area
Figure D-7



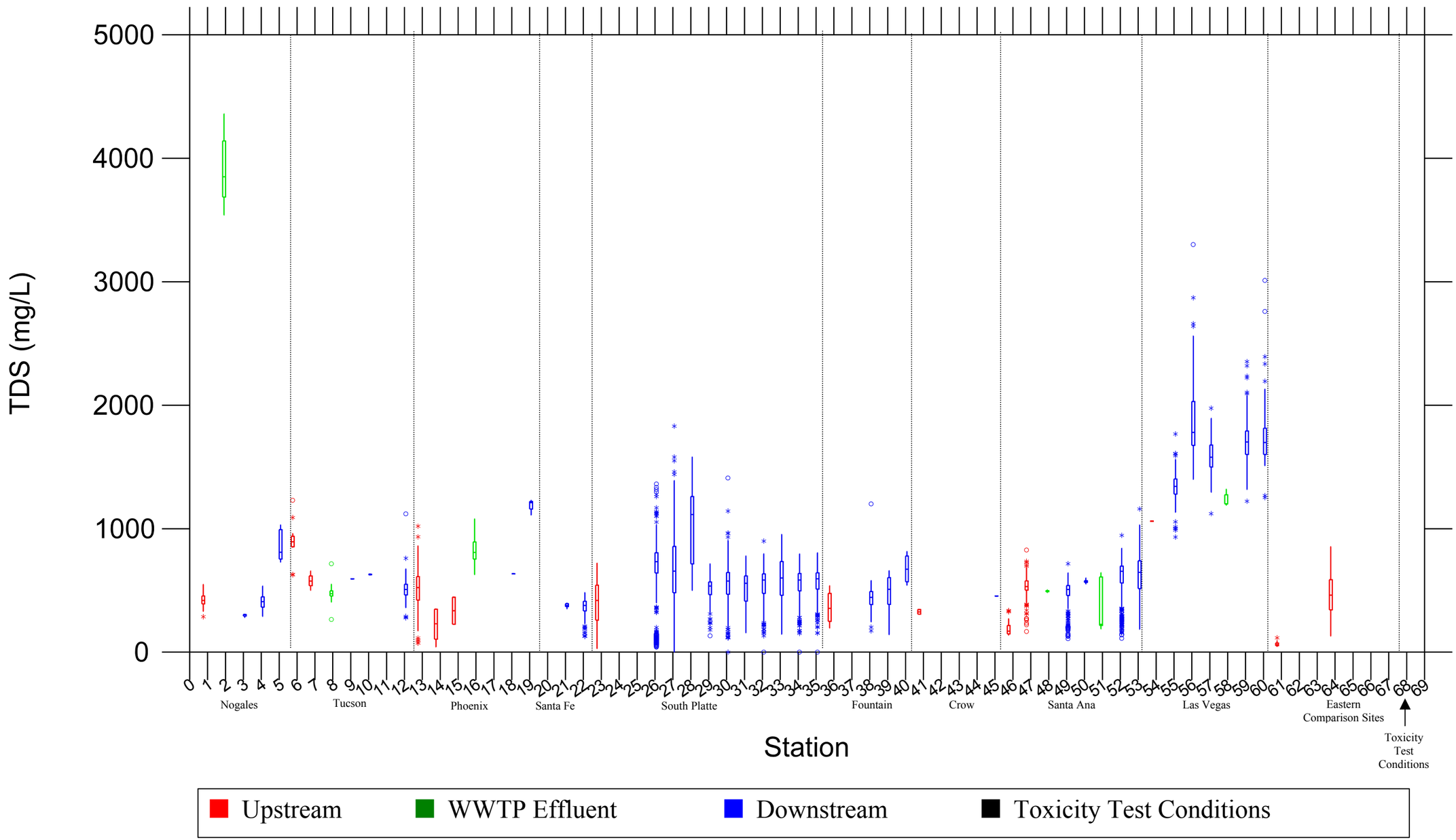
Piper Diagram for the Las Vegas Study Area
Figure D-8



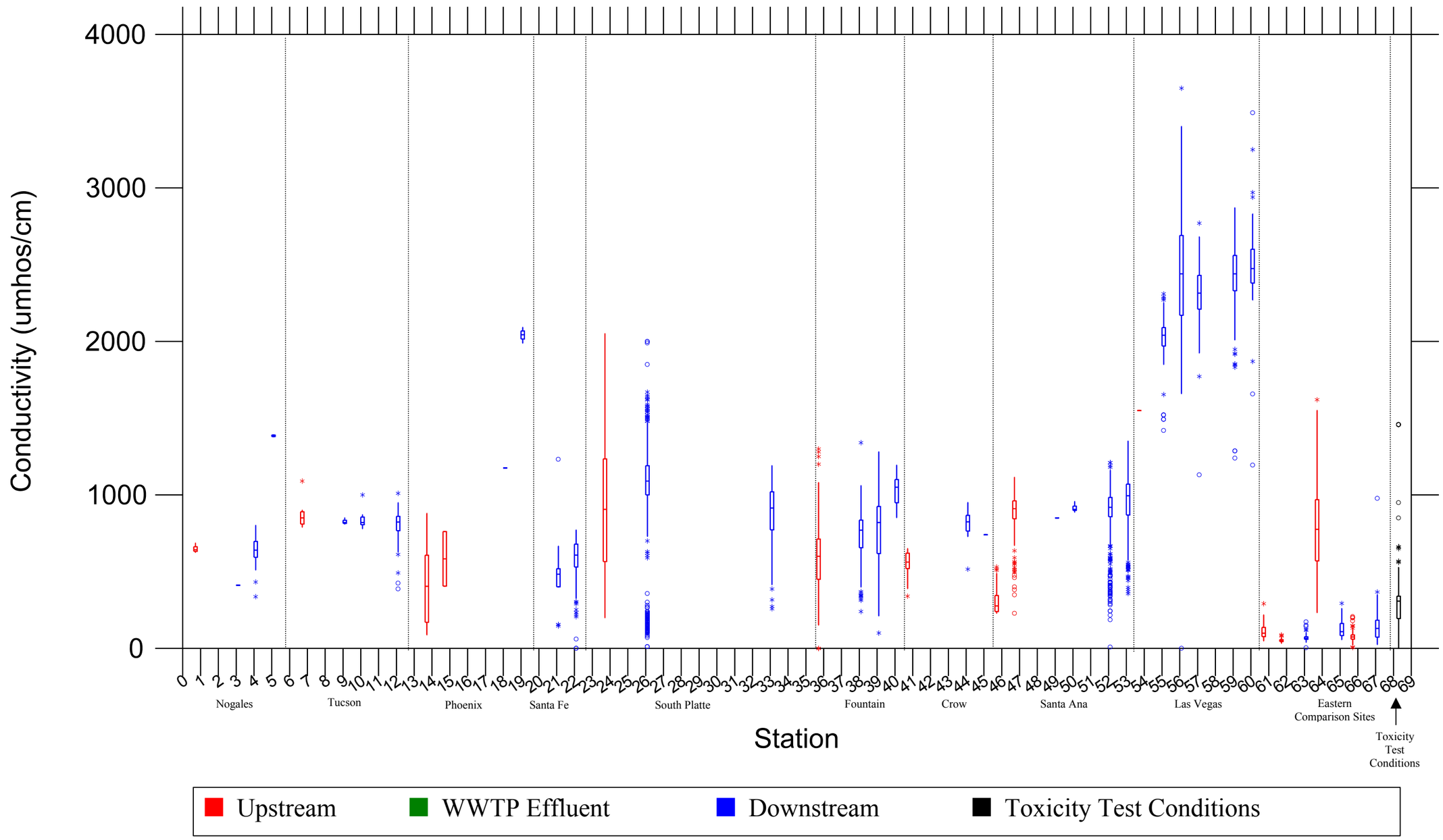
Piper Diagram for the Santa Fe Study Area
Figure D-9



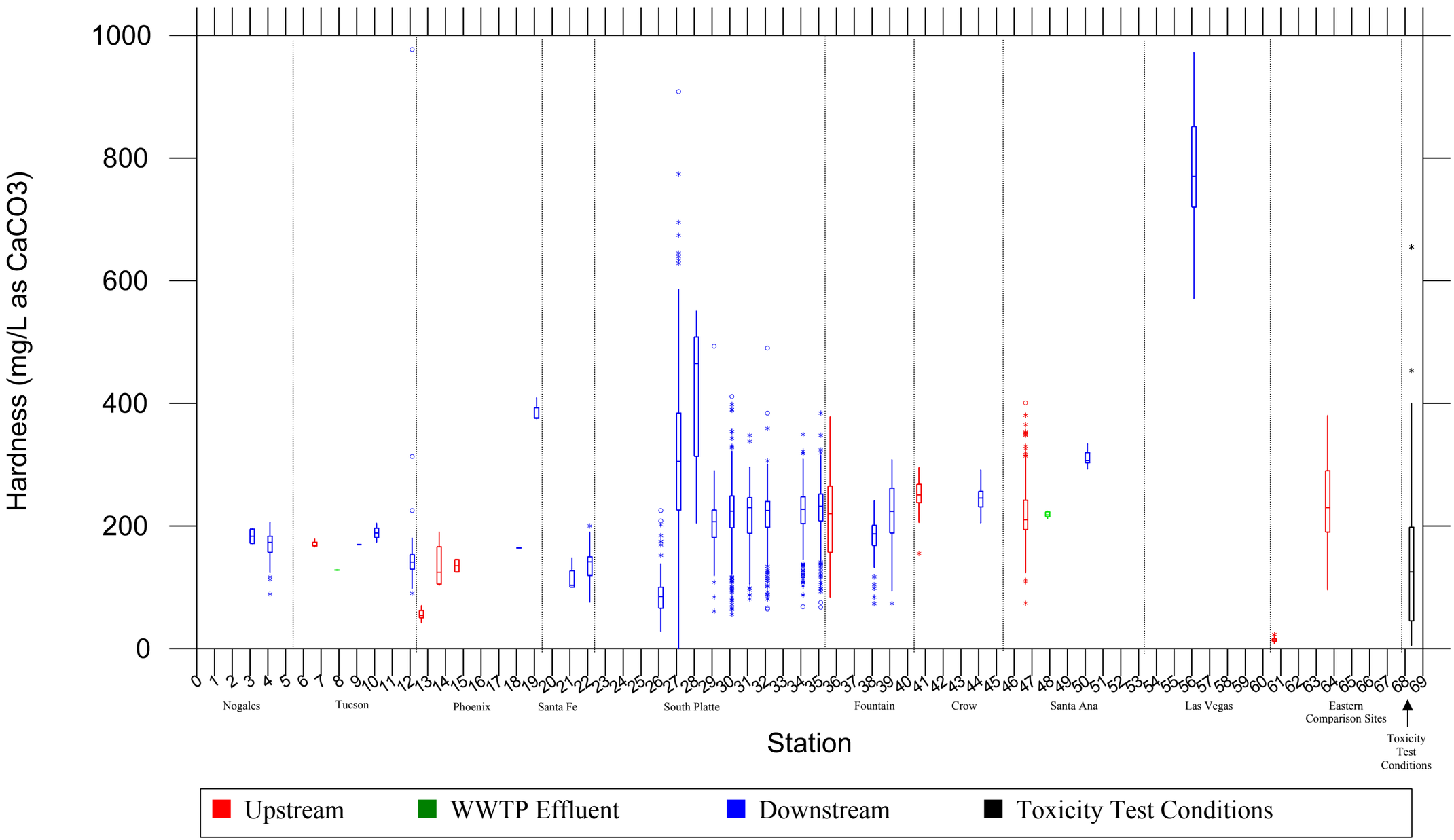
Piper Diagram Showing Average Cation and Anion Concentrations
for 7 of the 10 Study Areas and Eastern Streams
Figure D-10



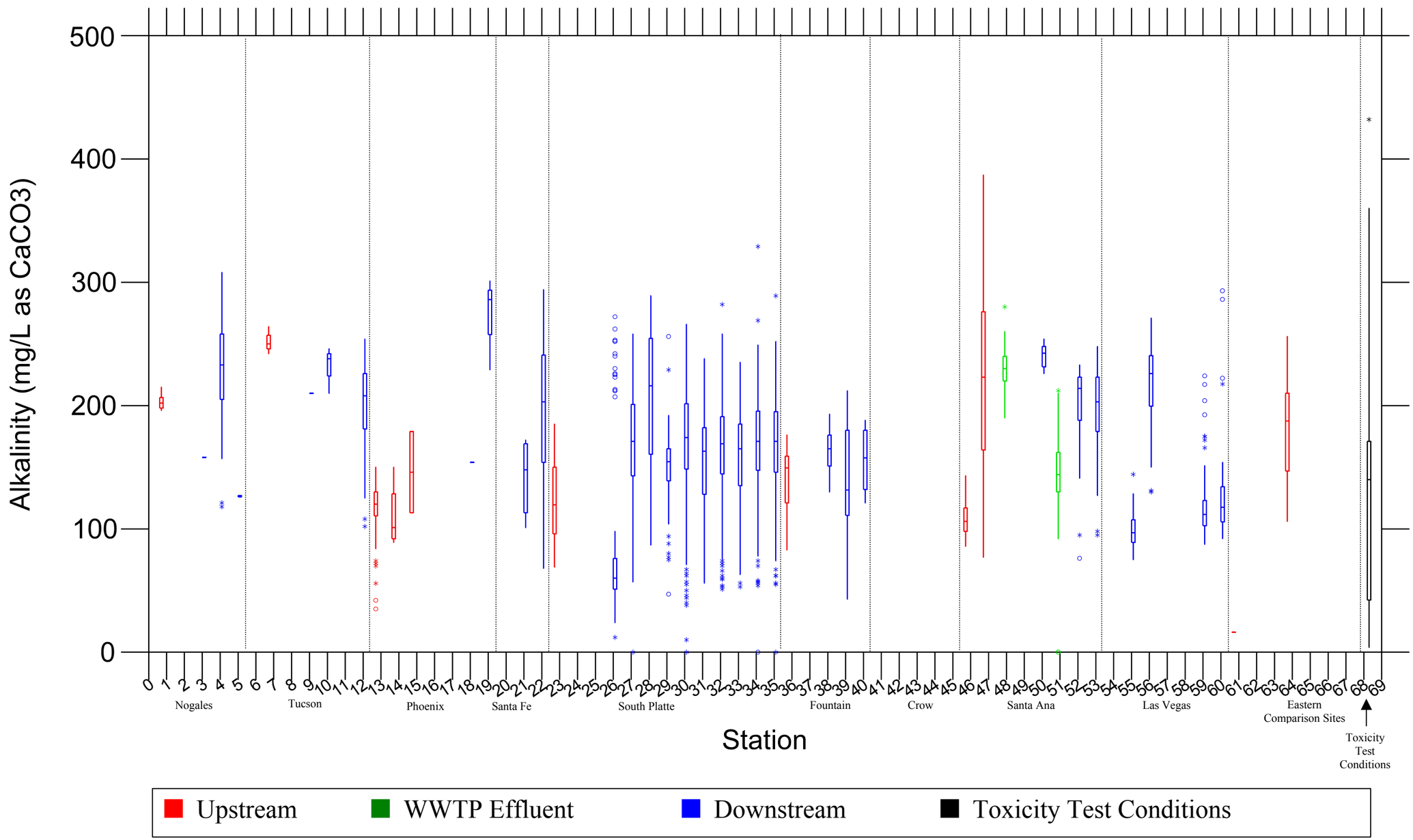
TDS Box Plots
Figure D-11



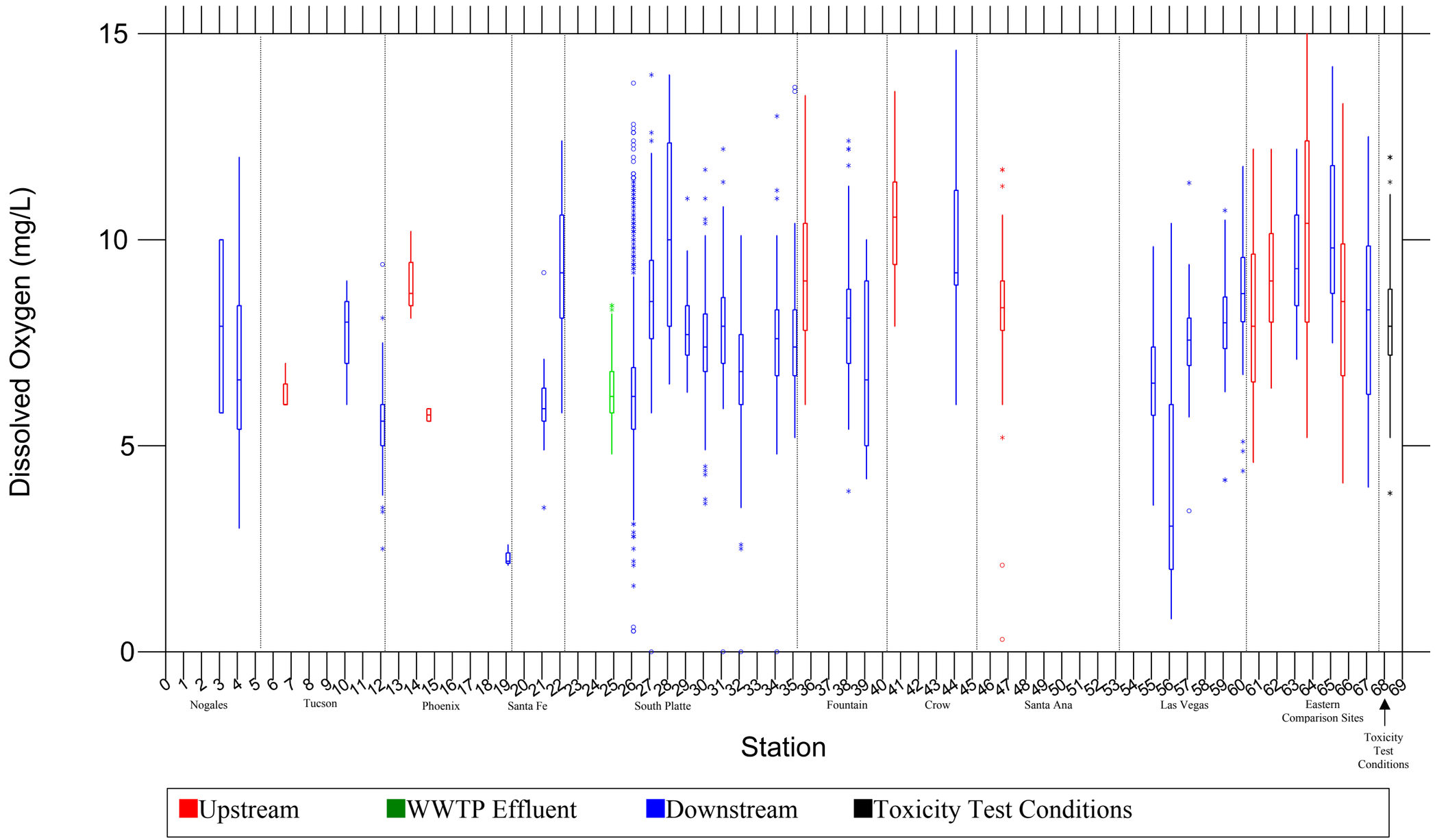
Conductivity Box Plots
Figure D-12



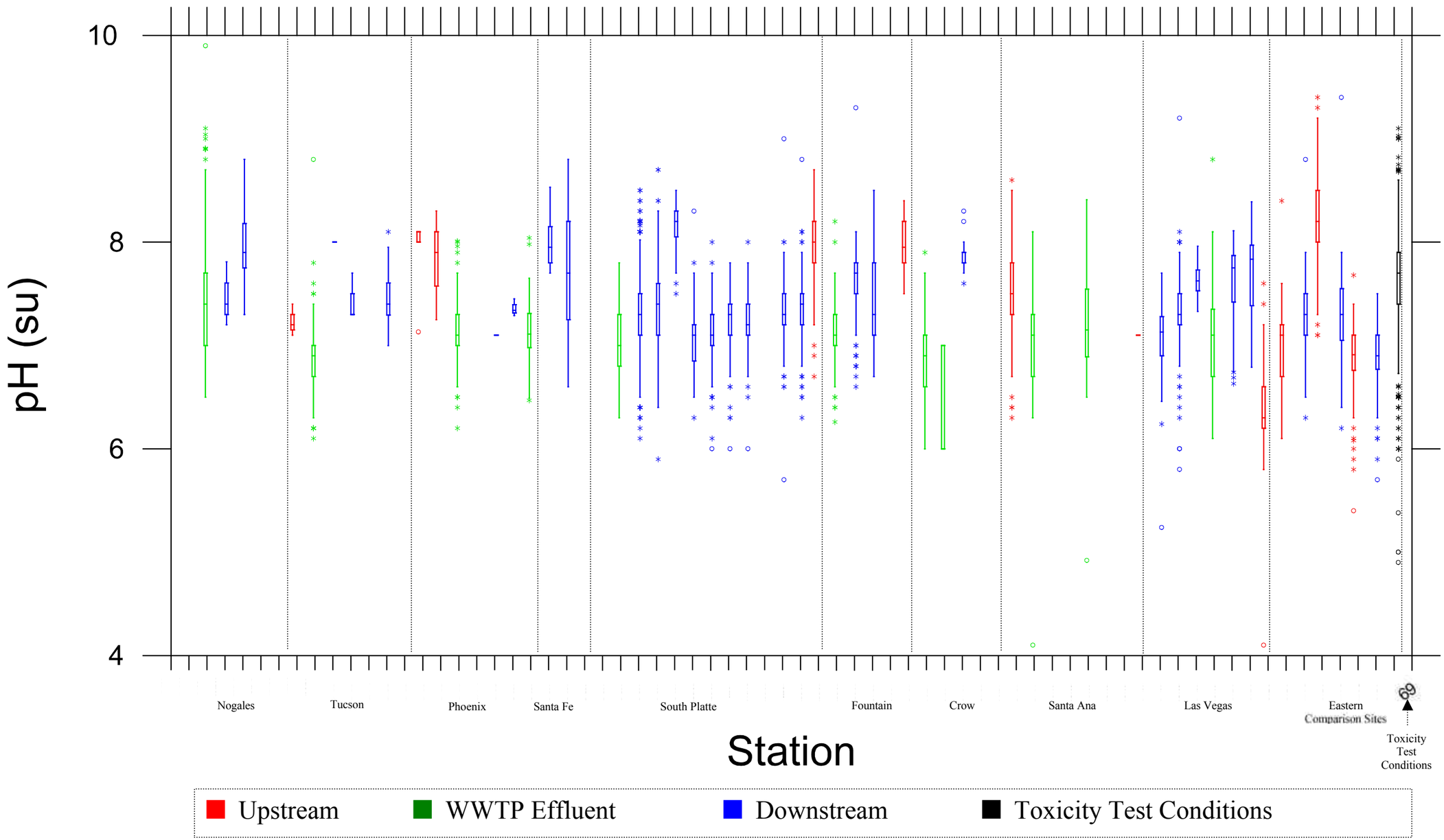
Hardness Box Plots
Figure D-13



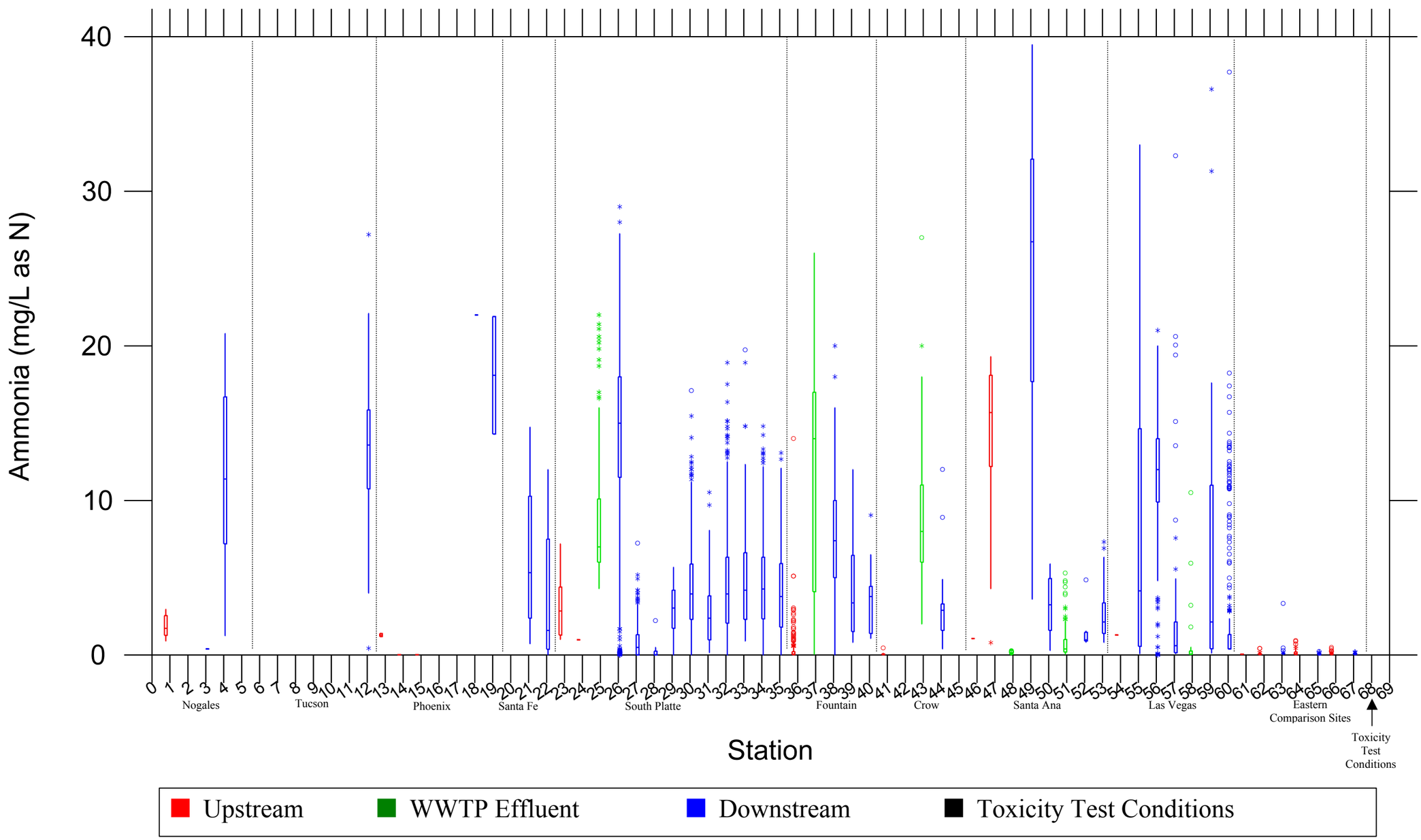
Alkalinity Box Plots
Figure D-14



Dissolved Oxygen Box Plots
Figure D-15



pH Box Plots
Figure D-16



Ammonia Box Plot
Figure D-17

levels for the toxicity tests (5 to 10 mg/L) generally are near the middle of the range measured for the arid streams (0 to 12 mg/L). pH conditions for the toxicity tests and measured pH values for the arid streams are similar, both generally ranging between 6 to 9 Standard Units (S.U.) (**Figure D-16**).

Figures D-11 through D-17 also contain box plots representing data compiled for several non-arid streams. These data are evaluated in **Section 1.2.2.3**.

Table D-2
Station ID Key for Box Plots

Box Plot Number	Station ID	Box Plot Number	Station ID
Santa Cruz River near Nogales, Arizona		**Fountain Creek near Colorado Springs, Colorado**	
1	09480500 (upstream)	36	07105500 (upstream)
2	Nogales WWTP	37	Colorado Springs WWTP
3	800000000028190 (downstream)	38	07105530 (downstream)
4	800000000028170 (downstream)	39	07105800 (downstream)
5	09481740 (downstream)	40	07106000 (downstream)
Santa Cruz River near Tucson, Arizona		**Crow Creek near Cheyenne, Wyoming**	
6	09482500 (upstream)	41	06755950 (upstream)
7	805750000000000 (upstream)	42	Dry Creek WWTP
8	Roger Road WWTP	43	Crow Creek WWTP
9	805770000000005 (downstream)	44	06756060 (downstream)
10	800000000017590 (downstream)	45	06756100 (downstream)
11	Ina Road WWTP	**Santa Ana River near San Bernardino, California**	
12	800000000016960 (downstream)	46	11051500 (upstream)
Salt River near Phoenix, Arizona		47	Y5110000 (upstream)
13	09502000 (upstream)	48	Colton WWTP
14	09512165 (upstream)	49	11059300 (downstream)
15	09512190 (upstream)	50	MWDCROSS (downstream)
16	Phoenix WWTPs	51	Riverside WWTP
17	09512405 (downstream)	52	11066460 (downstream)
18	000203 (downstream)	53	11074000 (downstream)
19	SLR1 (downstream)	**Las Vegas Wash near Las Vegas, Nevada**	
Santa Fe River near Santa Fe, New Mexico		54	LW11.2 (upstream)
20	Santa Fe WWTP	55	LW8.85 (downstream)
21	URG110.002045 (downstream)	56	09419700 (downstream)
22	08317200 (downstream)	57	LW6.05 (downstream)
South Platte River near Denver, Colorado		58	Henderson WWTP
23	06714000 (upstream)	59	LW3.7 (downstream)
24	06714215 (upstream)	60	LW0.55 (downstream)
25	Denver Metro WWTP	**Non-arid Comparison Sites**	
26	000009 (downstream)	61	French Broad River, NC (03451500) (upstream)
27	SP-64 (downstream)	62	French Broad River, Ashville NC (upstream)
28	SC (downstream)	63	French Broad River at SR 1634 (downstream)
29	SPR-CC (downstream)	64	Kansas River, KS (06892350) (upstream)
30	SP-78 (downstream)	65	Ararat River at SR 2019, NC (downstream)
31	SP-88 (downstream)	66	Tar River at HWY 97, NC (upstream)
32	SP-MCKAY (downstream)	67	Tar River at SR 1252, NC (downstream)
33	06720500 (downstream)	**Toxicity Test Dilution Water**	
34	SP-124 (downstream)	68	Toxicity Test Dilution Water
35	SP-160 (downstream)		

1.2.2.3 Principal Components Analysis

Table D-3 shows the results for the principal components analysis (PCA) for the three different data structures. These results indicate that a large percentage of the total variance in the chemical data is explained by only three principal components (PCs), from 70.3 to 94.1 percent depending on the data structure.

Table D-3
PCA Results – Percentage of Accumulated Explained Variance by the Principal Components

Data Structure	Cases [5]	PC1	PC2	PC3
PCA 1 (8 variables) [1]	371	69.1	81.6 (12.5) [4]	94.1 (12.5)
PCA 2 (13 variables) [2]	310	64.0	74.9 (10.9)	84.2 (9.3)
PCA 3 (19 variables) [3]	38	43.0	58.8 (15.8)	70.3 (11.5)

[1] Major cations (Ca, Mg, Na, K) and anions (SO_4, Cl, CO_3, HCO_3) only.
[2] Major cations and anions plus key parameters (conductivity, TDS, hardness, pH, temperature).
[3] Major cations and anions, key parameters, plus WWTP-associated parameters (DO, F, TKN, NH_3+NH_4, NO_2+NO_3, P).
[4] Values in parentheses are the percentages of nonaccumulated explained variance for the particular PC.
[5] Cases represents the number of analyses with complete sets of variables in the associated data structure.

PCA 1 Results – Major Cations and Anions Only

The first data structure (PCA 1) consisted of the major cations and anions only (eight variables). This data structure had 371 cases with complete sets of the eight major cation and anion variables. The number of cases broken down by study area were as follows:

Study Area	Number of Cases in PCA 1
Fountain	43 (7, 36) [1]
Las Vegas	44 (0, 44)
Nogales	29 (0, 29)
Phoenix	8 (4, 4)
Santa Ana	209 (194, 15)
Santa Fe	4 (0, 4)
Tucson	34 (0, 34)

[1] Values in parentheses indicate the number of upstream and downstream cases, respectively.

The breakdown of the number of cases indicates that the PCA 1 results will be most representative of the Santa Ana study area (209 cases) with a bias toward upstream reconnaissance sites (194 cases). The largest percentage of the total variation in the eight variables in this data structure was explained by the first PC (69.1 percent). **Figure D-18** provides the variable loadings for each of the three PCs. The first PC (PC1) had high loadings for six variables (Cl, SO_4, Ca, Mg, Na, and K). The second PC (PC2) had a high loading for one variable (HCO_3) but inversely. The third PC (PC3) had a high loading for one variable (CO_3).

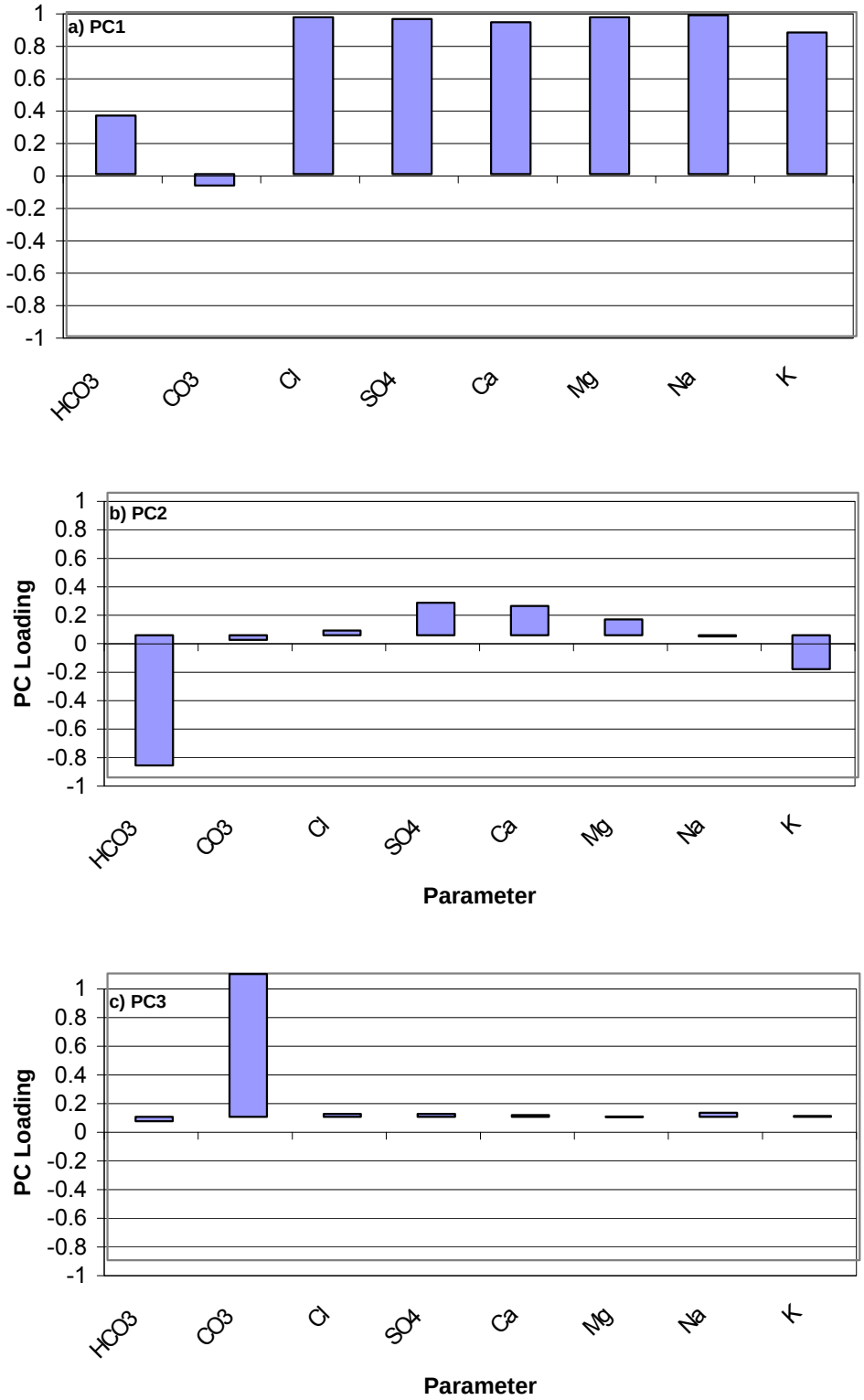


Figure D-18. Loadings for PC1, PC2, and PC3 for Cations and Anions Only Model

Figure D-19 shows the PC scores for the 371 cases in the PCA 1 data structure. The scores are shown as two-dimensional scatter plots of the PCs in which the study areas are represented by different symbols. For PC1 versus PC2 scores, most of the data points cluster together, with relatively low scores for PC1 and variable scores for PC2. An obvious exception is Las Vegas, which has relatively high scores for PC1. Phoenix also tends to have relatively higher PC1 scores. Las Vegas and Phoenix are also differentiated in the PC1 versus PC3 and PC2 versus PC3 plots. In addition, Nogales and Tucson are also differentiated on these plots.

PCA 2 Results – Major Cations and Anions Plus Key Parameters

The second data structure (PCA 2) consisted of the major cations and anions plus key parameters (13 variables). This data structure had 310 cases with complete sets of all 13 variables. The number of cases broken down by study area were as follows:

Study Area	Number of Cases in PCA 2
Fountain	27 (2, 25) [1]
Las Vegas	29 (0, 29)
Nogales	29 (0, 29)
Phoenix	7 (4, 3)
Santa Ana	180 (180, 0)
Santa Fe	4 (0, 4)
Tucson	34 (0, 34)

[1] Values in parentheses indicate the number of upstream and downstream cases, respectively.

The number of cases breakdown indicates that the PCA 2 results also will be most representative of the Santa Ana study area (180 cases) with a bias toward upstream locations (180 cases). The largest percentage of the total variation in the 13 variables in this data structure was explained by the first PC (64.0 percent). **Figure D-20** shows the variable loadings for each of the three PCs. The first PC (PC1) had high loadings for nine variables (Cl, SO_4, Ca, Mg, Na, K, conductivity, TDS, and hardness). The second PC (PC2) had high loadings for two variables (HCO_3 and temperature) but inversely. The third PC (PC3) had high loadings for three variables (HCO_3, CO_3, and pH).

Figure D-21 shows the PC scores for the 310 cases in the PCA 2 data structure. For PC1 versus PC2 scores, most of the data points cluster together, with relatively low scores for PC1 and variable scores for PC2. Similar to PCA 1, Las Vegas and certain Phoenix samples were the exception, having relatively high scores for PC1. Las Vegas and Phoenix (and to a lesser extent Nogales and Tucson) are also differentiated in the PC1, versus PC3 and PC2, versus PC3 plots.

PCA 3 Results – Major Cations and Anions, Key Parameters, and WWTP Parameters

The third data structure (PCA 3) consisted of the major cations and anions, the key parameters, and selected parameters associated with WWTP discharge (19 variables). This data structure had only 38 cases with complete sets of all 19 variables. The number of cases broken down by study area were as follows:

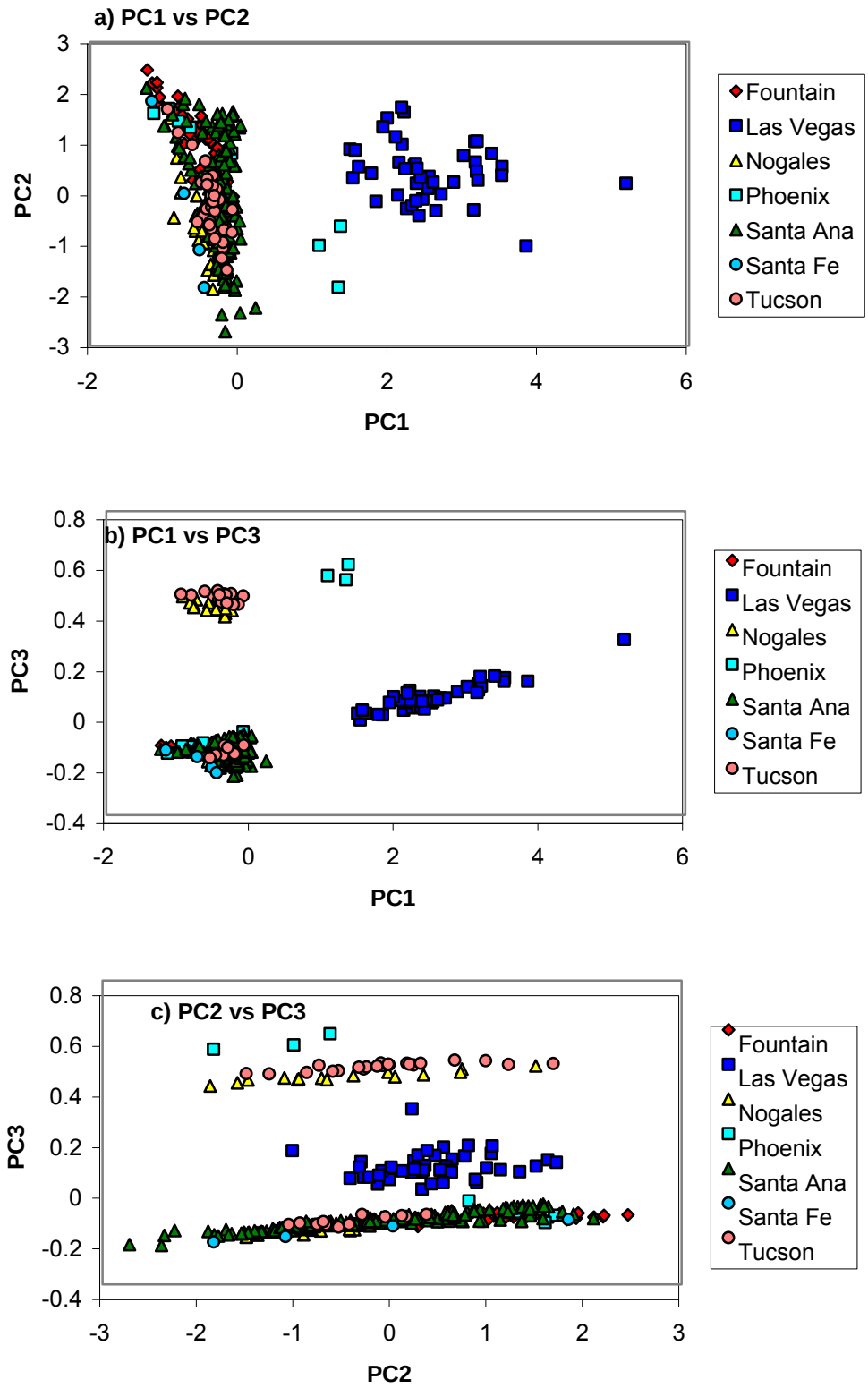


Figure D-19. Scores for PC1 vs. PC2, PC1 vs. PC3, and PC2 vs. PC3 for Cations and Anions Only Model

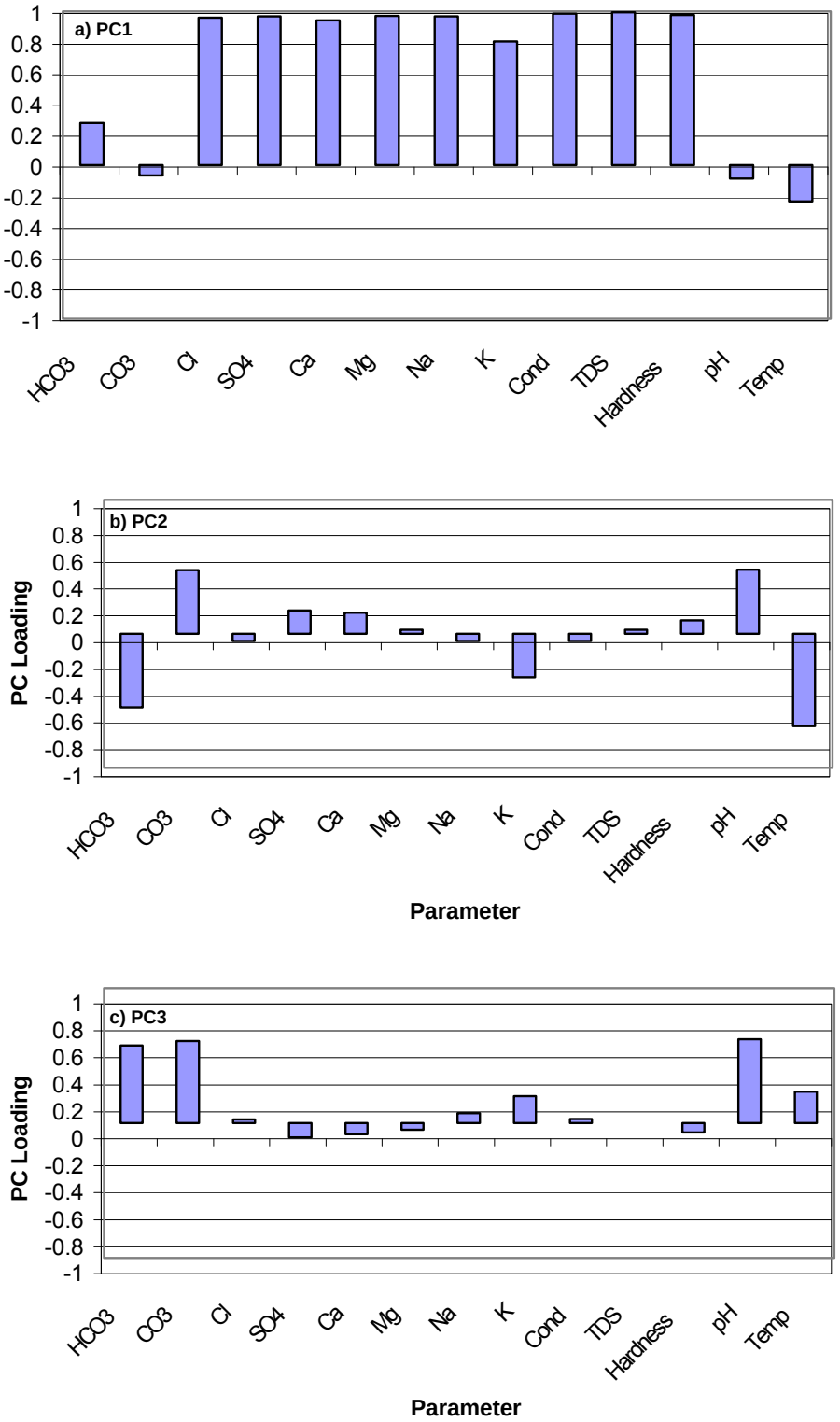


Figure D-20. Loadings for PC1, PC2, and PC3 for Cations/Anions + Key Parameters Model

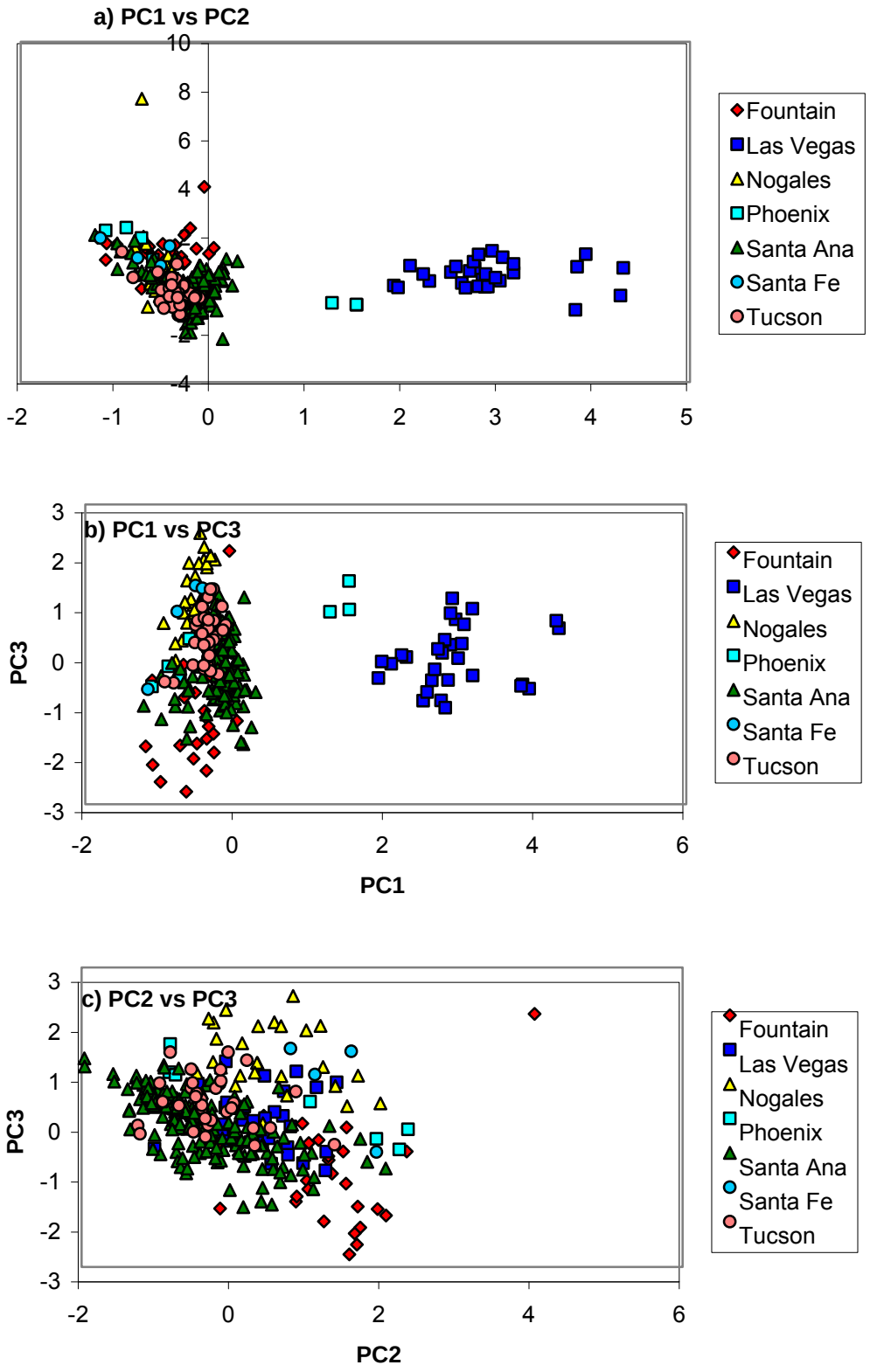


Figure D-21. Scores for PC1 vs PC2, PC1 vs PC3, and PC2 vs PC3 for Cations/Anions + Key Parameters Model

Study Area	Number of Cases in PCA 3
Fountain	0 (0, 0) [1]
Las Vegas	0 (0, 0)
Nogales	15 (0, 15)
Phoenix	1 (0, 1)
Santa Ana	0 (0, 0)
Santa Fe	1 (0, 1)
Tucson	21 (0, 21)

[1] Values in parentheses indicate the number of upstream and downstream cases, respectively.

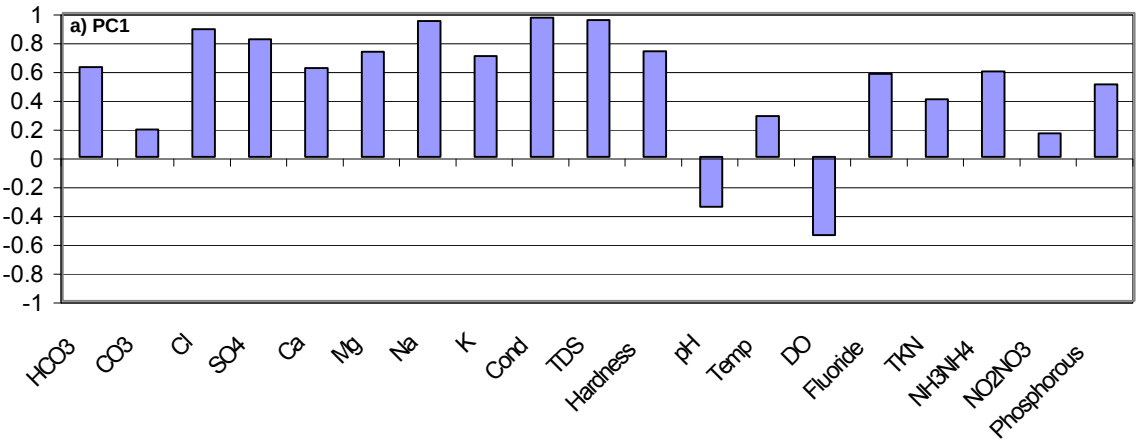
The number of cases breakdown indicates that the PCA 3 results will be most representative of the Nogales and Tucson study areas (15 and 21 cases, respectively). Note that this data structure contains only downstream locations. The largest percentage of the total variation in the 19 variables in this data structure was explained by the first PC (43.0 percent). **Figure D-22** gives the variable loadings for each of the three PCs. The first PC (PC1) had high loadings for 13 variables (HCO_3, Cl, SO_4, Ca, Mg, Na, K, conductivity, TDS, hardness, DO, F, and NH_3+NH_4). Only one of these high loadings (DO) was inverse. The second PC (PC2) had high loadings for seven variables (Ca, Mg, K, hardness, pH, temperature, and NO_2+NO_3), with two of these high loadings (temperature and NO_2+NO_3) inverse. The third PC (PC3) had high loadings for three variables (HCO_3, TKN, and NH_3+NH_4).

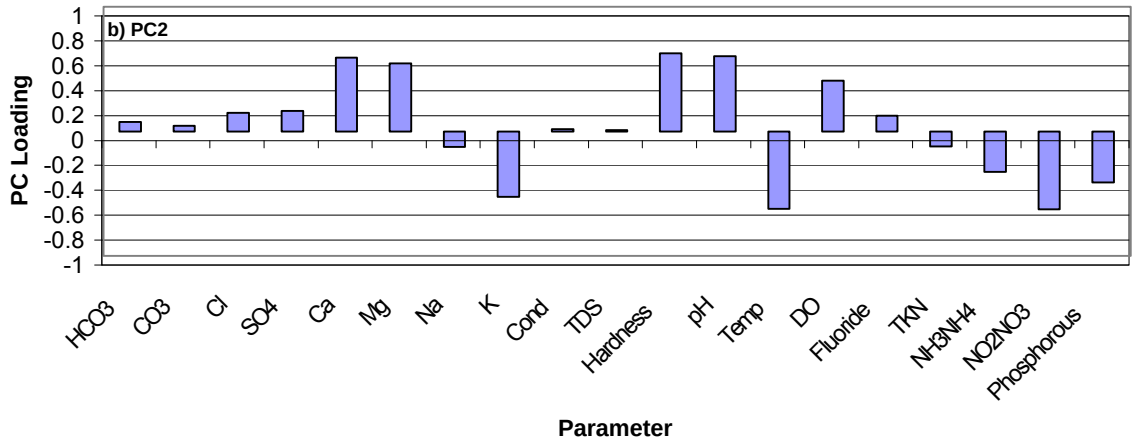
Figure D-23 shows the PC scores for the 38 cases in the PCA 3 data structure. The two locations primarily included in this data structure, Nogales and Tucson, appear to be fairly well differentiated, particularly for PC1 versus PC2 and PC2 versus PC3.

1.2.2.4 Comparisons with Non-Arid Streams

Chemical data from seven non-arid streams were compared with chemical data from the nine arid streams. The seven non-arid sites included three locations on the French Broad River, two locations on the Tar River, and one location on the Ararat River, all in North Carolina, and one location on the Kansas River in Kansas. The non-arid chemical data are provided on the box plots on **Figures D-11 through D-17**, along with the arid chemical data. Box plot comparison figures are provided for TDS, conductivity, hardness, alkalinity, DO, pH, and ammonia. The sets of box plots for each individual non-arid and arid stream location (e.g., South Platte River) are ordered on the figures from upstream to downstream locations on the stream, with upstream and downstream stations color coded. Box plots for WWTP effluent are shown at the location where the effluent is discharged to the stream (i.e., between the immediate upstream and downstream locations) and are also color-coded. An explanation of the components of the box plots for use in interpretation of these figures is provided in **Appendix B**.

Box plots for TDS and conductivity are provided on **Figures D-11 and D-12**, respectively. Note that only limited TDS data were available for the non-arid streams (primarily the Kansas River), whereas a larger amount of conductivity data was available for comparison with the arid streams. Therefore, since conductivity represents an "estimate" of TDS (the portion of TDS due to charged ionic species), **Figure D-12** (conductivity) was used for comparison of non-arid and arid





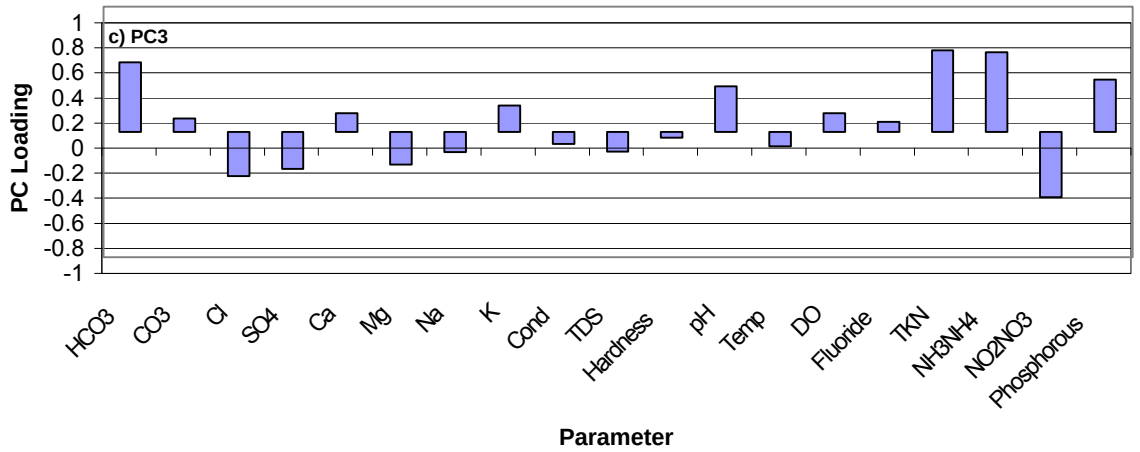


Figure D-22. Loadings for PC1, PC2, and PC3 for Cations/Anions + Key Parameters + WWTP Parameters Model.

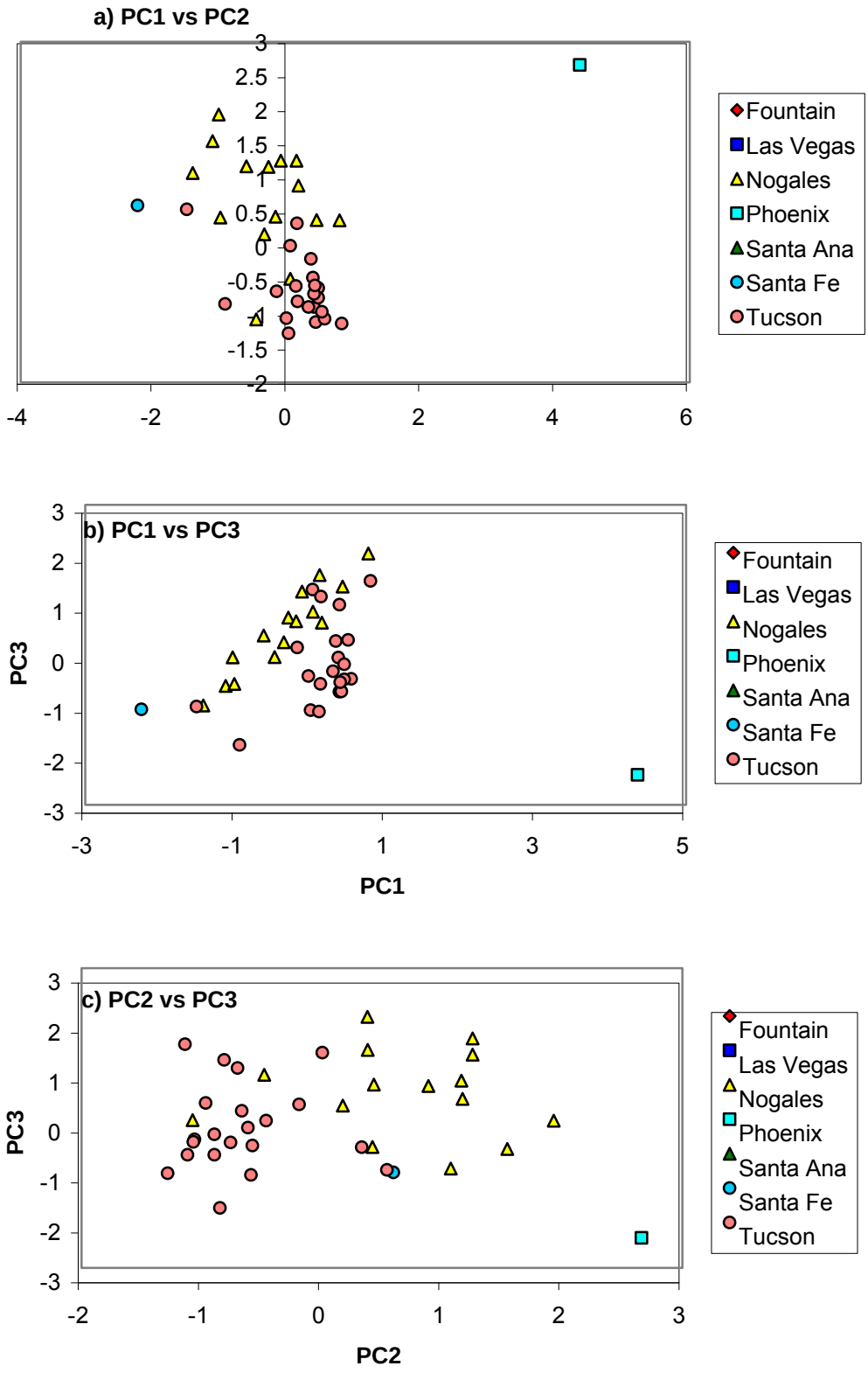


Figure D-23. Scores for PC1 vs PC2, PC1 vs PC3, and PC2 vs PC3 for Cations/Anions + Key Parameters + WWTP Parameters Model.

stream chemistry relative to ionic strength. As shown on **Figure D-12**, the conductivity of the non-arid streams is generally much lower than the conductivity of the arid streams. With the exception of the Kansas River, the conductivity of the non-arid streams generally reaches a maximum of about 400 µmhos/cm, whereas the conductivity of the arid streams, with the exception of Las Vegas Wash, ranges up to about 2,000 µmhos/cm. Most of the arid stream locations had conductivities ranging between about 500 to 1,200 µmhos/cm. The conductivity of the Kansas River is similar to the arid streams, ranging between about 300 to 1,600 µmhos/cm. Note also that some arid stream locations have very high conductivities. For example, Las Vegas Wash locations have conductivities ranging between about 1,500 to 3,500 µmhos/cm, which are higher than typical arid streams and considerably higher than the non-arid streams.

Only limited hardness and alkalinity data were available for the non-arid streams. As shown on **Figure D-13**, hardness for one non-arid stream (French Broad River, North Carolina, upstream) was generally less than 25 mg/L as $CaCO_3$, compared with the arid sites, which generally ranged between about 50 to 600 mg/L as $CaCO_3$. This indicates that Ca and/or Mg concentrations are probably much higher in the arid streams. The hardness of the Kansas River, which ranged between about 100 to 400 mg/L as $CaCO_3$, was similar to the hardness of the arid streams. Similarly, alkalinity (**Figure D-14**) for the non-arid stream (French Broad River, North Carolina, upstream) was less than 25 mg/L as $CaCO_3$, compared with the arid streams, which generally ranged between about 50 to 300 mg/L as $CaCO_3$. Again, the alkalinity of the Kansas River, which ranged between about 100 to 250 mg/L as $CaCO_3$, was similar to the arid streams.

DO concentrations in the non-arid streams (5-15) ranged between 4 to 15 mg/L. Most of the DO concentrations in the arid streams also fall within this same range, although at several locations, values below 4 mg/L occur downstream of WWTP discharges, indicating the possible impact of biochemical oxygen demand (BOD) related to the wastewater discharges.

pH levels in the non-arid and arid streams (**Figure D-16**) were similar, generally ranging between about 6 to 9 S.U.

Ammonia concentrations in the non-arid streams were much lower than in the arid streams (**Figure D-17**). Non-arid stream ammonia concentrations were generally lower than 1 mg/L as N, compared with the arid streams, which reached levels as high as 40 mg/L as N. In general, the highest ammonia levels for the arid streams occurred downgradient of WWTP discharges. At some locations (e.g., South Platte, Fountain) the impact of the WWTP discharge clearly results in an immediate increase in ammonia levels at the nearest downstream location, followed by decreased levels further downstream. This trend indicates the influence of relatively high ammonia levels in the WWTP discharge, which then decrease as the ammonia is oxidized by nitrifying bacteria further downstream. This trend, however, is not apparent in the non-arid sites, probably due to the more significant influence of dilution at these locations.

Comparison of the non-arid stream chemistries to the toxicity test composition indicated that conductivity (**Figure D-12**), DO (**Figure D-15**), and pH (**Figure D-16**) were similar. Insufficient data were available to make a comparison for TDS, hardness, alkalinity, and ammonia. In

general, however, the non-arid stream concentrations appear to more closely resemble the toxicity test concentrations than the arid stream concentrations.

Two of the non-arid streams (French Broad River, North Carolina and Kansas River, Kansas) had major cation and anion data, and were therefore plotted on the Piper diagram on **Figure D-10**. The Kansas River had a similar average major ion composition as the arid streams shown on the Piper diagram. The French Broad River, however, had a higher Na+K composition (70% meq/L) and lower Ca composition (20% meq/L) than the arid streams (43 to 63 and 25 to 42% meq/L, respectively). In addition, the TDS of the French Broad River appears to be lower than for the arid streams, as indicated by the relative sizes of the calculated TDS circles on the Piper diagram.

Compared with the arid streams, the French Broad River anion composition more closely resembles the WET test composition. However, the French Broad River cation composition is different than both the WET composition and the arid streams, especially for Na+K, which is relatively higher in the French Broad River.

1.2.2.5 Conclusions Regarding Differences in Chemical Composition

Based on the chemical data for arid streams, non-arid streams, toxicity test composition, and WET test composition discussed in **Sections 1.2.2.1 to 1.2.2.3**, the following preliminary conclusions are made:

- There are differences between the arid streams' chemical compositions and the WET chemistry.
- The arid streams contain higher ionic strength, as measured by TDS, conductivity, hardness, and alkalinity than the non-arid streams used in this study.
- Additional chemical data for non-arid streams are needed for a more definitive evaluation of differences in chemical composition between non-arid and arid streams, especially for hardness, alkalinity, and major cations and anions.
- Inclusion of a larger set of geographically diverse non-arid streams is required to verify the chemical differences between non-arid and arid streams identified in this report.

1.2.3 Taxonomic Richness

Evaluating the role of habitat in limiting taxonomic richness is especially important when conducting bioassessments to determine if an aquatic life use is impaired. For example, assumptions that effluent-dependent waters should have similar aquatic species richness as naturally flowing streams (e.g., naturally warmwater or coldwater streams) may be inappropriate if habitat limitations preclude attainment of higher taxonomic richness.

A discussion of taxonomic richness in arid West streams is contained in **Appendix G**. A summary of this analysis stated that aquatic communities in "harsh" environments are expected to be trophically simple, have relatively low species richness, and be stable and persistent in the

face of disturbance, due to the dominance of flood and/or drought resistant taxa (Poff and Ward 1989; Reice et al. 1990). Communities in more "benign" environments are expected to be trophically complex, have intermediate or high species richness, and experience decreased persistence and stability in the face of unpredictable disturbances (Peckarsky 1983; Poff and Ward 1989).

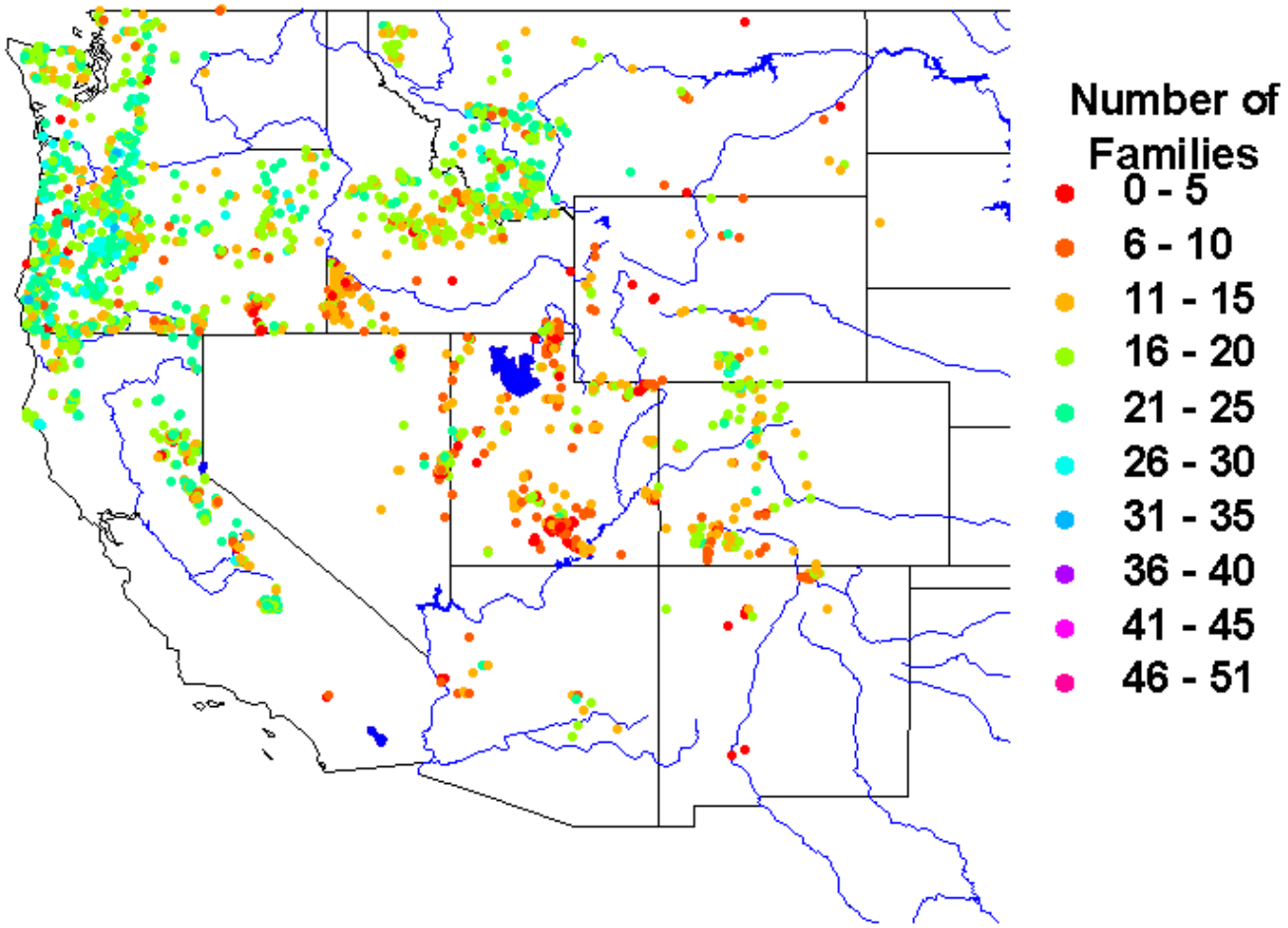
An evaluation of the number of species typically found in arid West streams requires data from a variety of stream types in both arid and non-arid environments. In addition, the level of comparison must be equal (e.g., consistent level of taxonomic identification and consistent level of effort and frequency of sampling). A comprehensive database containing such consistency is unavailable for this evaluation. However, for invertebrate species we can summarize macroinvertebrate data from the historic data compilation and site reconnaissance effort to characterize "typical" taxonomic richness at a consistent taxonomic level of analysis. The results for invertebrates can be compared to what is known regarding "typical" richness in specific areas. This review for aquatic invertebrates is far from exhaustive, but does provide an indication of the degrees of difference of species richness, if any, between arid West and non-arid West streams.

The National Aquatic Monitoring Center (NAMC) recently summarized the results of more than 11,000 benthic invertebrate samples collected across the western United States (NAMC 2000). Although sampling methods could not be controlled, the following parameters apply to the majority of samples: samples were collected from stream riffle habitats with fixed area benthic samplers (kicknets, Surber samplers, Hess Nets) and the total sampling area for most of the estimates was approximately 7.5 square feet.

The compilation of samples shows that in the arid West, especially in lowland areas, the number of families was most commonly found to be between 6 and 15 (**Figure D-24**). Sites with five or fewer families were rare and sites with samples containing more than 20 families were increasingly uncommon, except possibly at higher elevations. Less arid areas (e.g., west of the Cascade Mountains, northern Idaho, western Montana, and northern California) most commonly had samples with 16 to 25 families. These results were matched by the results obtained for the typical number of genera. The NAMC results do not distinguish between impacted and non-impacted sites.

Arizona recently summarized the results of aquatic invertebrate samples collected from warmwater sites (<5,000 feet elevation) from 1992 to 1997. Results are based on riffle samples collected with a 500-micron mesh kicknet from an area of approximately 10.75 square feet during the spring season. At the "family" level of identification (insects to family; non-insects to family, order, or class) the median number of taxa was approximately 16 (ADEQ 1998). This median number of taxa represents only reference sites (i.e., sites that are minimally impacted).

Table D-4 summarizes the number of taxa (family/order/class) collected at each reconnaissance site, and **Table D-5** summarizes the number of taxa (family/order/ class) from sites with the best historical data. The median number of taxa per site for 2,000 reconnaissance sites where there was water was at the lower end of the NAMC range at 7 taxa per site; the number of taxa ranged from 2 to 17. The median number for historical data at selected sites was a little higher at 9 taxa



Number of Families from Macroinvertebrate Studies
Conducted across Western United States
Figure D-24

per site with a range of 2 to 20 taxa. The slightly higher median value for the historical data set likely represents improved data quality (i.e., the site reconnaissance data was based on field identification only, while the historical data was based on laboratory analysis of field samples).

Table D-4
Number of Taxa (Family/Order/Class) Collected from Each Reconnaissance Site in Spring 2000

Water Body	Reconnaissance Site				
	1	2	3	4	5
Santa Fe River	0	10	11	14	17
Santa Cruz River, Tucson	0	2	4	3	7
Santa Cruz River, Nogales	0	9	6	15	11
Salt River	0	5	7	11	7
Las Vegas Wash	11	5	6	6	5
Santa Ana River	9	3	10		
Carrizo Creek	15	5	11	7	
South Platte River	9	6	5	8	7
Fountain Creek	5	11	7	5	6
Crow Creek	16	2	6	13	6

Table D-5
Number of Taxa (Family/Order/Class) from Selected Historical Records Gathered from Habitat Characterization Study Sites

Water Body	Upstream ----------> Downstream						
Santa Ana River 1991 (mean of quarterly samples)	9	8	11	5	11	8	12
Santa Ana River August 1998	13	10	6	20	10	6	11
Crow Creek 1993	17	4	4	8			
South Platte River 1988	8	2	7	5	7		
South Platte River 1989	14	4	5	11	7		
South Platte River 1990	8	11	7	8	8		
South Platte River 1991	10	9	9	8	6		
South Platte River 1993	9	9	9	6	7		
Santa Fe River 1994	9	11	11	9	10	13	
Santa Cruz River, Nogales 1993 (mean of monthly samples)	17	9	14	13			
Fountain Creek 1980	17	18	9	12	10	11	12
Fountain Creek Fall 1999	13	15	12	13	14	17	11

With few exceptions the number of taxa falls within the range of 6 to 15 families found by NAMC. This result was consistent for both historical and reconnaissance data. The 7 or 9 taxa per site values are considerably lower than the 16 taxa found as the median number of families/orders/classes in warmwater streams in Arizona. However, as indicated above, the Arizona results are based on so-called reference sites representing expectations for sites minimally impacted from anthropogenic activities.

Several of the reconnaissance sites had flow upstream of the WWTP discharge. These sites, if minimally impaired, could provide an indication of typical taxonomic richness in the arid West.

However, in all cases stressors unassociated with wastewater effluent impacted the upstream site. As a consequence, while richness was greater upstream of the WWTP discharge in five of six cases, it is unknown if the upstream richness was indicative of typical expectations for arid West streams as a whole.

Data from the limited review of non-arid streams found the North Carolina streams to have greater taxanomic richness than the arid streams: 20 to 25 taxa at sites downstream of WWTP discharges (family/orders/class level) and about 30 taxa upstream of the effluent discharge. Kansas sites had similar or lower richness than this study's arid areas; however, the use of artificial substrates as the sampling method, while appropriate for that specific study, likely does not provide an accurate representation of overall site species richness. Clearly, the North Carolina streams provide only a small representation of non-arid streams, but it was interesting that richness was so much greater in these areas than the sites reviewed for this study.

The results from the review of historical and reconnaissance data strongly suggest that for effluent-dependent waters in the arid West the number of taxa (families/ orders/classes) is lower than would be expected for non-effluent-dependent waters. However, insufficient data are available to evaluate whether taxonomic richness is lower in all arid West streams, including non-effluent-dependent waters, when compared to non-arid West streams. NAMC data suggest that arid West streams as a whole do have lower richness, but without gathering historical data on a wider range of sites this suggestion cannot be confirmed.

Although we cannot state unequivocally that species richness is lower in arid West streams as a whole, we can state with some confidence that effluent-dependent waters have lower species richness than non-effluent-dependent waters. The second part of the stated supposition is that lower species richness in arid West streams is due to habitat limitations. This statement presumes that water quality is not the limiting factor.

Determining what influences species richness in stream ecosystems has been a subject of considerable interest to stream ecologists for decades. Vinson and Hawkins (1998) recently evaluated more than 30 years of stream studies for the purpose of evaluating what local, basin, and regional factors influence biodiversity of stream insect communities. They concluded that richness was most consistently influenced by the following factors:

- Local Scale: (1) substrate size - positive relationship exists between median particle size and richness; (2) disturbance - negative relationship between disturbance intensity/frequency and richness.

  Substrate size is a highly localized factor that influences aquatic species richness regardless of the location (i.e., whether or not the stream is in the arid West). Locales dominated by small particle sizes tend to have lower species richness, but this tendency can be mitigated by the presence of other substrates such as macrophytes and algae. The conclusion by Vinson and Hawkins (1998) that flood intensity and frequency negatively influence species richness in general supports the findings present in **Appendix G**.

- Basin Scale: (1) annual temperature range - positive relationship between annual temperature range and richness; (2) flow intermittency - increasing richness with increasing flow duration.

  It is not possible to evaluate the role of annual temperature range for all arid West streams; however, it is important to note the potential importance of this factor in effluent dependent waters. Since these waters consist primarily of a relatively constant rate and source of discharge, the annual temperature range of the effluent at the point of discharge and for some distance downstream will be small. This narrow temperature range could negatively influence species richness below the discharge. However, the extent of the influence depends on other factors such as other sources of surface water, degree of groundwater influence, volume of discharge, and discharge location (cold versus warm climate).

  Species richness in arid streams as a whole will be negatively influenced by flow intermittency, a common occurrence in the arid West, especially seasonally. Effluent-dependent waters subject to variability in discharge volume and water withdrawals downstream of the discharge, especially seasonally, also may be impacted by flow intermittency.

- Regional Scale: biome type - (1) typically lower species richness occurs in plains versus subalpine and montane streams; (2) tundra/alpine streams have lower richness than forested streams; and (3) richness is highest in transition zones between montane and valley sites.

  The degree of impact of biome type across all arid West streams is unknown. However, we note that the types of streams documented by Vinson and Hawkins (1998) as the types with the lowest richness are similar in nature to many lowland arid West streams (i.e., lack of forest), plains (i.e., low gradient) type waters. It is also notable that Arizona has found that the median number of families/orders/classes is typically slightly lower in warmwater streams (lowland) than in coldwater streams (upland).

Several of the above factors identified by Vinson and Hawkins (1998) can be associated with characteristics of arid West stream ecosystems and one could easily conclude that their combined influence on species richness in the arid West might be significant (i.e., their combined influence leads to lower species richness). However, while the potential link is interesting, the lack of data directly linking lower richness to habitat limitations precludes making any firm conclusions at this time. Furthermore, to fully evaluate whether lower species richness results from habitat limitations, it is necessary to eliminate water quality as a factor.

1.2.4 Toxicity Database Species

EPA methodology for establishing aquatic life criteria was formally published in 1985 (EPA 1985). Since that time the method has served as the basis by which EPA generates national water quality criteria documents. The method established that derivation of national freshwater criteria

for the protection of aquatic organisms should include acute toxicity test data from at least eight different families from the following groups:

- Family Salmonidae in Class Osteichthyes
- Second family in Osteichthyes, preferably a commercially or recreationally important warmwater species such as bluegill or channel catfish
- Third family in phylum Chordata (may be in the class Osteichthyes or other group)
- Planktonic crustacean (e.g., Cladocera or Copepoda)
- Benthic crustacean (e.g., Ostracoda, Isopoda, Amphipoda, Decapoda)
- Aquatic insect family (e.g., Ephermoptera, Plecoptera, Trichoptera, Diptera)
- Family in phylum other than Chordata or Arthropoda (e.g., Rotifera, Annelida or Mollusca)
- Family in any order of insect or any phylum not already represented

In addition, acute to chronic ratios should be available from at least three different families from the following:

- At least one family of fish
- At least one family of invertebrate
- At least one acutely sensitive freshwater species

Of interest to the arid west is the degree to which these requirements can be fulfilled in arid West streams, which may have a limited aquatic fauna.

To evaluate this issue, historical and reconnaissance site taxonomic data were reviewed to determine how often these minimum requirements are met at the study areas. This review was generally limited to the effect species limitations may have on use of these guidelines. A secondary problem associated with limited taxa may occur when pollutant-specific national water quality criteria documents are used to establish water quality criteria. These documents contain species-specific toxicity data for use in establishing criteria. It is possible that even though we find that a site may have the requisite number of families to generate, a criterion the criteria document used to generate the criteria will not contain toxicity data relevant to the families that are present at a given site. This kind of evaluation is pollutant/document specific and will be fully carried out for specific pollutants by subsequent research supported by the Arid West Water Quality Research Project.

Table D-6 provides a comparison of the above listed categories with what has been recorded from each of the study areas. With the exception of the first category, family Salmonidae, most

areas appear to have the other requisite fauna necessary to meet the method's eight family minimum. Salmonidae have been recorded only from Fountain Creek, Colorado. Crow Creek does have a record of the salmonid brown trout, but these trout were stocked to determine the viability of a coldwater fishery in this stream and it appears that this trout population is not self-propagating. These results for Salmonidae are not surprising given that most arid study areas are located in biomes that typically have warmwater type streams.

Cladocera have been recorded at most of the study areas, the exceptions being Fountain Creek, Santa Fe River, and Carrizo Creek. We cannot assume that the lack of a record for Cladocera in these areas precludes the possibility that they are actually present, since failure to collect Cladocera can be related to sampling methods.

Of greater interest is the presence/absence of the cladoceran family Daphnidae. This is of interest because the majority of acute and chronic toxicity tests reported for Cladocera species are from this family (for example, see EPA 1996b). To evaluate this issue, we can rely only on historical data since the site reconnaissance effort was limited to field identifications that do not allow for identification of Cladocera at the family or genus level. The historical data show that only 2 of the 10 study areas have documented the presence of Daphnidae (Salt/Gila River - *Ceriodaphnia*; Santa Cruz River, Tucson - *Daphnia*). Again, failure to document these taxa from the other eight study areas does not preclude the possibility of their presence since zooplankton sampling has not been routinely carried out at most study areas.

1.2.5 Flow-Augmentation Effects

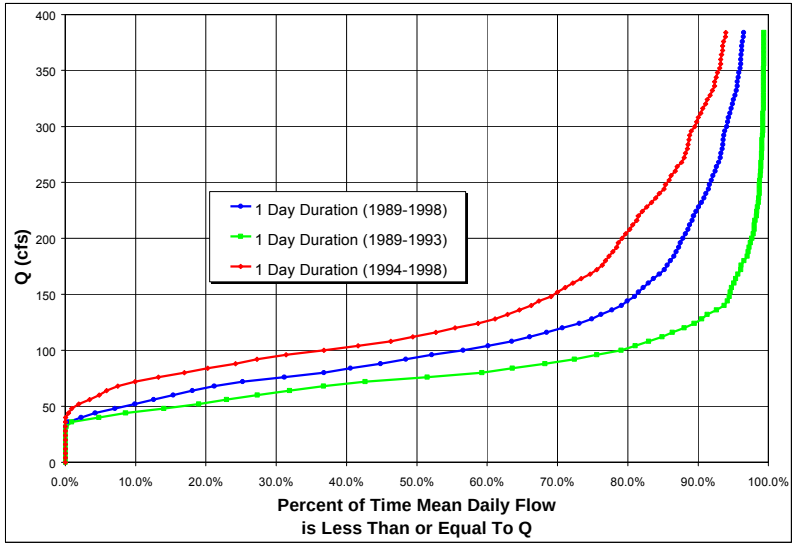
Study areas where flows occur consistently upstream of the WWTP outfall (i.e., effluent-dominated systems) include the Santa Ana River, Fountain Creek, South Platte River, Las Vegas Wash, and Crow Creek. Site where flows upstream of the WWTP outfall are typically zero (i.e., effluent-dependent streams) include the Santa Cruz River (Nogales), Santa Cruz River (Tucson), Salt River, and Santa Fe River.

Figure D-25 presents cumulative frequency distributions through time for four sites that have temporal aquatic biological data. For these four sites, **Figure D-25** illustrates that flows have increased throughout time. As an example, a flow of 50 cubic feet per second (cfs) in Fountain Creek was equaled or exceeded 80 percent of time during the years of 1989 to 1993. During the years of 1993 to 1999, a flow of 80 cfs in Fountain Creek was equaled or exceeded 80 percent of time.

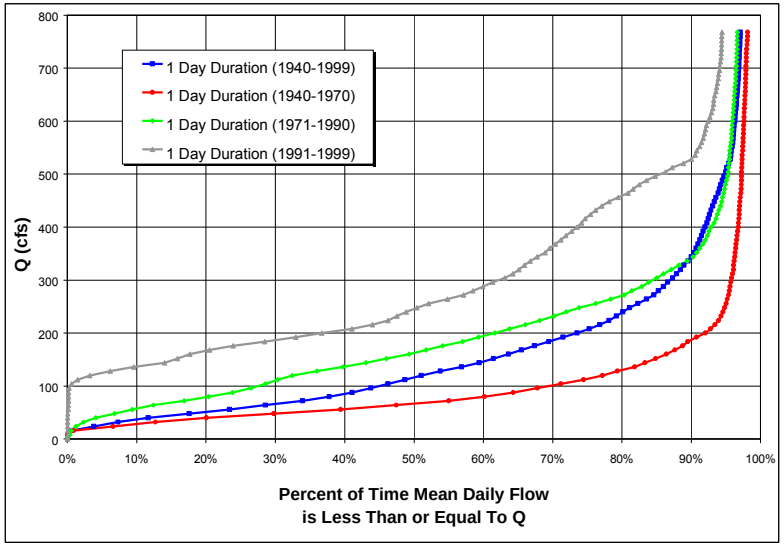
The creation of perennial stream ecosystems by the discharge of treated wastewater has the potential to create a net ecological benefit in the arid West where demands on water resources are resulting in less water in streambeds and the loss of riparian habitat. Benefits include support of aquatic organisms that provide food for higher trophic levels (e.g., piscivorous birds and mammals) and riparian vegetation that provides food, cover, and nesting opportunities for terrestrial fauna. In some parts of the arid West, lotic aquatic-habitat is limited and the habitat created by the discharge of treated wastewater can be an important resource.

Table D-6
Comparison of Taxonomic Groups Known to be Present at Habitat Characterization Study Sites and Taxonomic Categories Required for Calculation of Acute Toxicity Criteria

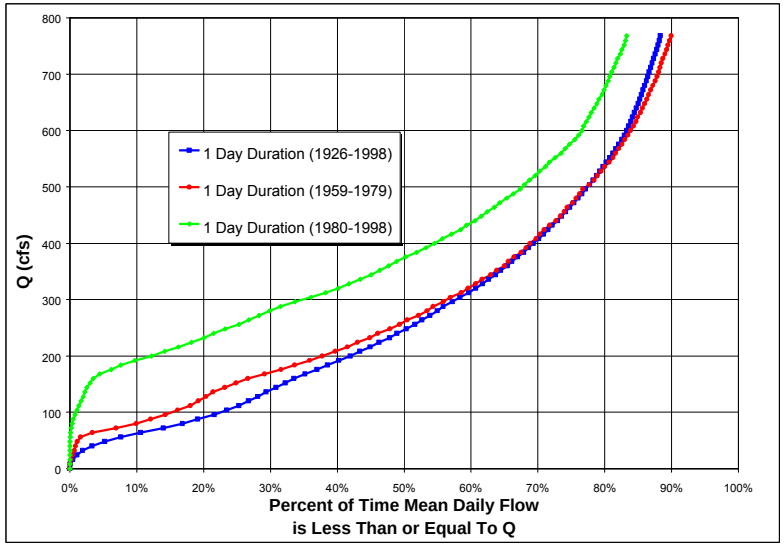
Categories	Salt/Gila River	Santa Cruz River, Nogales	Santa Cruz River, Tucson	Santa Ana River	Fountain Creek	South Platte River	Santa Fe River	Crow Creek	Las Vegas Wash	Carrizo Creek
Osteichthyes (Salmonidae)	No	No	No	No	Yes	No	No	Stocked	None	None
Osteichthyes (2nd family) Chordata (family in Osteichthyes or other group)	7 other fish families present; no data on other Chordata	4 other fish families present; no data on other Chordata	3 other fish families possibly present; no data on other Chordata	6 other fish families present; no data on other Chordata	6 other fish families present; no data on other Chordata	8 other fish families present; no data on other Chordata	4 other fish families present (only 2 families within 5 - 7 miles of discharge; no data on other Chordata	3 other fish families present; no data on other Chordata	No records	No records
Planktonic crustacean (Cladocera or Copepoda)	Copepoda and Cladocera, including *Ceriodaphnia*	Copepoda, Cladocera	Copepoda and Cladocera including *Daphnia*	Cladocera	No record	Cladocera	No record	Copepoda	Cladocera	No record
Benthic crustacean (Amphipoda, Ostracoda, Isopoda, Decapoda)	Ostracoda & Amphipoda	Ostracoda	Ostracoda	Ostracoda, Amphipoda, Isopoda, Decapoda	Ostracoda, Amphipoda, Isopoda	Ostracoda, Amphipoda, Isopoda	Amphipoda	Isopoda, Amphipoda, Decapoda	Ostracoda	Amphipoda, Decapoda
Insect	At least 15 families at each site; some sites with more than 30								7 families	11 families
Phylum (not Arthropoda or Chordata)	Platyhel-minthes, Nematoda, Rotifera, Annelida, Mollusca	Nematoda, Annelida, Mollusca	Platyhel-minthes, Annelida	Platyhel-minthes, Annelida, Mollusca	Cnidaria, Platyhel-minthes, Nematoda, Annelida, Mollusca	Platyhel-minthes, Nematoda, Annelida, Mollusca	Platyhel-minthes, Annelida, Mollusca	Annelida, Mollusca	Platyhel-minthes, Annelida, Mollusca	Mollusca
Family (any order of insect or any phylum not already represented)	See Above									



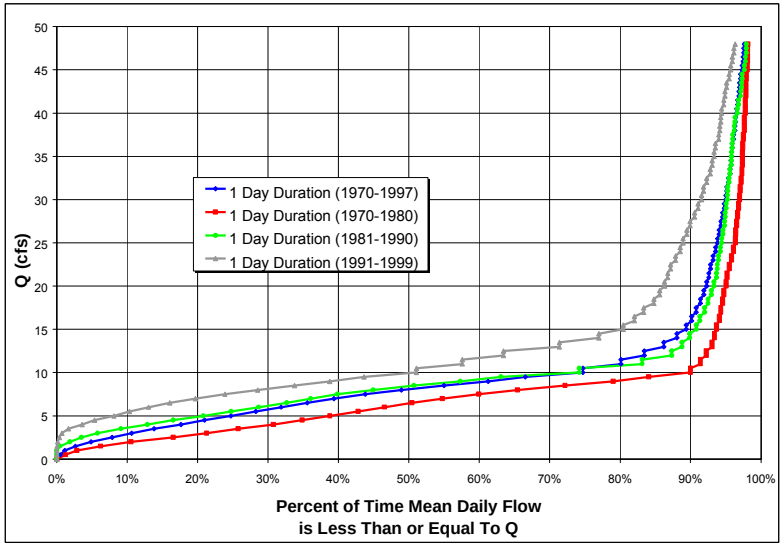
Fountain Creek Downstream of WWTP near Colorado Springs, Colorado



Santa Ana River Downstream of WWTPs near San Bernardino, California



South Platte River Downstream of WWTP near Denver, Colorado



Santa Fe River Downstream of WWTP near Santa Fe, New Mexico

Cumulative Frequency Distributions through Time
Figure D-25

1.2.5.1 Aquatic Species

To evaluate this issue for aquatic species, we primarily reviewed data collected during the site reconnaissance effort. We also evaluated historical data if data were available from above and below the discharge of treated wastewater. Number of taxa served as the point of comparison in

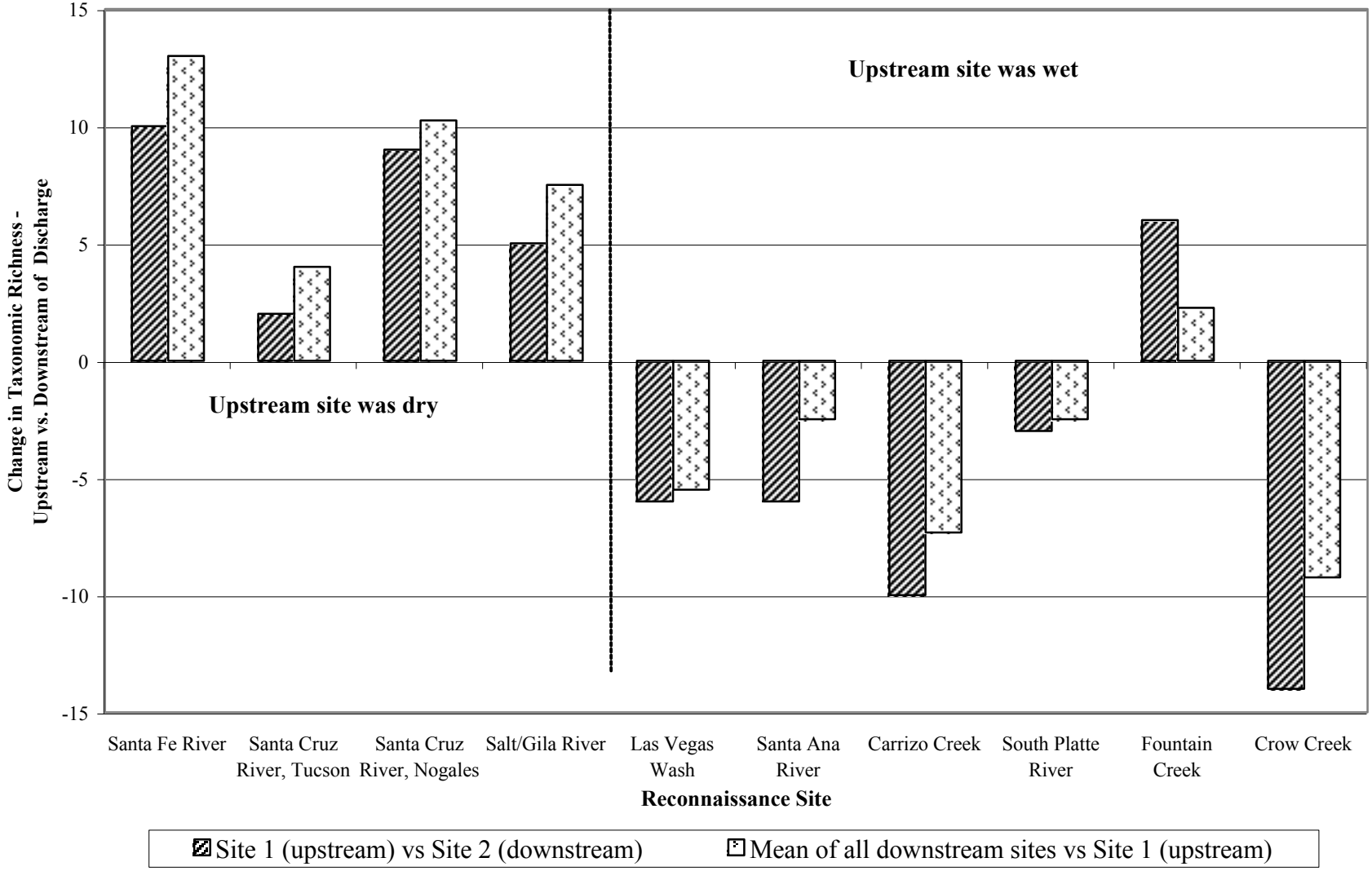
all cases. If historic data were used, the taxonomic level of identification was not modified. This comparison only evaluates changes in taxanomic richness regardless of changes in other factors, (e.g., habitat). Consequently, it must be kept in mind that even if increased taxonomic richness is found downstream of effluent discharges, the cause of this increased number of taxa could be the result of factors in addition to or other than increased flow.

Aquatic taxanomic richness was greater downstream of the effluent discharge at 5 of 10 study areas (**Figure D-26**). For four of these areas increased richness occurs simply because the riverbed upstream of the discharge is normally dry. Fountain Creek differs from these four study areas in that there typically is flow in the river upstream of the discharge from the Las Vegas Avenue facility. Five of 10 study areas had a decline in aquatic taxonomic richness downstream of the WWTP. In all cases the largest difference was between the upstream site and the site immediately below the discharge. Three of these areas have multiple wastewater effluent discharges.

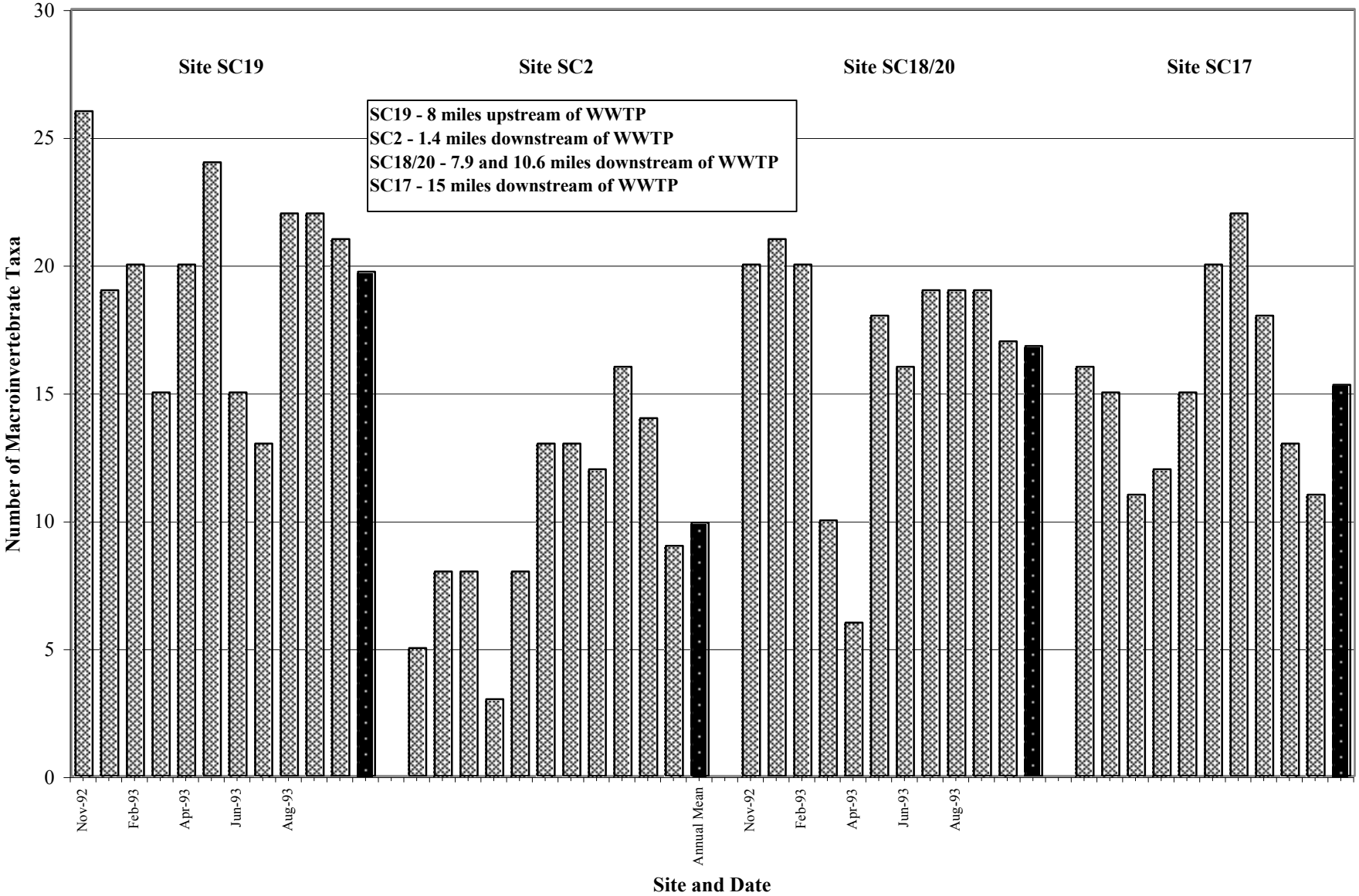
A review of historical data sets show mixed results, as described below.

Santa Cruz River, Nogales. Study conducted by Lawson (1995) showed that on a month to month basis over the period of a year taxanomic richness was typically greater upstream of the wastewater treatment discharge at Nogales (**Figures D-27 and D-28**). It should be noted that the upstream site for this study was 5.5 miles upstream of the wastewater discharge. It is also notable that without the discharge, the Santa Cruz River below Nogales would be dry for most, if not all, of the almost 20 miles of riverbed that currently have flow as a result of the effluent discharge.

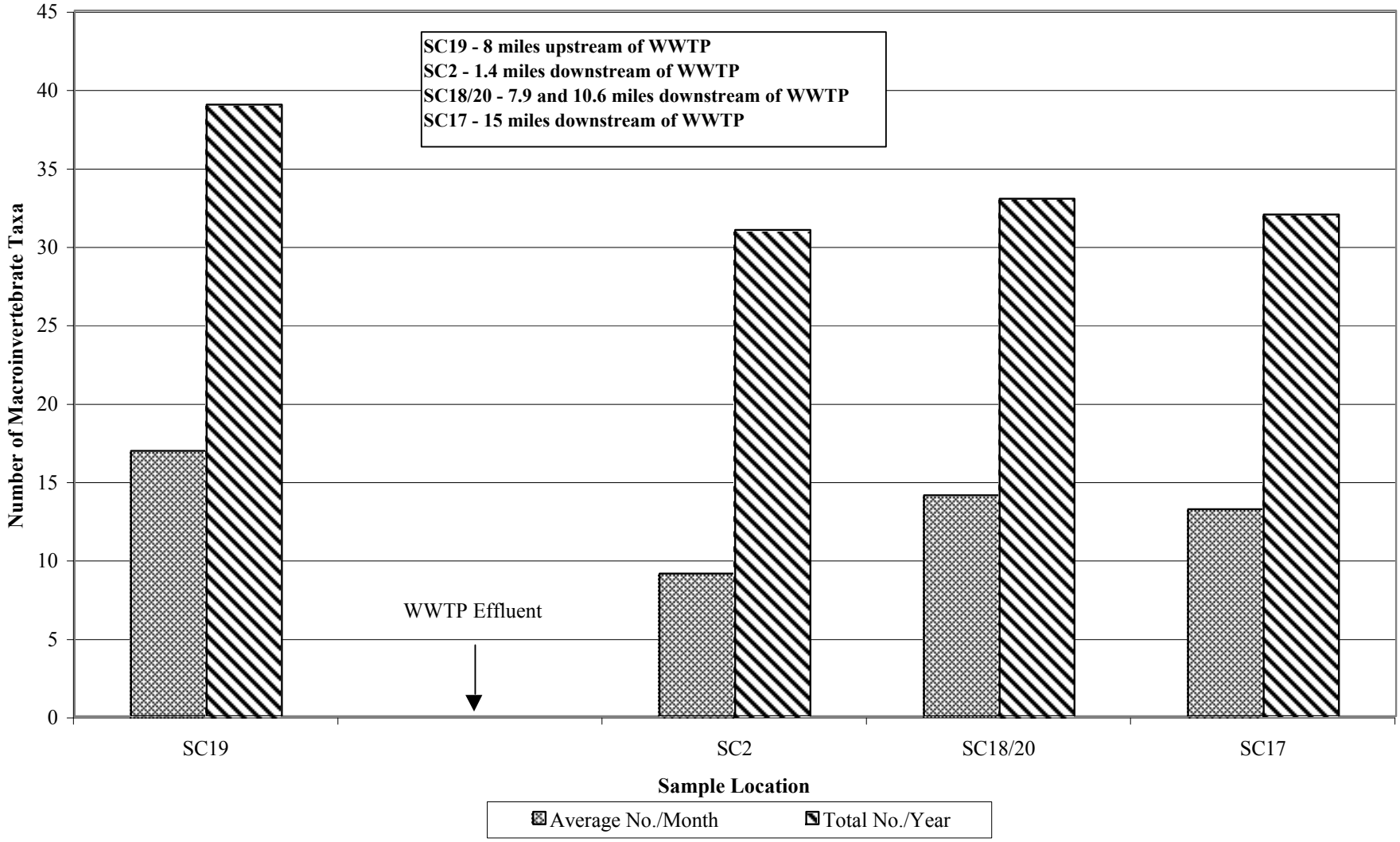
Santa Ana River. Figures D-29and D-30 suggest that macroinvertebrate taxanomic richness typically increased downstream of wastewater discharges during a study conducted in 1991. For example, mean taxanomic richness at reconnaissance sites 5 and 7 (mean of four sample dates), sites immediately below wastewater discharges, was greater below the discharge than above the discharge (**Figure D-29**). Overall, **Figure D-30** also suggests that taxanomic richness generally increased from the most upstream site to the most downstream site. Total taxanomic richness (total number of taxa over all sample dates) was much higher at reconnaissance sites 5 and 7, both of which are below wastewater treatment facility discharges (**Figure D-30**). The Santa Ana River reach, sampled quarterly in 1991, was re-sampled in August 1998. Results from that month show that taxanomic richness, while declining at sites immediately below an effluent discharge, generally increased from the most upstream site to the most downstream site suggesting that the augmentation of flow to the riverbed was beneficial in terms of numbers of macroinvertebrate species (**Figure D-31**). Fish taxanomic richness was clearly highest at the most downstream sites in 1991, sites that are downstream of wastewater treatment plant discharges (**Figures D-32 and D-33**).



Change in Species Richness: Above and Below Wastewater Treatment Effluent Discharge
Figure D-26

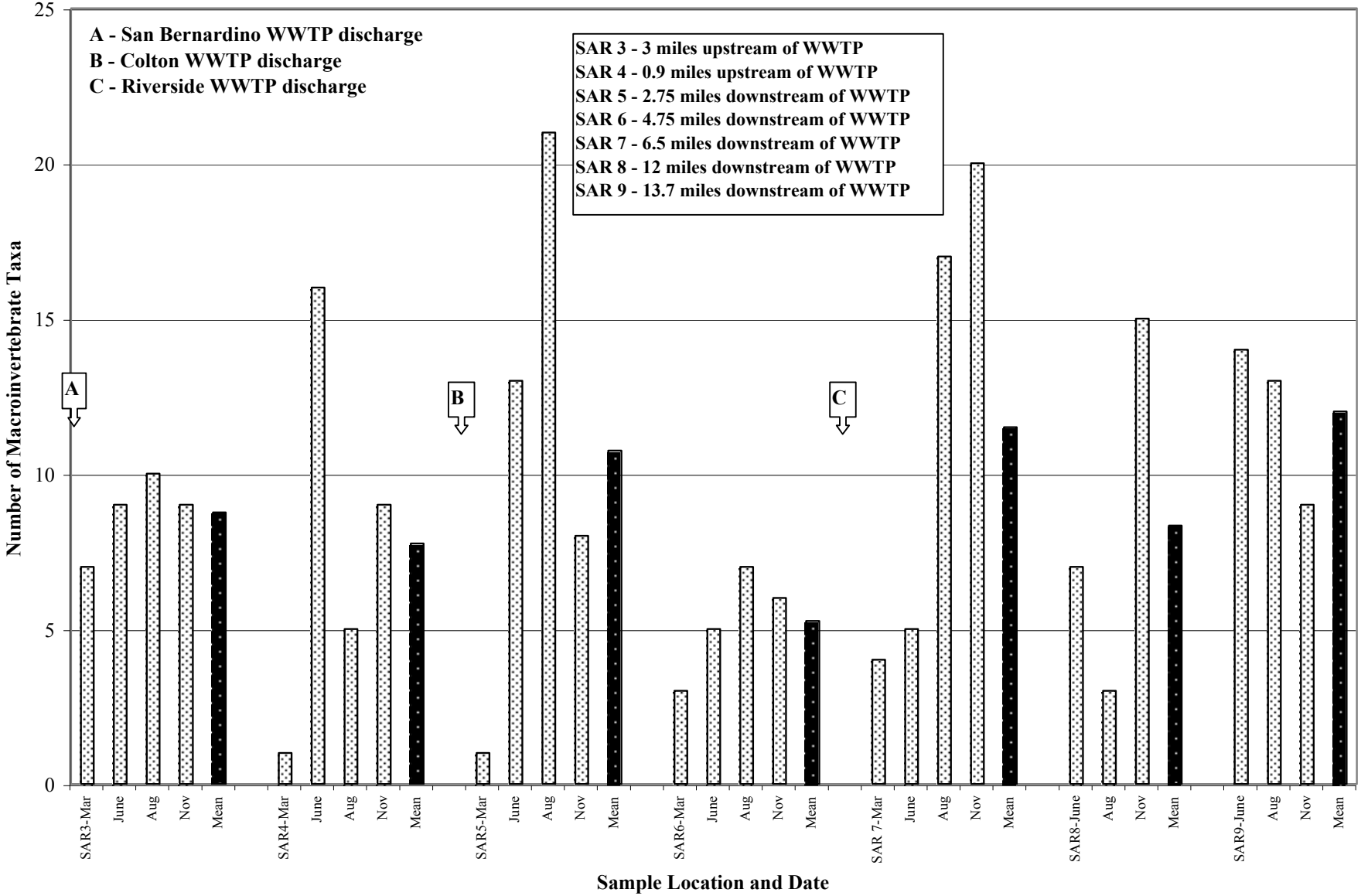


Santa Cruz River, near Nogales - Total Number of Macroinvertebrate Taxa
(Lawson 1995 - Taxa Identified to Lowest Practical Level)
Figure D-27

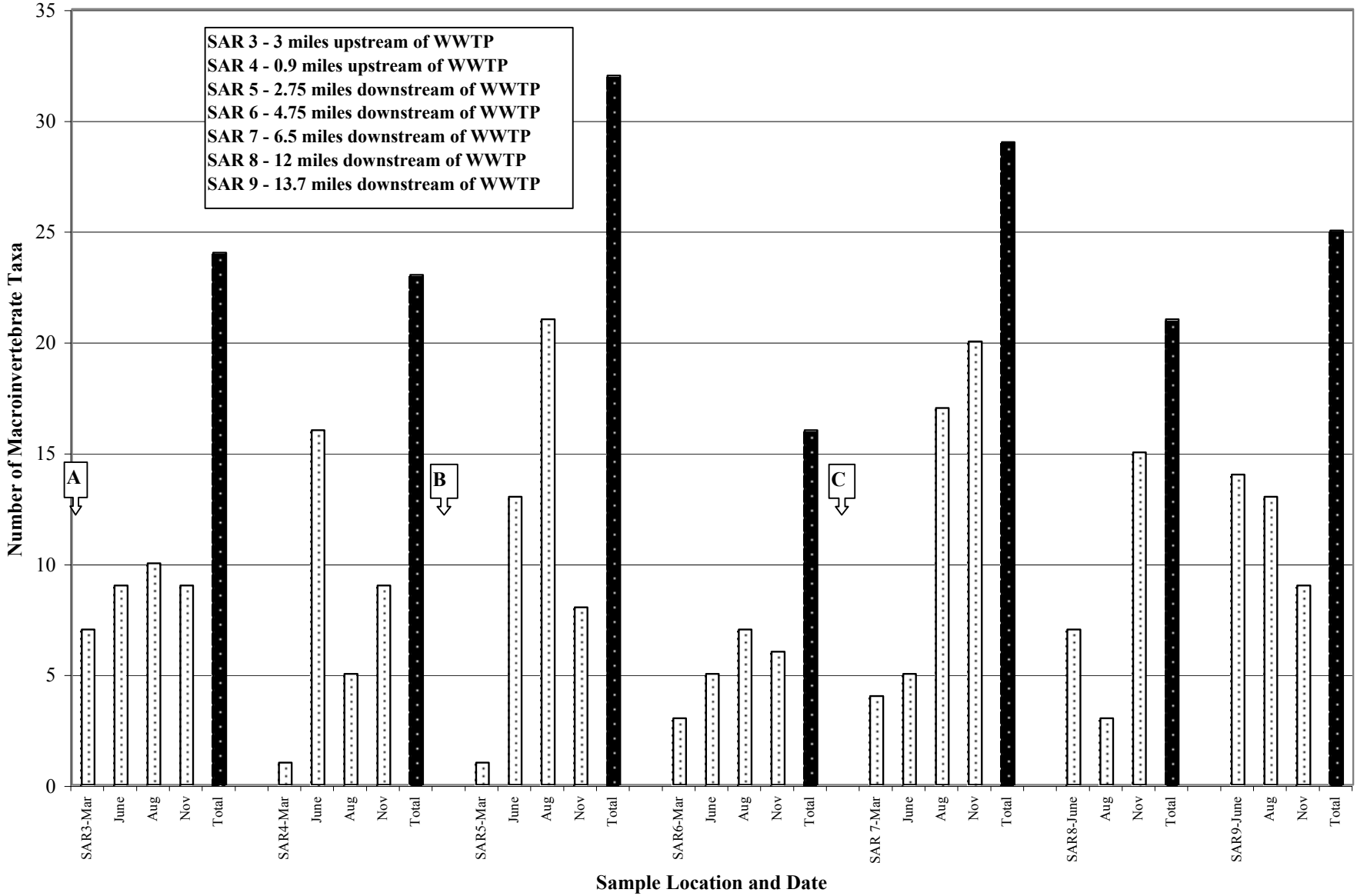


Santa Cruz River, near Nogales - Average Number of Macroinvertebrate Taxa per Month vs. Total Number of Macroinvertebrate Taxa per Year (Lawson 1995)

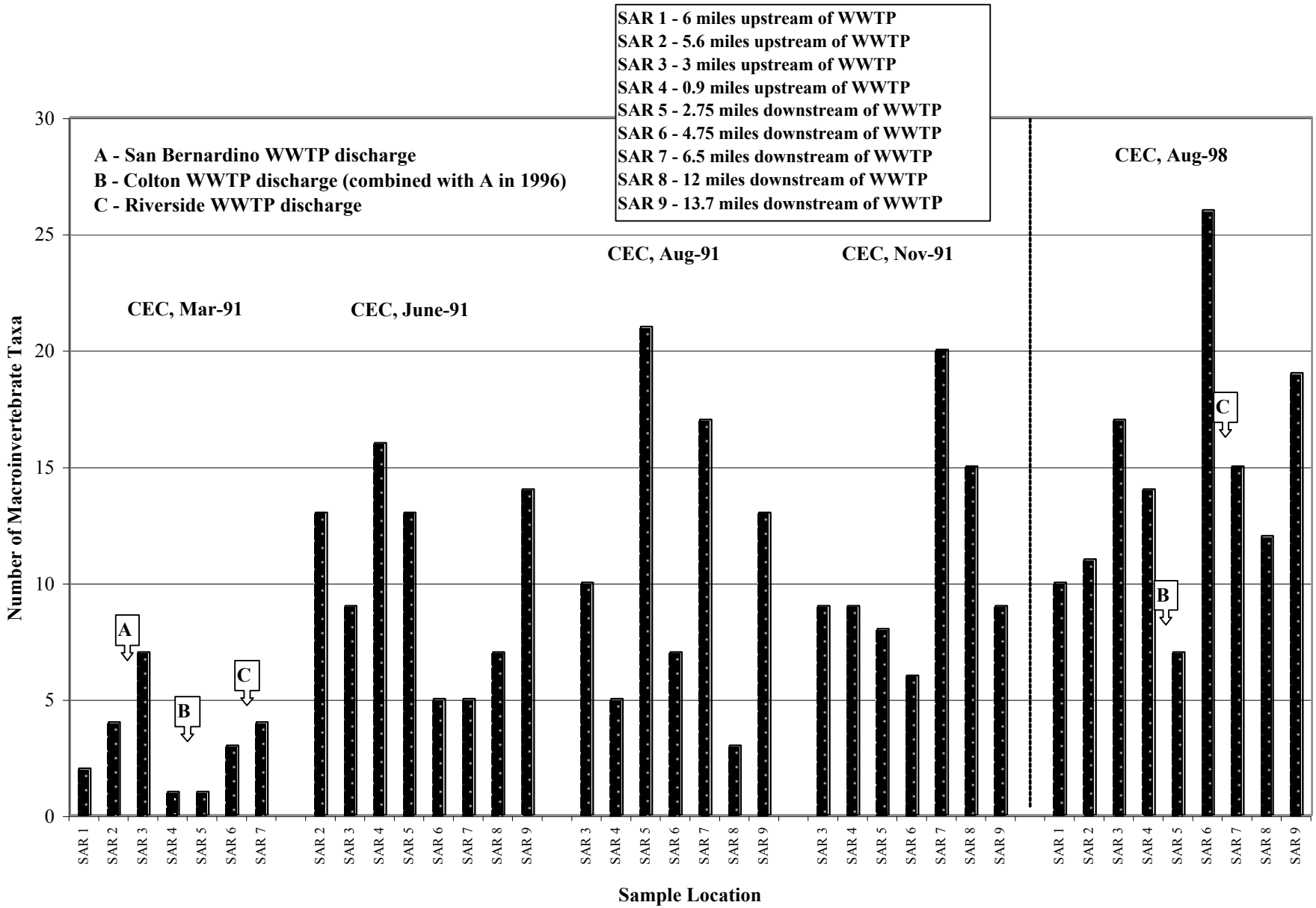
Figure D-28



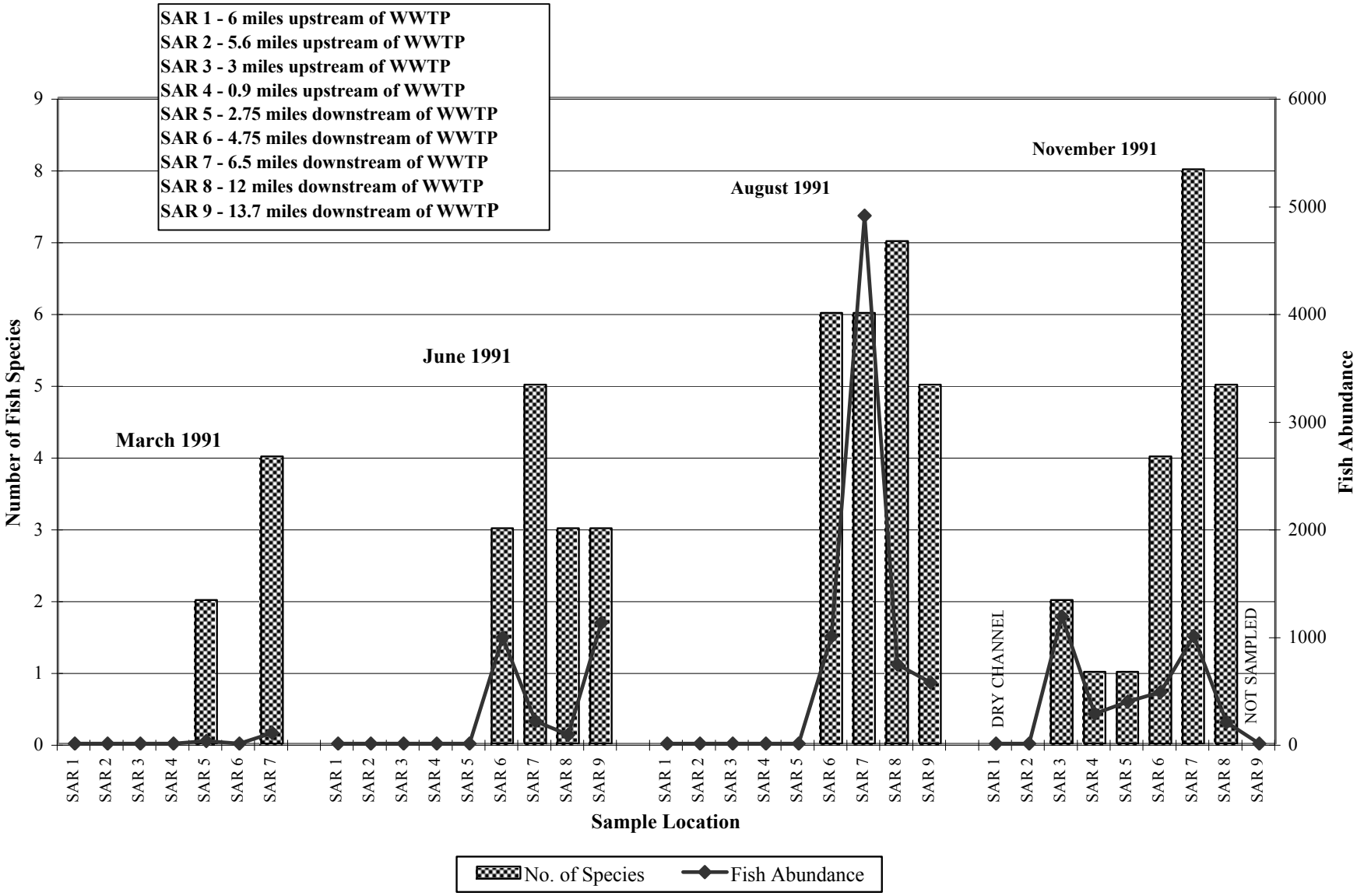
Santa Ana River - Seasonality of Macroinvertebrate Taxa: 1991 (CEC)
(Taxa Identified to Lowest Practical Level)
Figure D-29



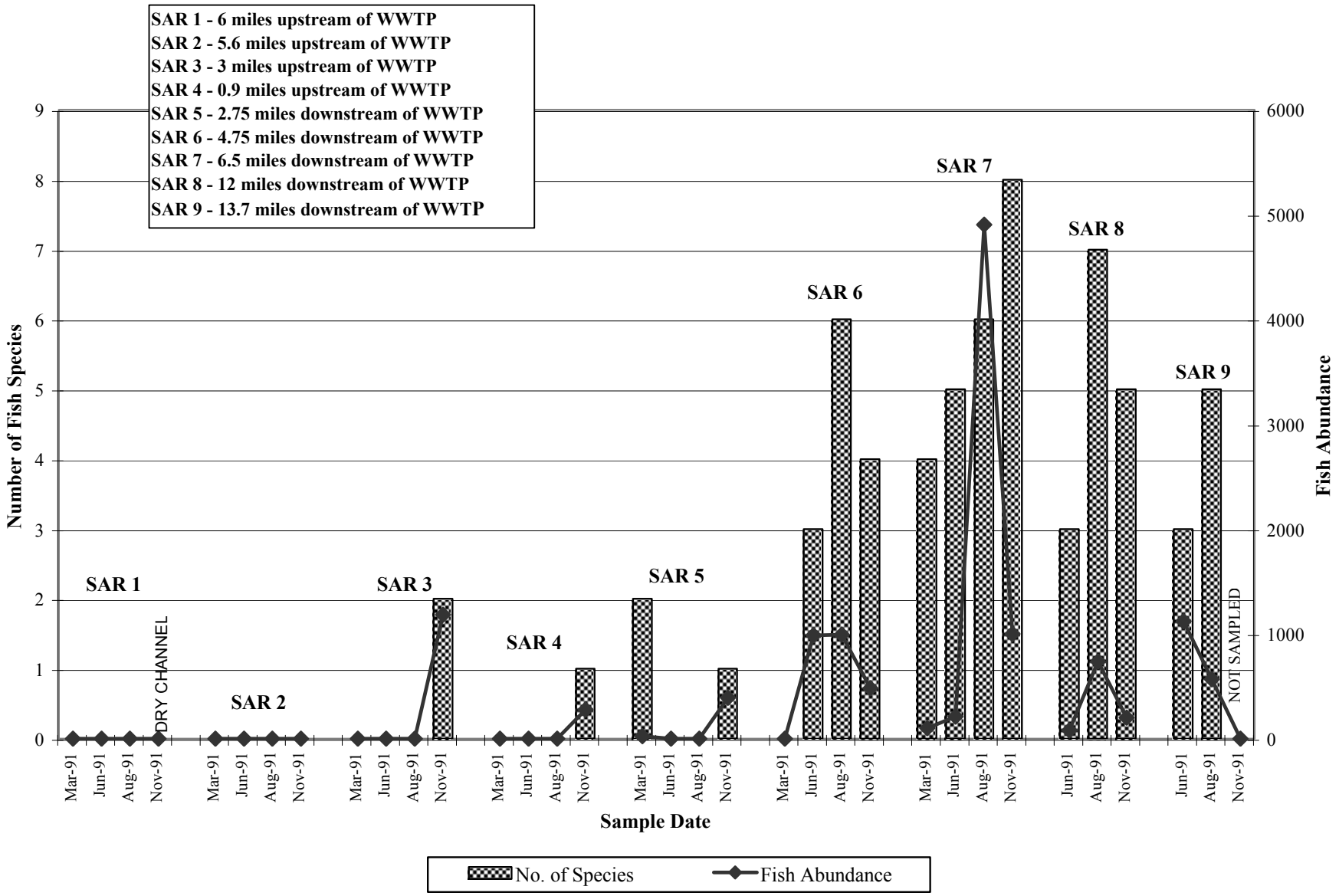
Santa Ana River - Seasonal vs. Annual Macroinvertebrate Taxa Richness: 1991 (CEC)
(Taxa Identified to Lowest Practical Level)
Figure D-30



Santa Ana River - Number of Macroinvertebrate Taxa: 1991, 1998 (CEC)
(Taxa Identified to Lowest Practical Level)
Figure D-31



Santa Ana River - Longitudinal Variation in Fish Abundance and Species Richness (CEC)
Figure D-32



Santa Ana River - Annual Variation in Fish Abundance and Species Richness (CEC)
Figure D-33

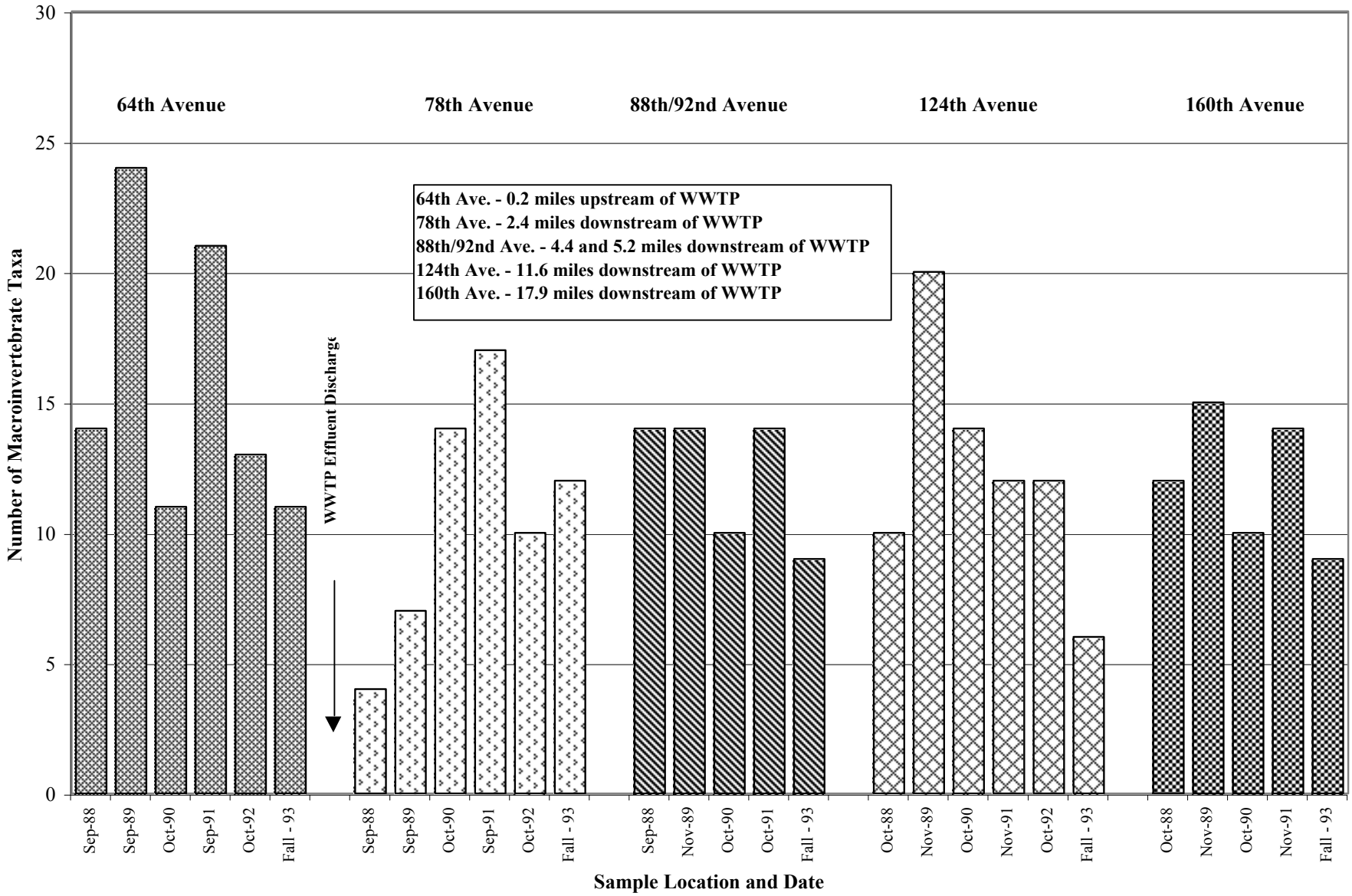
South Platte River. Between 1988 and 1993, macroinvertebrate taxanomic richness varied significantly from year to year upstream of the Denver Metro effluent discharge to the South Platte River (**Figure D-34**). In 1989 and 1991, taxanomic richness was much higher at upstream sites than at downstream sites. But in other years, 1988, 1990, 1992, and 1993, taxanomic richness tended to be the same upstream and downstream of the effluent discharge

(**Figure D-34**). Annual variation in fish taxanomic richness was also significant over an 11-year period of record. Consequently, fish taxanomic richness was greater either above or below the effluent discharge depending on the year sampled (**Figure D-35**). Over 11 years, average fish taxanomic richness at the two upstream sites (31st and 64th Avenues) was 8.4 and 9.4 species, respectively. Downstream of the discharge, taxanomic richness was 7.1, 6.7, and 9.5, at 88th, 124th, and 160th Avenues, respectively. These mean values suggest that overall fish taxanomic richness was not enhanced by the augmentation of flow resulting from the discharge.

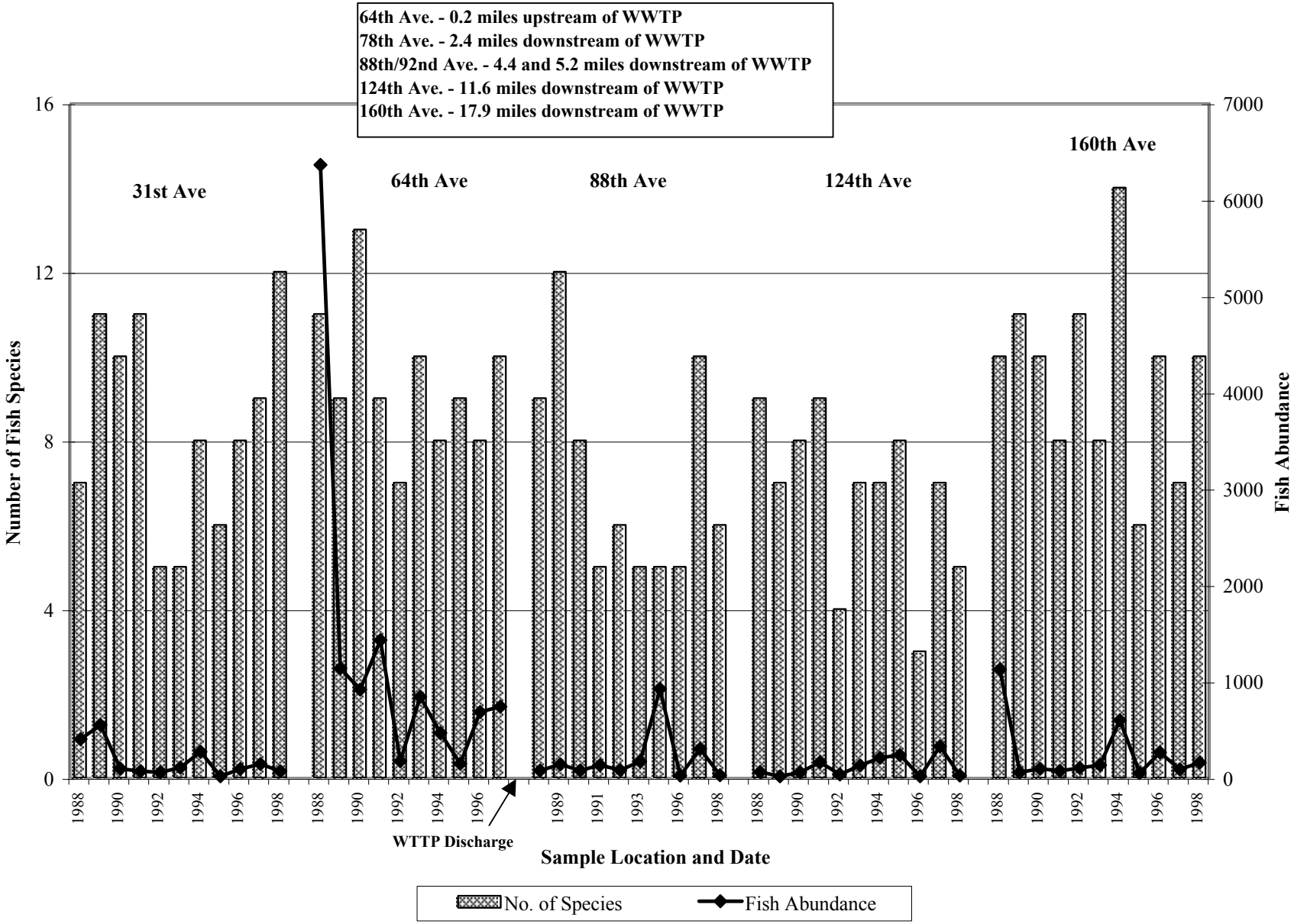
Fountain Creek. Macroinvertebrate data have been collected from Fountain Creek in 1980, 1989, 1998, and 1999. This 20-year record shows that taxanomic richness has changed considerably over this period of time (**Figures D-36 and D-37**). In 1980, richness was around 30 taxa upstream of the effluent discharge, but only about 20 taxa downstream of the discharge. By 1989, taxanomic richness had declined considerably to 7 to 10 taxa per site. Richness was similar upstream and downstream of the discharge. By 1998 (fall) and 1999, richness had generally recovered to typically 20 to 25 taxa at most sites upstream and downstream of the effluent discharge. Spring 1998 had low taxanomic richness compared to fall 1998 and 1999. The typical similarity of taxanomic richness upstream and downstream of the effluent discharge suggest that flow augmentation resulting from effluent has not increased taxanomic richness in this stream.

Crow Creek. A study conducted in 1994 on Crow Creek included two sites upstream of the effluent discharge. This study found that macroinvertebrate taxanomic richness several miles upstream of the discharge was significantly greater than taxanomic richness at a site immediately above the discharge and two sites below the discharge (**Figure D-38**). These results contrast somewhat from the site reconnaissance results, which found significantly higher macroinvertebrate richness immediately above the effluent discharge than what was found downstream of the discharge (**Figure D-39**). Fish taxa data from 1991 show similar results to the 1994 macroinvertebrate data (i.e., taxanomic richness was greatest several miles upstream of the effluent discharge, but immediately above the discharge taxanomic richness was similar to sites downstream of the addition of effluent) (**Figure D-40**).

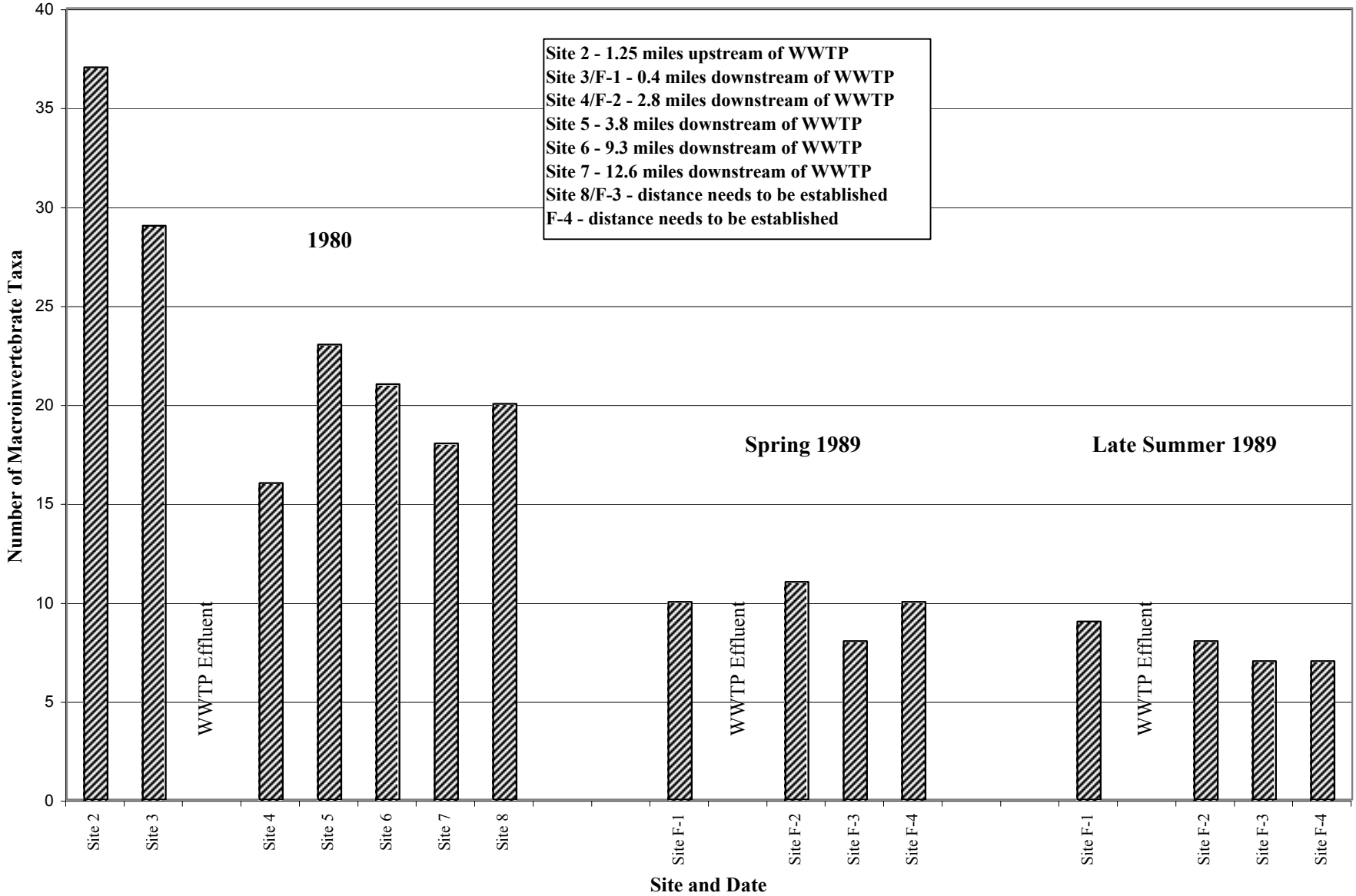
Summary. Other than streams that are dry upstream of the effluent discharge, flow augmentation does not appear to have been important for increasing taxanomic richness in arid West streams. However, it must be reiterated that for many of these streams if no discharge occurred at all, then taxanomic richness would be severely limited because of either the complete lack of water or likely flow intermittency that would occur on a seasonal basis. In addition, with regards to aquatic taxa, differences in taxanomic richness above and below effluent discharge may be related to factors other than flow augmentation, including both water quality and habitat quality.



South Platte River - Number of Macroinvertebrate Taxa:
Denver Metro Data, 1988-1993 (Taxa Identified to the Lowest Practical Level)
Figure D-34

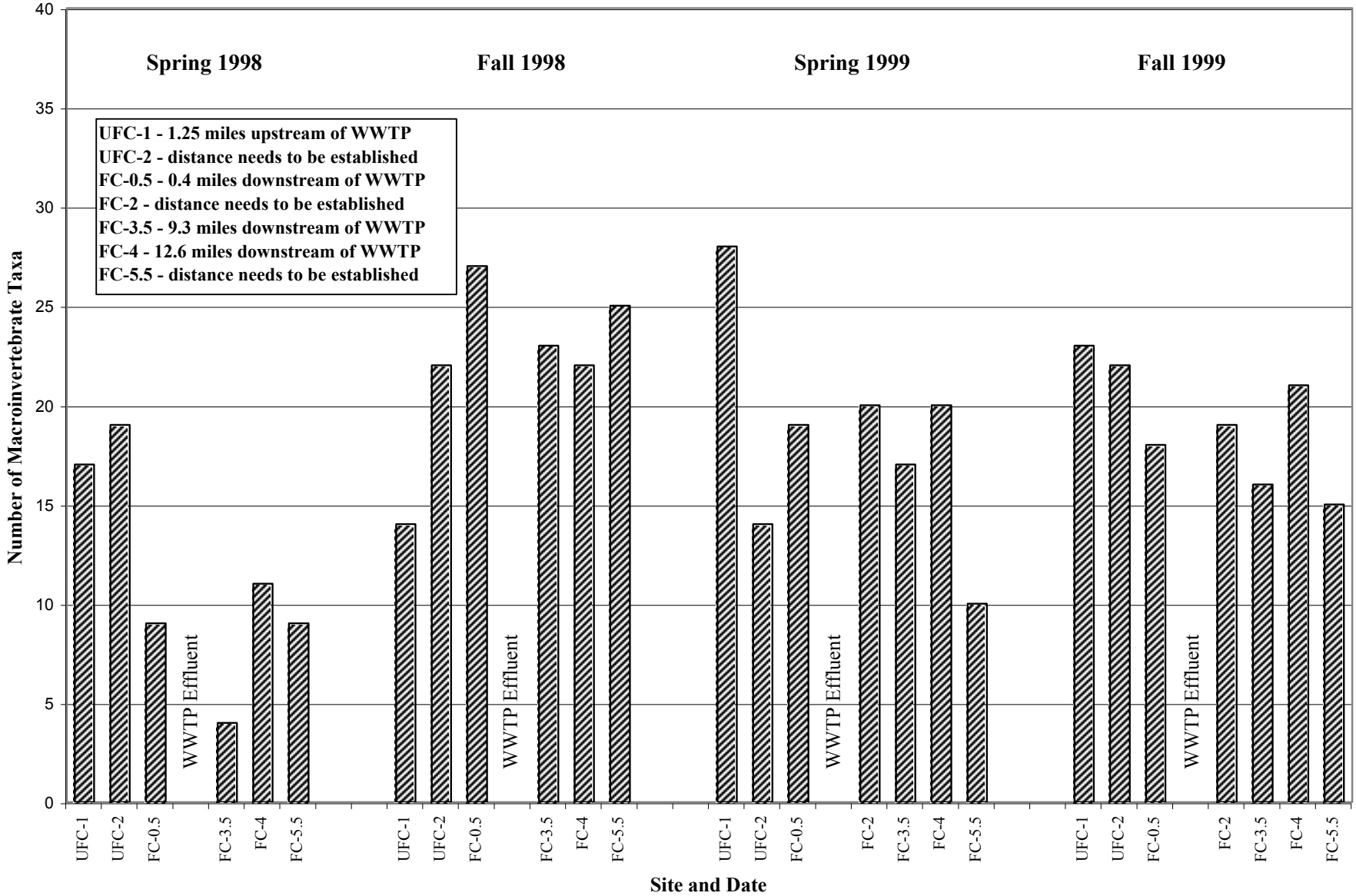


South Platte River - Annual Variation in Fish Abundance and Species Richness: Denver Metro Data 1988-1998
Figure D-35



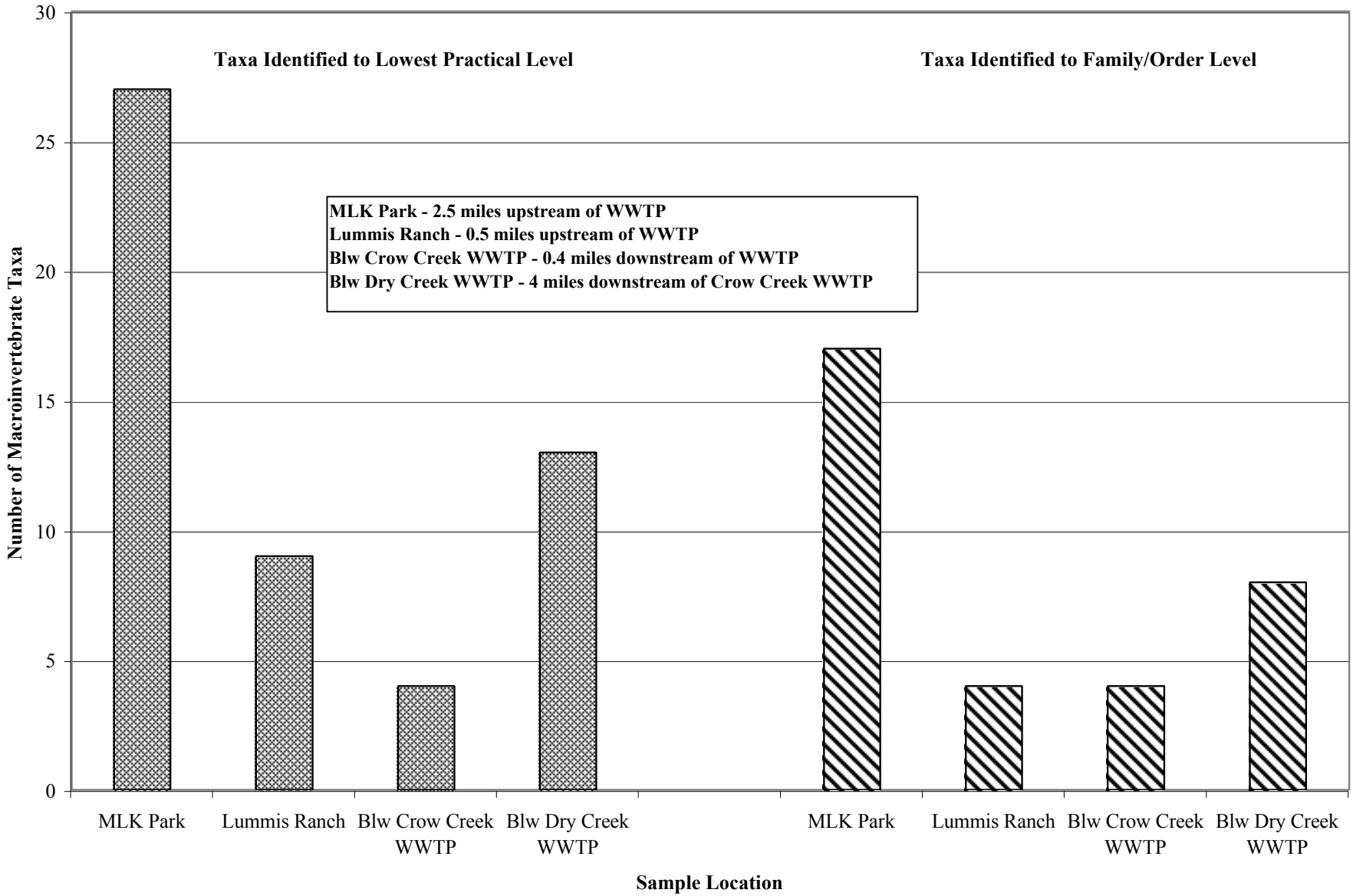
Fountain Creek - Number of Macroinvertebrate Taxa 1980 vs. 1989 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)

Figure D-36



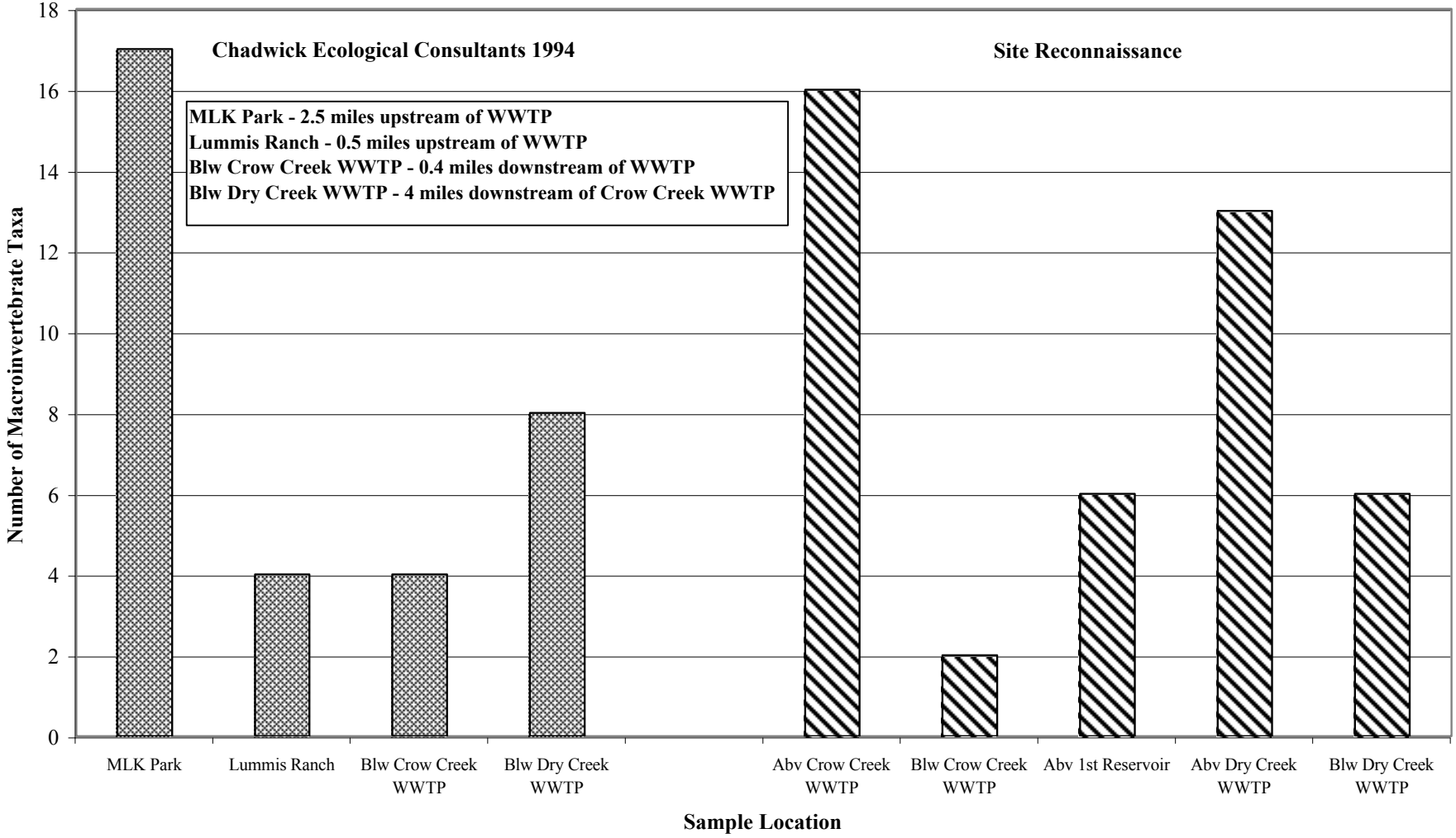
Fountain Creek - Number of Macroinvertebrate Taxa: 1998 vs. 1999 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)

Figure D-37



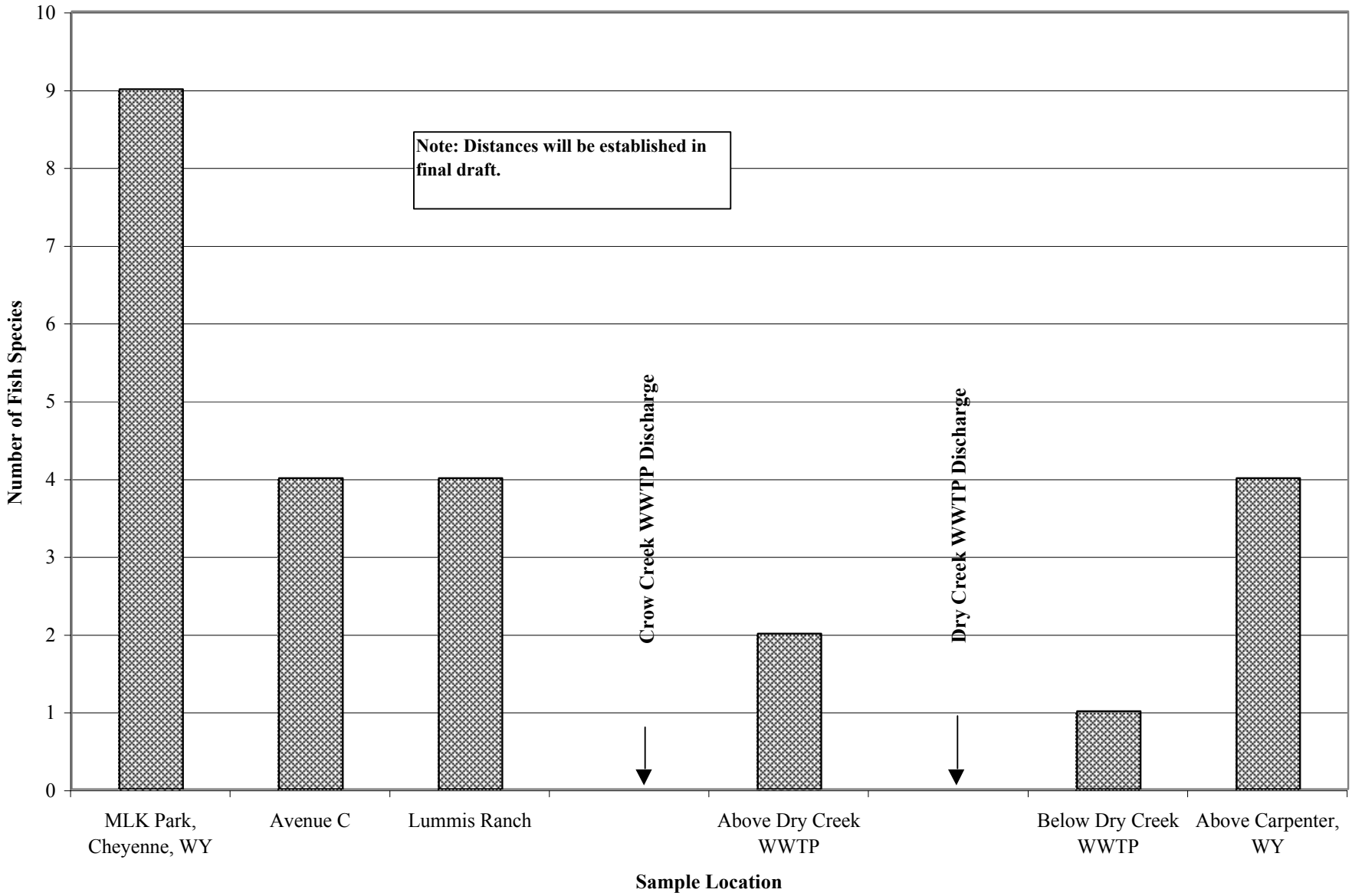
Crow Creek - Number of Macroinvertebrate Taxa: Taxonomic Identification to Lowest Practical Level vs. Family/Order Level (Chadwick Ecological Consultants 1994)

Figure D-38



Crow Creek - Number of Macroinvertebrate Taxa, 1993 (Chadwick Ecological Consultants 1994) vs. Site Reconnaissance 2000 (Taxa Identified at Family/Order Level)

Figure D-39



Crow Creek - Number of Fish Species in Crow Creek, August 1991 (Wyoming Game Fish Data reported by Chadwick Ecological Consultants 1994)
Figure D-40

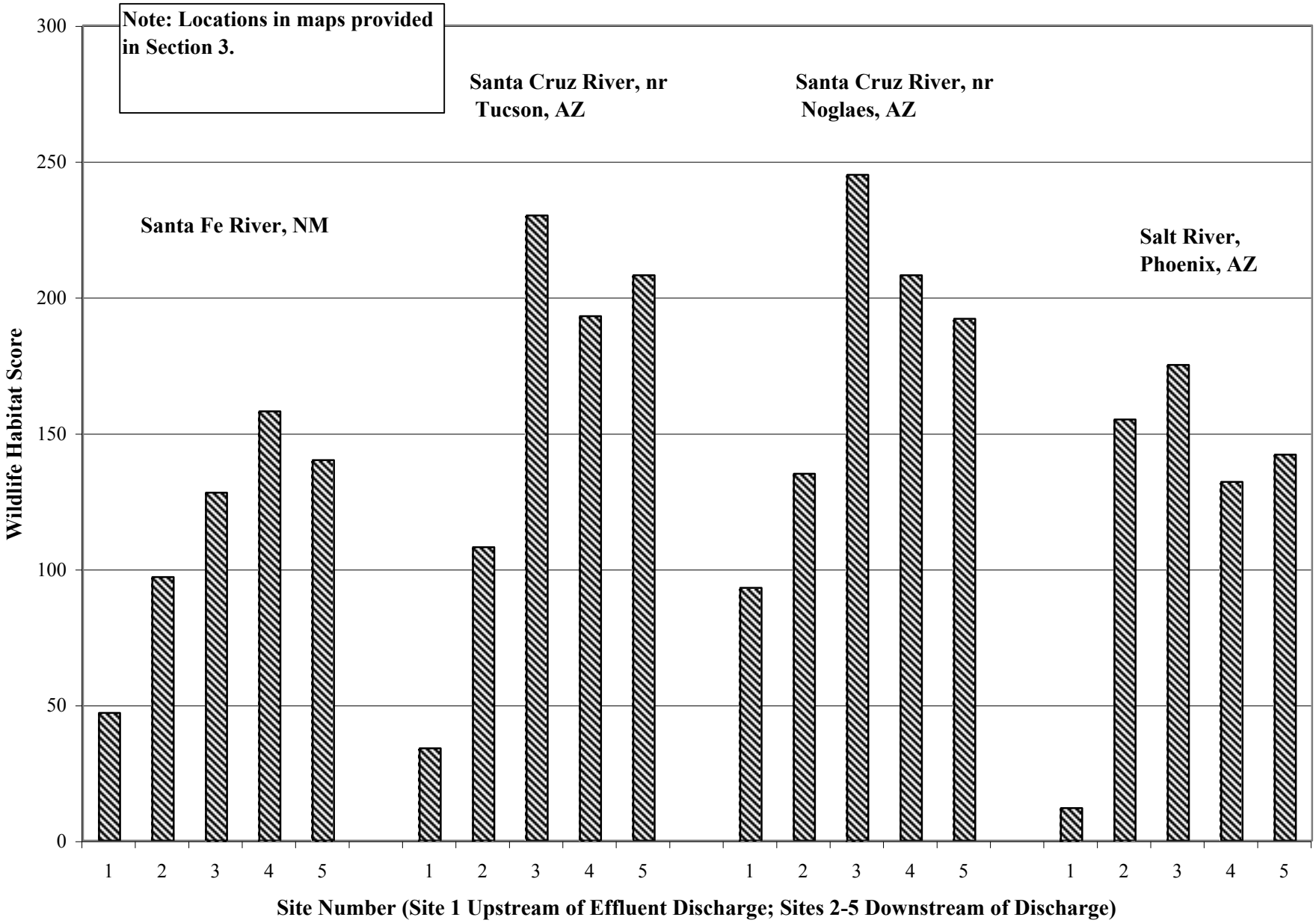
1.2.5.2 Terrestrial Species

Riparian ecosystems appear in the arid western United States in direct response to water beyond that which occurs as a result of normal precipitation events. Well-developed riparian systems are almost always associated with streams in which flow is perennial or nearly so. Lowe (1985) defines a riparian association as "one that occurs in or adjacent to drainageways and/or their floodplains and that is further characterized by species and/or life-forms different from that of the immediately surrounding nonriparian climax." This definition works quite well in the arid West, but does tend to be less applicable to eastern woodlands and streams. The presence of riparian habitats along streams, whether containing treated effluent or normal runoff, is of immense importance to all classes of wildlife. The value of riparian habitats to birds especially has been well documented (Anderson and Ohmart 1984; Carothers et al. 1974; Rosenberg et al. 1991; and many others).

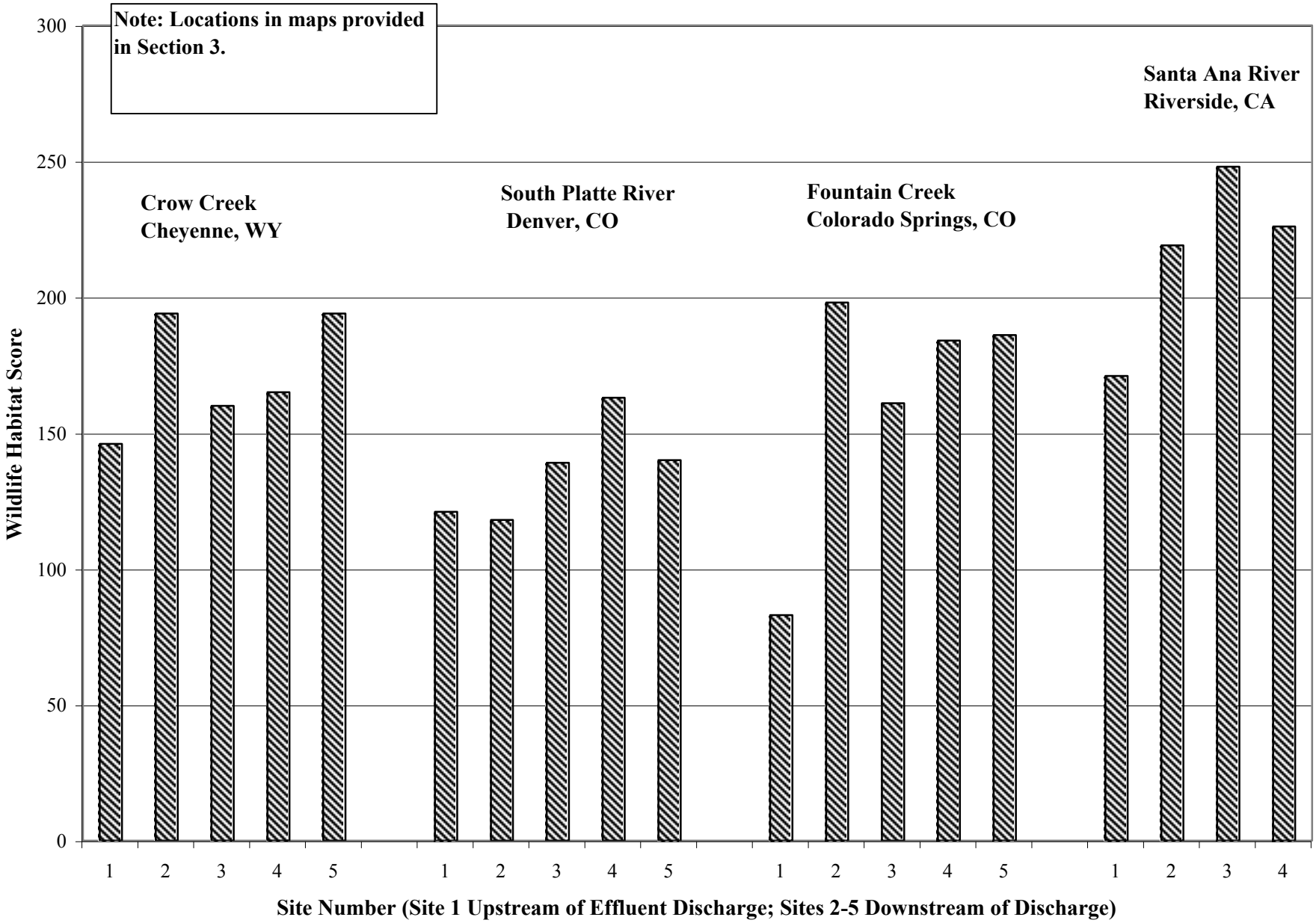
That the discharge of treated wastewater influences the presence and structure of riparian systems in otherwise dry streambeds is unequivocal. Of the set of municipal dischargers studied for this research effort, those sites where effluent was being discharged into normally dry stream channels showed marked differences in vegetation characteristics upstream from the effluent outfall compared with downstream. The project vegetation scoring scheme (**Table D-7**) clearly shows that riparian habitat values are lower upstream in those situations where upstream waters are ephemeral (**Figure D-41**). For example, upstream vegetation values at Santa Fe, Tucson, Las Vegas, and Phoenix were noticeably lower than scores for vegetation downstream of the outfall. In contrast, those sites where upstream waters were intermittent or perennial showed little or no difference in vegetation scores (e.g., Cheyenne, Denver, Colorado Springs, Santa Ana, and Carrizo Springs) (**Figure D-42**). Other habitat characteristics exhibited a very similar pattern as did potential wildlife utilization scores (**Table D-7**).

Comparisons between Study Areas and Non-Arid Areas – Terrestrial Vertebrates. The primary premise of this project is that there is a fundamental difference between effluent waste streams in the arid and semi-arid western United States and waste streams in the more humid eastern United States. One of the hypotheses derived from this premise is that the riparian zones in the arid and semi-arid West are quite different from the surrounding habitats, while in the East there is less difference between the riparian and the upland habitats. The lists of terrestrial vertebrate species for each site provide one method for testing this hypothesis. For each arid and non-arid study area, lists of potential mammals, birds, and reptiles and amphibians were compiled, based on the best available local or regional information. These lists accounted for the geographic distribution and general habitat requirements of each species. For each species, a determination was made as to whether the species would be likely to be restricted to the riparian zone or the adjacent uplands, or whether the species could use either possible habitat.

For each study area, the numbers of species that were primarily riparian, primarily upland, or able to use both habitats were determined for the three groups of terrestrial vertebrates. These numbers are listed in **Table D-8** for the study areas and **Table D-9** for the non-arid areas. Because of relatively small numbers of amphibians at most of the western sites, these were lumped with the reptiles. With large differences in the total numbers of species among sites,



Terrestrial Habitat Values: Sites Where Effluent Constitutes Almost 100% of Instream Flow
Figure D-41



Terrestrial Habitat Values: Sites Where Effluent Constitutes Majority of Instream Flow
Figure D-42

direct comparisons were not possible, and the data were standardized by taking percentages of each group in each habitat category.

Table D-7
Terrestrial Habitat Characterization - Standardized Scores for Each Study Area

River	Site Name	TBRV	TBHC	TBWO	TBDC
Santa Fe River	SFR1	15	0	32	11
	SFR2	26	8	63	4
	SFR3	44	33	51	27
	SFR4	37	42	79	42
	SFR5	48	36	56	42
Santa Cruz River, Tucson	SCRT1	26	8	0	23
	SCRT2	33	42	33	46
	SCRT3	74	89	67	42
	SCRT4	70	72	51	27
	SCRT5	74	83	51	65
Santa Cruz River, Nogales	SCRN1	48	19	26	0
	SCRN2	19	53	63	12
	SCRN3	70	94	81	42
	SCRN4	48	83	77	27
	SCRN5	48	67	77	15
Crow Creek, Cheyenne	CC1	74	28	44	69
	CC2	89	28	77	38
	CC3	41	47	72	46
	CC4	67	31	67	54
	CC5	70	47	77	46
Las Vegas Wash	LVW1	22	56	37	35
	LVW2	48	89	77	50
	LVW3	26	58	60	65
	LVW4	63	61	77	65
	LVW5	11	67	60	65
Salt River, Phoenix	SR1	0	3	9	42
	SR2	37	44	74	35
	SR3	52	72	51	12
	SR4	30	53	49	12
	SR5	33	67	42	46
South Platte River, Denver	SPR1	30	75	16	27
	SPR2	26	22	70	38
	SPR3	26	50	63	50
	SPR4	26	72	65	38
	SPR5	30	50	60	31
Fountain Creek, Colorado Springs	FC1	41	42	0	35
	FC2	93	42	63	0
	FC3	44	70	47	27
	FC4	59	67	58	35
	FC5	52	69	65	69
Santa Ana River, San Bernardino	SAR1	66	72	33	54
	SAR2	59	83	77	50
	SAR3	70	78	100	54
	SAR4	59	100	67	27
Carrizo Creek, Carrizo	CS1	74	78	33	100

Table D-7
Terrestrial Habitat Characterization - Standardized Scores for Each Study Area

River	Site Name	TBRV	TBHC	TBWO	TBDC
Springs	CS2	100	47	51	92
	CS3	89	39	26	65
	CS4	74	33	33	65

TBRV Terrestrial Biology - Riparian vegetation
TBHC Terrestrial Biology - Habitat characteristics
TBWTE Terrestrial Biology - Wildlife, including threatened, endangered, and sensitive species
TBDC Terrestrial Biology - Disturbance characteristics

Table D-8
Numbers of Potential Species at Each of the Study Areas, Sorted by Numbers of Species Restricted to the Riparian Habitats, Restricted to Upland Habitats, or Able to Use Both Habitats

Study Site/Habitat	Mammals		Birds		Reptiles and Amphibians	
	Number of Species	Percent	Number of Species	Percent	Number of Species	Percent
Santa Ana River, San Bernardino, California						
Riparian	4	8.2	95	49.2	12	31.6
Upland	7	14.3	16	8.3	12	31.6
Both	38	77.5	82	42.5	14	36.8
Total	49		193		38	
Las Vegas Wash, Las Vegas, Nevada						
Riparian	9	17.6	123	52.8	5	13.2
Upland	7	13.8	8	3.4	21	55.3
Both	35	68.6	102	43.8	12	31.6
Total	51		233			
Santa Cruz River, Nogales, Arizona						
Riparian	6	10.3	81	47.1	12	21.1
Upland	14	24.2	12	7.0	18	31.5
Both	38	65.5	79	45.9	27	47.4
Total	58		172		57	
Santa Cruz River, Tucson, Arizona						
Riparian	7	13.7	78	44.6	9	18.0
Upland	10	19.6	15	8.5	23	46.0
Both	34	66.7	82	46.9	18	36.0
Total	51		175		50	
Salt River, Phoenix, Arizona						
Riparian	5	10.4	81	46.6	5	10.4
Upland	10	20.8	19	10.9	24	50.0
Both	23	68.8	74	42.5	19	39.6
Total	48		174			
Crow Creek, Cheyenne, Wyoming						
Riparian	10	22.7	98	52.1	9	36.0
Upland	13	29.6	21	11.2	5	20.0
Both	21	47.7	69	36.7	11	44.0
Total	44		188		25	

Table D-8
Numbers of Potential Species at Each of the Study Areas, Sorted by Numbers of Species Restricted to the Riparian Habitats, Restricted to Upland Habitats, or Able to Use Both Habitats

Study Site/Habitat	Mammals		Birds		Reptiles and Amphibians	
	Number of Species	Percent	Number of Species	Percent	Number of Species	Percent
South Platte River, Denver, Colorado						
Riparian	13	24.1	105	53.0	8	28.6
Upland	13	24.1	18	9.1	5	17.9
Both	28	51.8	75	37.9	15	53.6
Total	54		198		28	
Fountain Creek, Colorado Springs, Colorado						
Riparian	10	15.4	102	51.0	9	26.5
Upland	19	29.2	20	10.0	8	23.5
Both	36	55.4	78	39.0	17	50.0
Total	65		200		34	
Santa Fe River, Santa Fe, New Mexico						
Riparian	12	17.6	89	43.0	8	17.0
Upland	10	14.7	28	13.5	18	38.3
Both	42	67.7	90	43.5	21	44.7
Total	68		207		47	
Carrizo Creek, Carrizo Springs, Texas						
Riparian	5	9.6	81	13.7	10	21.7
Upland	11	21.2	26	42.9	13	28.3
Both	36	69.2	82	43.4	23	50.0
Total	52		189		46	
Study Site Averages and Standard Deviations	**Mean Percent**	**S.D.**	**Mean**	**S.D.**	**Mean**	**S.D.**
Riparian	14.96	5.539	48.23	3.945	22.41	8.193
Upland	21.15	5.767	9.56	3.069	34.24	12.830
Both	63.88	9.216	42.21	3.344	43.37	7.159

Table D-9
Numbers of Potential Species at Each of the Non-Arid Areas, Sorted by Numbers of Species Restricted to the Riparian Habitats, Restricted to Upland Habitats, or Able to Use Both Habitats

Study Site/Habitat	Mammals		Birds		Reptiles and Amphibians	
	Number of Species	Percent	Number of Species	Percent	Number of Species	Percent
Kansas River, Topeka, Kansas						
Riparian	8	15.7	72	32.7	16	27.6
Upland	8	15.7	21	9.5	7	12.1
Both	35	68.6	127	57.7	35	60.3
Total	51		220		58	
French Broad River, Asheville, North Carolina						
Riparian	10	16.1	20	12.8	13	20.6
Upland	8	12.9	10	6.4	7	11.1

Table D-9
Numbers of Potential Species at Each of the Non-Arid Areas, Sorted by Numbers of Species Restricted to the Riparian Habitats, Restricted to Upland Habitats, or Able to Use Both Habitats

Study Site/Habitat	Mammals		Birds		Reptiles and Amphibians	
	Number of Species	Percent	Number of Species	Percent	Number of Species	Percent
Both	44	71.0	126	80.8	43	68.3
Total	62		156		63	
Ararat River, Ararat, North Carolina						
Riparian	8	18.2	40	23.1	11	23.9
Upland	4	9.1	11	6.4	4	8.7
Both	32	72.7	122	70.5	31	67.4
Total	44		173		46	
Tar River, Rocky Mount, North Carolina						
Riparian	8	19.0	62	32.0	24	32.9
Upland	4	9.5	12	6.2	9	12.3
Both	30	71.5	120	61.9	40	54.8
Total	42		194		73	
Study Site Averages and Standard Deviations						
Riparian	17.25	1.601	25.15	1.586	26.25	5.275
Upland	11.80	3.109	7.13	9.321	11.05	1.652
Both	70.95	1.721	67.72	10.216	62.70	6.367

If our hypothesis is correct, the selected non-arid streams should have a greater proportion of species that are found in both the riparian and the upland habitats, while these habitats in the West should have fewer species in common. Comparisons of the average percentages of species in both habitats between the study areas and the reference sites can be tested statistically. The *t*-test can be used to test for significant differences between means of the arid and non-arid areas.

For mammals, the mean percentage of species using both habitats is 70.95 for the non-arid areas and 63.89 for the study areas. These numbers suggest that there is less difference between the riparian and upland habitats in the non-arid areas than in the study areas. However, a statistical test ($t_s = 1.486$, $t_{0.05,12} = 2.179$) shows that there is no difference between these means at the 95 percent confidence level. The total number of degrees of freedom for this comparison is 12 (d.f. $= n_1 + n_2 - 2$). The lack of a significant difference for mammals may be a result of relatively few mammals in any part of the country being restricted to either a riparian or an upland habitat. Most mammals in all parts of the country are capable of using both habitats.

For birds, the mean percentage of species using both habitats is 67.72 for the non-arid areas and 42.21 for the study areas. These numbers indicate much less difference between the riparian and upland habitats in the non-arid areas than in the study areas. The statistical test ($t_s = 7.344$, $t_{0.05,12} = 2.179$) confirms that there is a significant difference between these means at the 95 percent confidence level. The primary reason for this difference in bird distributions is probably related to the high proportion of bird species in the arid and semi-arid areas that are restricted to riparian habitats. In the mesic conditions of the East, fewer birds are restricted to the riparian zones adjacent to the rivers.

For reptiles and amphibians, the mean percentage of species using both habitats is 62.70 for the non-arid areas and 43.37 for the arid study areas. These numbers also indicate much less difference between the riparian and upland habitats in the non-arid areas than in the study areas. The statistical test (t_s = 4.688, $t_{0.05,12}$ = 2.179) confirms that there is a significant difference between these means at the 95 percent confidence level. The difference for this group appears to be more complicated than that for birds. In the most arid study areas (Las Vegas, Phoenix, and Tucson) there are very few amphibians, and many of the reptiles are primarily upland species. In the least arid study areas (Cheyenne, Denver, and Colorado Springs) more amphibians are present in the riparian habitats, and fewer reptiles are limited to upland areas, but the proportion of species using both habitats is comparable to the arid areas. In the non-arid areas, very few reptiles and amphibians are limited to upland sites, a moderate number are limited to riparian habitats, and many species use both habitats.

Sensitive Species in Non-Arid Areas. Sensitive species, including state and federally listed species, potentially occur in the vicinity of all three river systems under study in North Carolina. There is a total of 20 sensitive species known to occur in Surry County (Ararat River). Of these, two are federally listed and all are listed by the state of North Carolina. None of the Surry County species are expected to use riparian or aquatic habitats associated with the Ararat River (**Table D-9**).

In Buncombe County (French Broad River) there is a total of 112 state and federally listed species of plants and animals. Of this total, nine are federally listed as threatened or endangered and an additional 29 are identified as federal species of special concern. All 112 species are listed by the state. Thirty-four of the 112 species are likely to use aquatic or riparian habitats associated with the French Broad River including 2 listed species (Gray Bat and Spotfin Chub) and 11 species of concern (**Table D-9**).

In Edgecombe County (Tar River) there is a total of 11 species listed by the federal government and/or the state of North Carolina. Only one of the 11 is federally listed (red-cockaded woodpecker) and it is unlikely to use riparian habitats associated with the Tar River. There are three species of federal concern in the county, and a total of nine listed species likely to be associated with habitats along the river (**Table D-9**).

1.2.6 Ambient Background Concentrations

Table D-10 shows the criteria developed by EPA for protection of aquatic life for arsenic, mercury, and selenium. These elements can have natural (background) occurrence in groundwater and streams. These metals were selected for a cursory comparison of the available data from the 10 study areas and the national criteria.

The summary of all water quality data for the 10 study areas is contained in **Appendix R**. Data for these metals are limited except for the Fountain Creek study area. The data from this summary suggest that a background concentration of selenium is higher than the national criteria for Fountain Creek. In fact, studies in this basin confirmed that the elevated selenium concentrations are a result of natural weathering of marine shales adjacent to Fountain Creek. Sampling of benthic invertebrate and fish communities indicated that these populations were not

adversely affected by the elevated selenium. This unique condition was recognized by state and federal agencies through the establishment of site-specific selenium standards based on these elevated levels.

Table D-10
National Criteria to Protect Aquatic Life

Criteria	Arsenic	Mercury	Selenium
CMC (Acute)	340 micrograms per Liter (µg/L)	1.4 µg/L	–
CCC (Chronic)	150 µg/L	0.77 µg/L	5 µg/L
Fountain Creek [a]	1/1/24	0/0/61	-/28/71

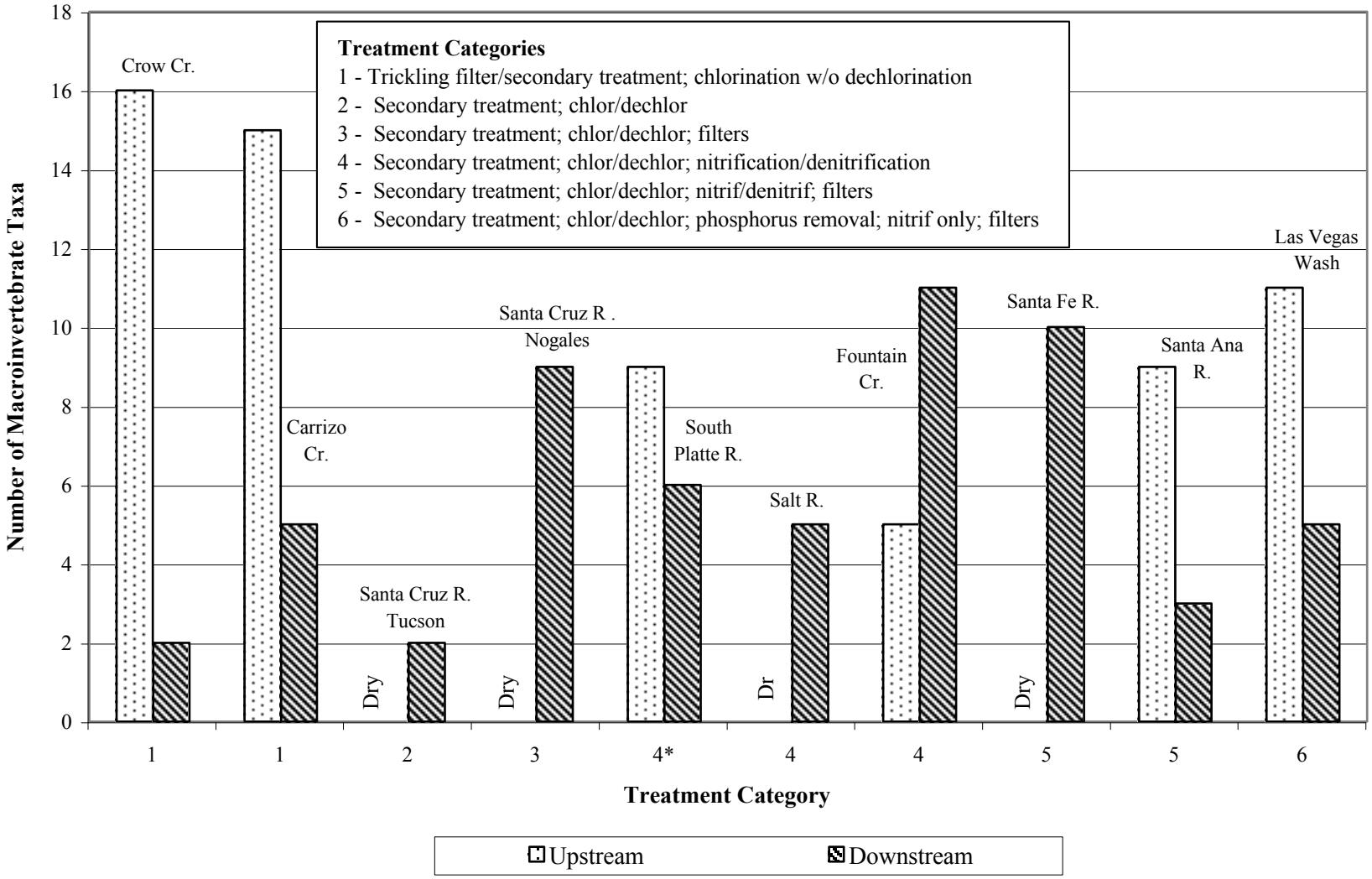
[a] Data summary indicates number of samples above the CMC (first number), number of samples above the CCC (second number), and total number of sample measurements (third number).

1.2.7 Wastewater Treatment Improvements

It is presumed that improvements in wastewater treatment resulting in improved effluent quality will result in increased species richness downstream of the discharge of treated wastewater. In most instances, the most stringent water quality criterion is associated with the protection of aquatic life. Because National Pollutant Discharge Elimination System (NPDES) permits for discharges to arid West streams are often established with the presumption that the critical low flow value is zero (i.e., no allowance for instream dilution), the effluent limitation incorporated into NPDES permits is equivalent to the water quality standard. If these standards are set at a level to protect 95 percent of all aquatic species regardless of their presence or absence (as is the presumption if national criteria are used), then one should expect that wastewater treatment improvements should result in increased aquatic species richness.

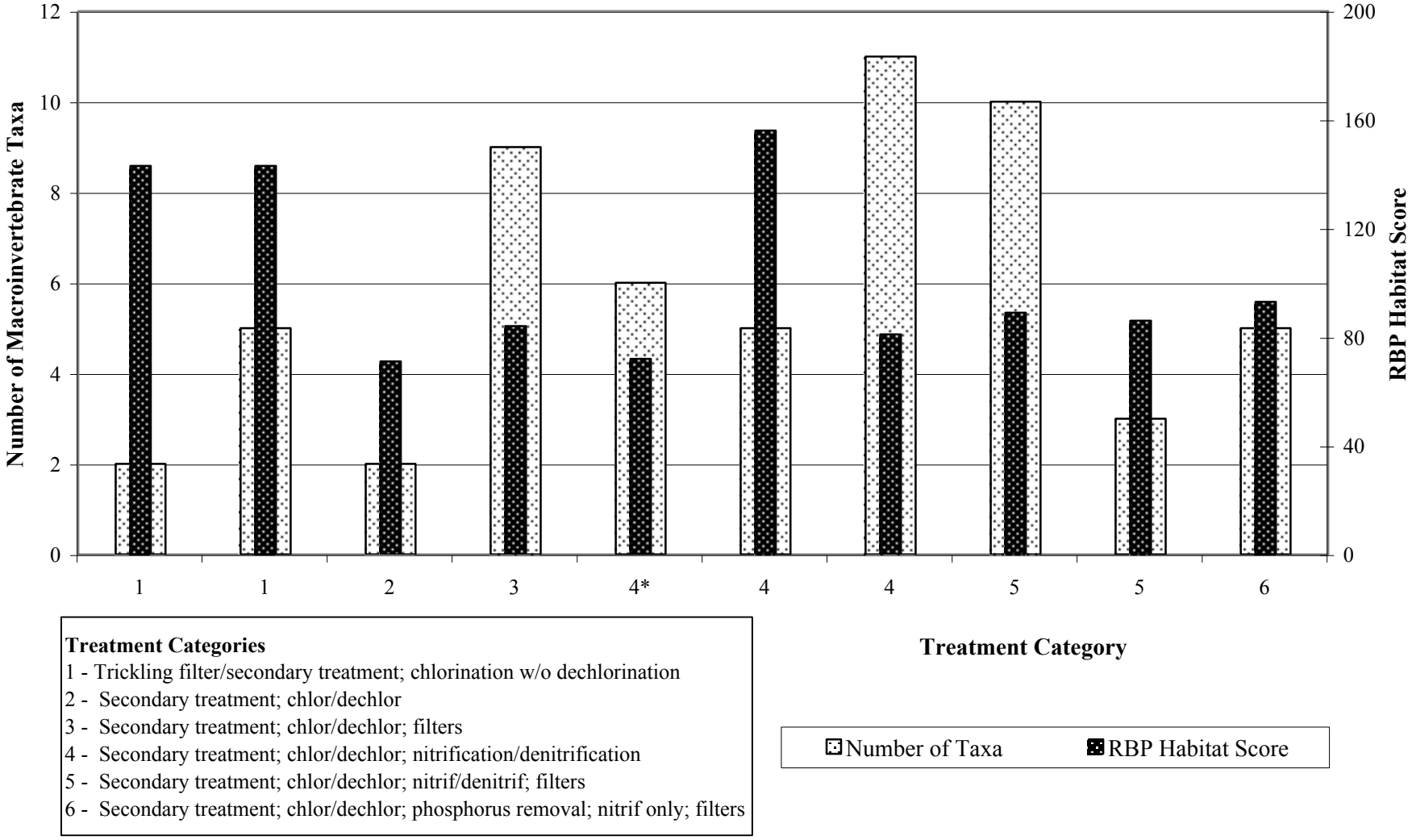
With the data available, this investigation item can be evaluated in two ways: (1) comparing aquatic communities observed upstream and downstream of the effluent discharge during the site reconnaissance in the context of treatment levels; and (2) evaluating long-term changes in aquatic communities at a specific sites in the context of changes in wastewater treatment levels over the same period of time.

A comparison of treatment levels and taxonomic richness (samples collected during the site reconnaissance) found no consistent pattern associated with improved treatment levels. Instead, richness was variable across all study areas (**Figure D-43**), and this variability could not be associated with habitat quality (**Figure D-44**). While treatment level did not appear to influence richness, it did appear to influence taxonomic composition. The distinction was fairly dramatic. Study areas with the highest level of treatment (i.e., nitrification/denitrification and filtration) had a relatively high proportion of *Ephemeroptera, Plecoptera,* and *Trichoptera* (EPT) organisms present (primarily mayflies), while sites without filtration had few, if any, EPT organisms (**Figure D-45**).



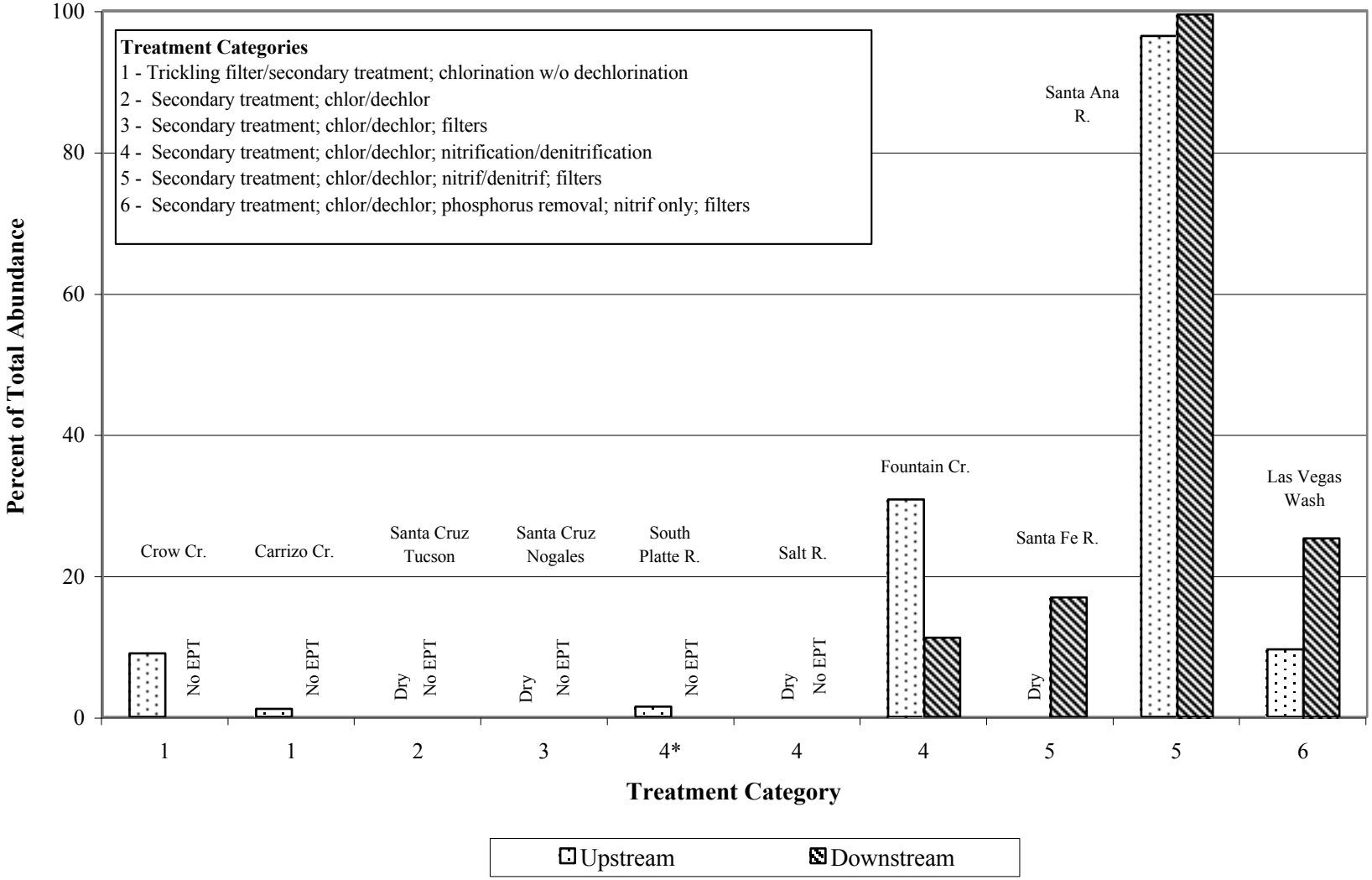
* Only portion of wastestream undergoes nitrification/denitrification.

Number of Macroinvertebrate Taxa Above and Below Effluent Discharge
(Taxa Identified to Family/Order/Class Level)
Figure D-43



Treatment Categories
1 - Trickling filter/secondary treatment; chlorination w/o dechlorination
2 - Secondary treatment; chlor/dechlor
3 - Secondary treatment; chlor/dechlor; filters
4 - Secondary treatment; chlor/dechlor; nitrification/denitrification
5 - Secondary treatment; chlor/dechlor; nitrif/denitrif; filters
6 - Secondary treatment; chlor/dechlor; phosphorus removal; nitrif only; filters

Number of Macroinvertebrate Taxa below Effluent Discharge vs. RBP Habitat Assessment Score
(Taxa Identified to Family/Order/Class Level)
Figure D-44



* Only portion of wastestream undergoes nitrification/denitrification.

Percentage of Mayflies, Stoneflies, and Caddisflies (EPT Taxa): Above and Below Effluent Discharges
Figure D-45

An evaluation of long-term changes in aquatic communities can be evaluated only in areas where aquatic species data are available over a sufficient period of time during which upgrades in wastewater treatment were implemented. The study areas available for this evaluation are limited to the South Platte River, Fountain Creek, Santa Ana River, and Santa Fe River. Evaluation will include a review of changes in wastewater treatment over the period in which aquatic biological data have been collected. These changes will be compared to taxanomic richness data (macroinvertebrate and fish, if available) and relative abundance of mayflies, stoneflies, and caddisflies (EPT taxa) data collected before and after the implementation of these facility changes. Increased relative abundance of EPT taxa is often associated with improved water quality.

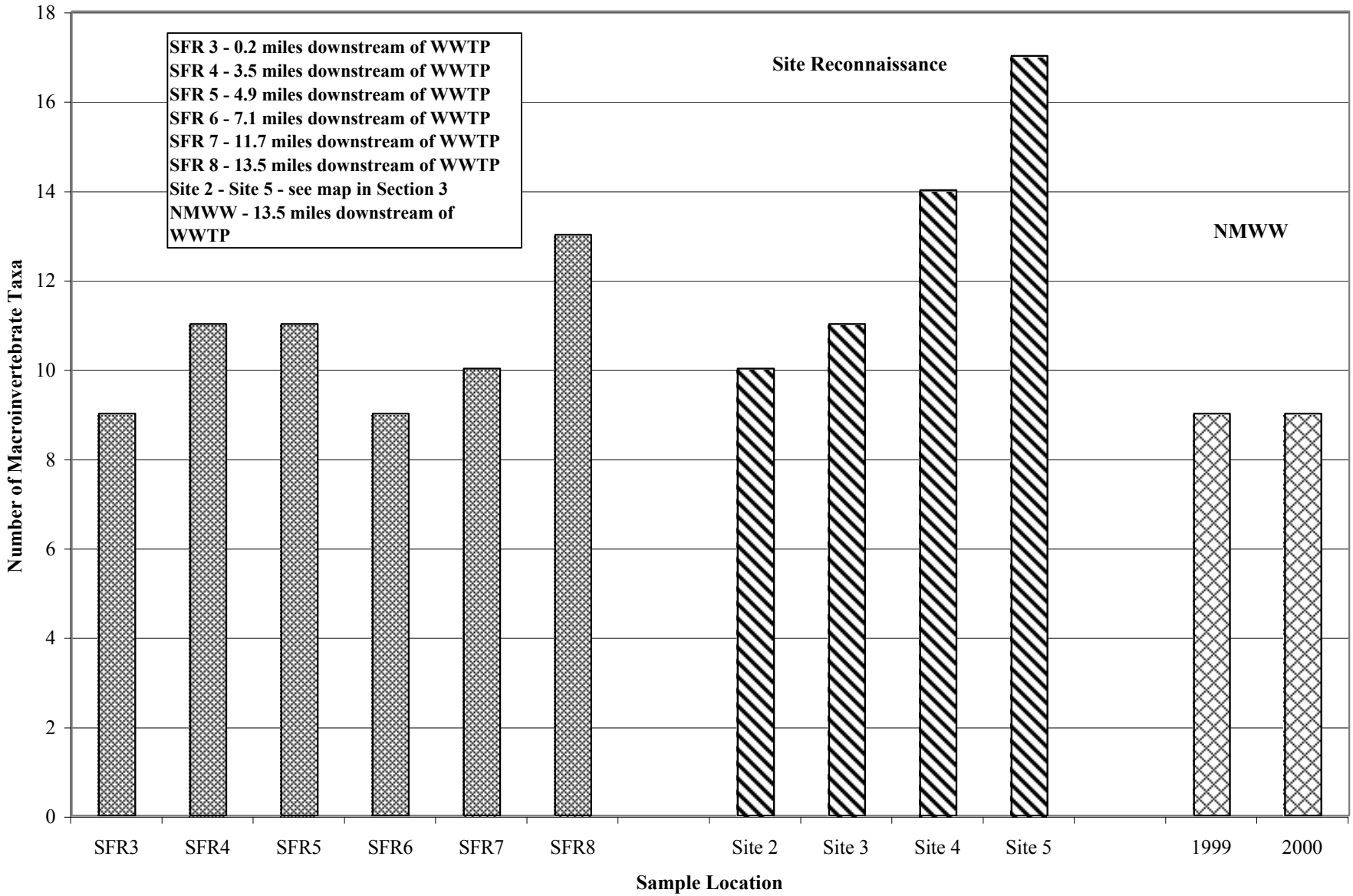
Santa Fe River. The Santa Fe River was sampled in 1994 and again during the site reconnaissance in May 2000. Between 1994 and 2000, the City of Santa Fe upgraded the WWTP to include filtration and replace chlorination with ultraviolet disinfection.

Macroinvertebrate taxanomic richness appears to have increased somewhat several miles downstream of the point of discharge since 1994 (**Figure D-46**). However, three factors limit our ability to state that taxanomic richness has definitely increased. First, the 1994 samples were collected at different times of the year: October 1994 versus May 2000. Second, the 1994 data had to be collapsed to the family/order/class level to allow comparison at the appropriate taxonomic level, and third, the 2000 sampling event was limited to field identification. However, the difference between 9 and 10 taxa at reconnaissance sites SFR6 and SFR7, respectively, collected in 1994 and the 14 and 17 taxa collected from similar locations in 2000 suggest that taxanomic richness very likely increased between 1994 and 2000.

EPT (mayflies, stoneflies, and caddisflies) abundance (primarily mayflies) increased considerably between sample dates at all reconnaissance sites, especially at sites near the point of discharge (**Figure D-47**). For example, in 1994 the site immediately below the discharge had no EPT taxa and at a site 2 miles downstream approximately 6 percent of the macroinvertebrates were EPT taxa. In contrast, in 2000 EPT taxa constituted about 18 percent of the macroinvertebrates at the site immediately below the discharge and the next site less than 2 miles below the discharge had about 90 percent EPT taxa.

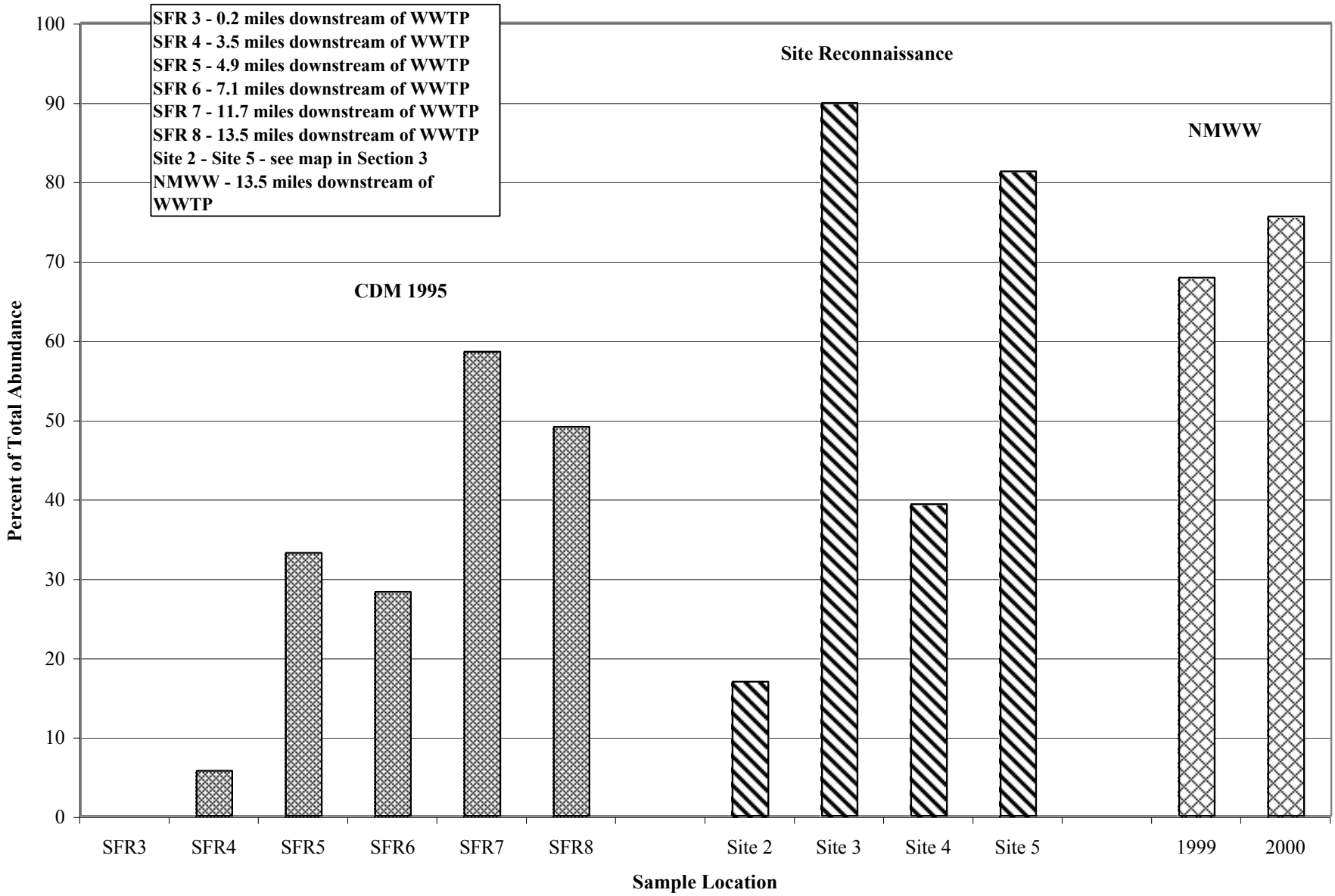
Fish taxanomic richness remained limited to two species below the effluent discharge. [Note: migration of fish species from the Rio Grande River is prevented by a river control structure.] However, fish abundance increased significantly between 1994 and 2000 especially near the point of discharge (**Figure D-48**).

The combined information gathered from changes in macroinvertebrate richness, EPT abundance and fish abundance suggest that changes in effluent quality at this facility have resulted in changes in the taxanomic richness and community composition.

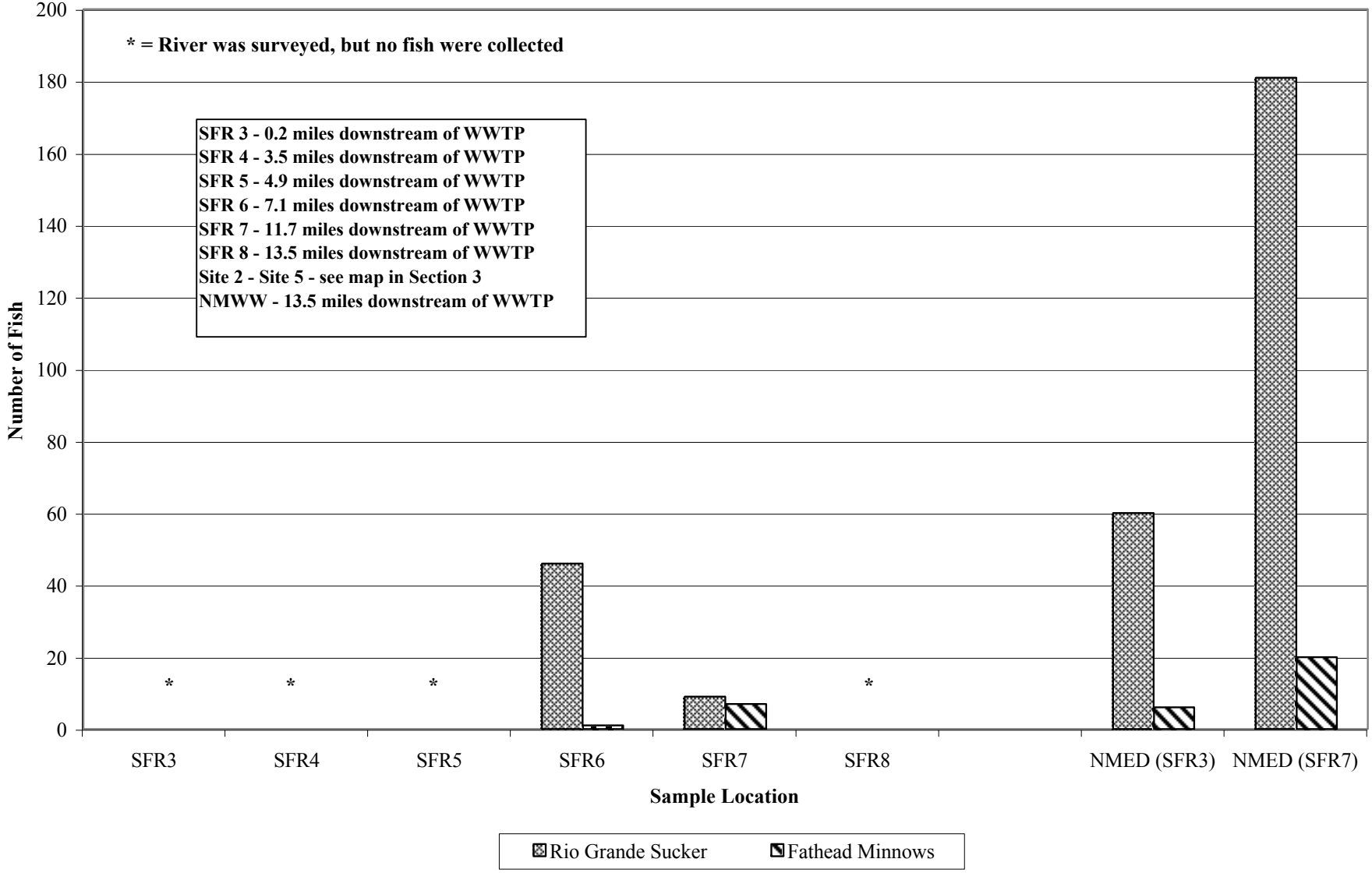


Santa Fe River - Number of Macroinvertebrate Taxa: 1994 Study (CDM 1995), Site Reconnaissance 2000, and NMWW (1999, 2000) (Taxa Identified at Family/Order Level)

Figure D-46



Santa Fe River - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): 1994 Study (CDM 1995), Site Reconnaissance 2000, and NMWW (1999, 2000)
Figure D-47



Santa Fe River - Fish Abundance in Samples Collected in 1994 (CDM 1995) and NMED (2000)
Figure D-48

Santa Ana River. The Santa Ana River was sampled quarterly in 1991 to support the development of a use attainability analysis. Sampling was conducted again in August 1998 to evaluate how the aquatic community had changed in the interim. Between 1991 and 1998 wastewater treatment improvements included combining the Colton and San Bernardino discharges to implement tertiary treatment on that effluent and implementation of nitrogen removal processes at the Riverside facility to meet strict nutrient standards.

To evaluate changes in taxanomic richness, the macroinvertebrate community results from August 1991 and August 1998 were compared (refer to **Figure D-31**). Taxanomic richness increased at five of seven sample sites in 1998. Overall, mean taxanomic richness of all seven sites was 10.9 in 1991 and 15.7 in 1998, suggesting that macroinvertebrate species richness increased.

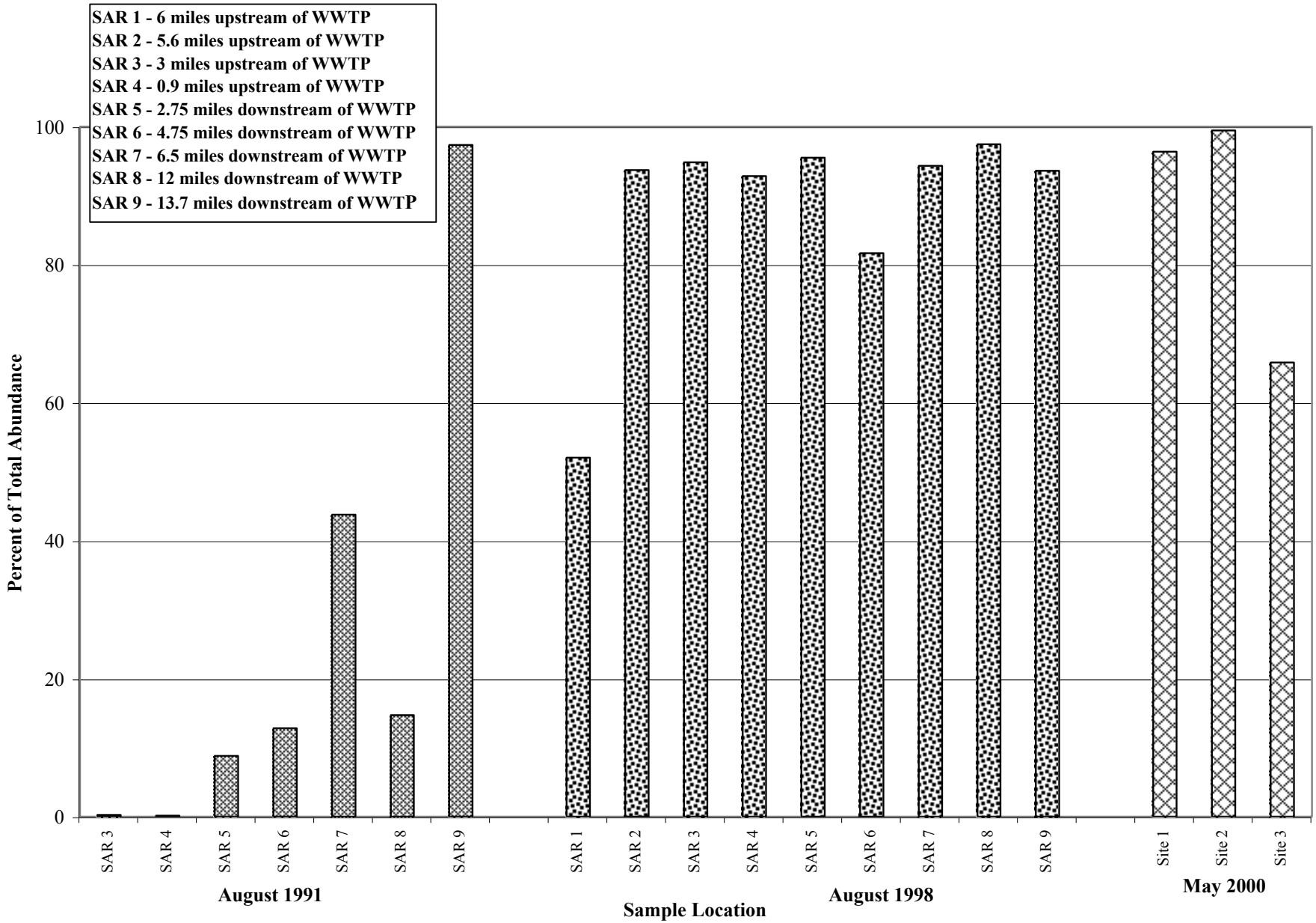
The relative abundance of EPT organisms (primarily mayflies) changed dramatically between August 1991 and August 1998 (**Figure D-49**). In 1991, EPT taxa constituted more than 20 percent of community abundance at only two of seven sites. In 1998 this same group constituted more than 80 percent of community abundance at all seven sites. The number of EPT taxa also increased substantially between 1991 and 1998 sampling events (**Figure D-50**), but these gains appear to have occurred both upstream and downstream of effluent discharge. Moreover, in 1991, prior to facility upgrades, the highest numbers of EPT taxa were found at the sites downstream of effluent discharges. Increases in taxa were distributed fairly even between mayfly and caddisfly taxa; no stonefly taxa have been observed in the Santa Ana River.

Fish taxanomic richness has increased at the upper five sites in the Santa Ana River since 1991 (**Figure D-51**). This increase has occurred both upstream and downstream of effluent discharges. At the most downstream sites, fish taxanomic richness has remained the same or declined slightly between 1991 and 1998. Similarly, fish abundance remained somewhat similar from one site to the next since 1991 (**Figure D-52**).

The combined information gathered from changes in macroinvertebrate taxanomic richness and EPT abundance suggest that changes in effluent quality at facilities discharging to the Santa Ana River have resulted in improvements in taxanomic richness.

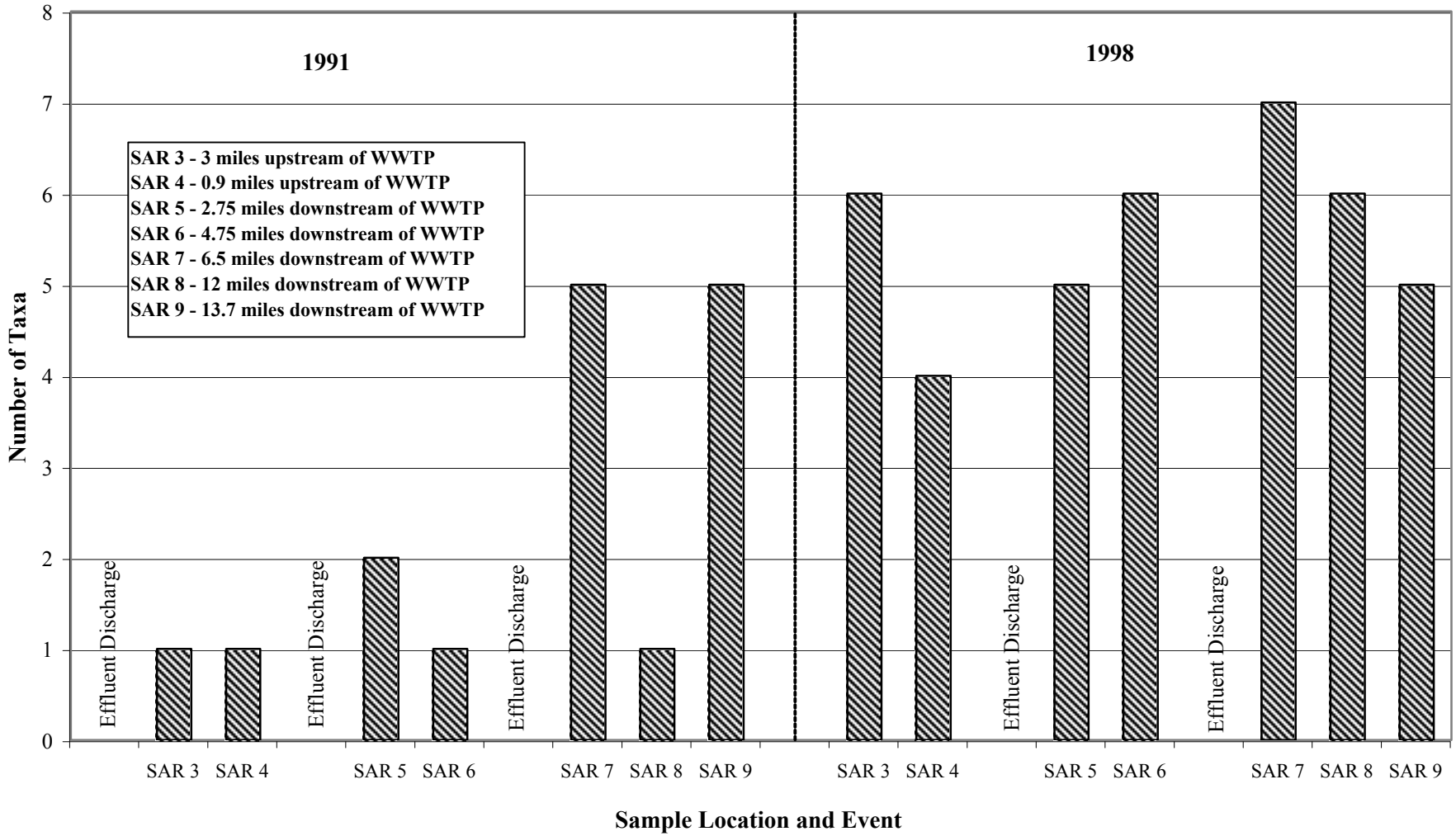
South Platte River. The South Platte River has been sampled annually for macroinvertebrates and fish since 1988. Macroinvertebrate data were available from 1988 to 1993; fish data were available from 1988 to 1998. The primary change in the wastewater treatment facility occurred in 1990/1991 with the addition of nitrification and denitrification at the north complex of the Denver Metro wastewater facility.

Macroinvertebrate taxanomic richness has remained essentially the same between 1988 and 1993 (refer to **Figure D-34**). The only suggestion of some change in taxa occurred at the downstream site closest to the effluent discharge where after 1990 taxanomic richness was considerably higher than during 1988 and 1989. However, because of the year-to-year variability evident in the data, it is unclear whether this change can be attributed to changes in treatment. Although a significant data gap exists between 1993 and 2000, it is interesting to note that taxonomic

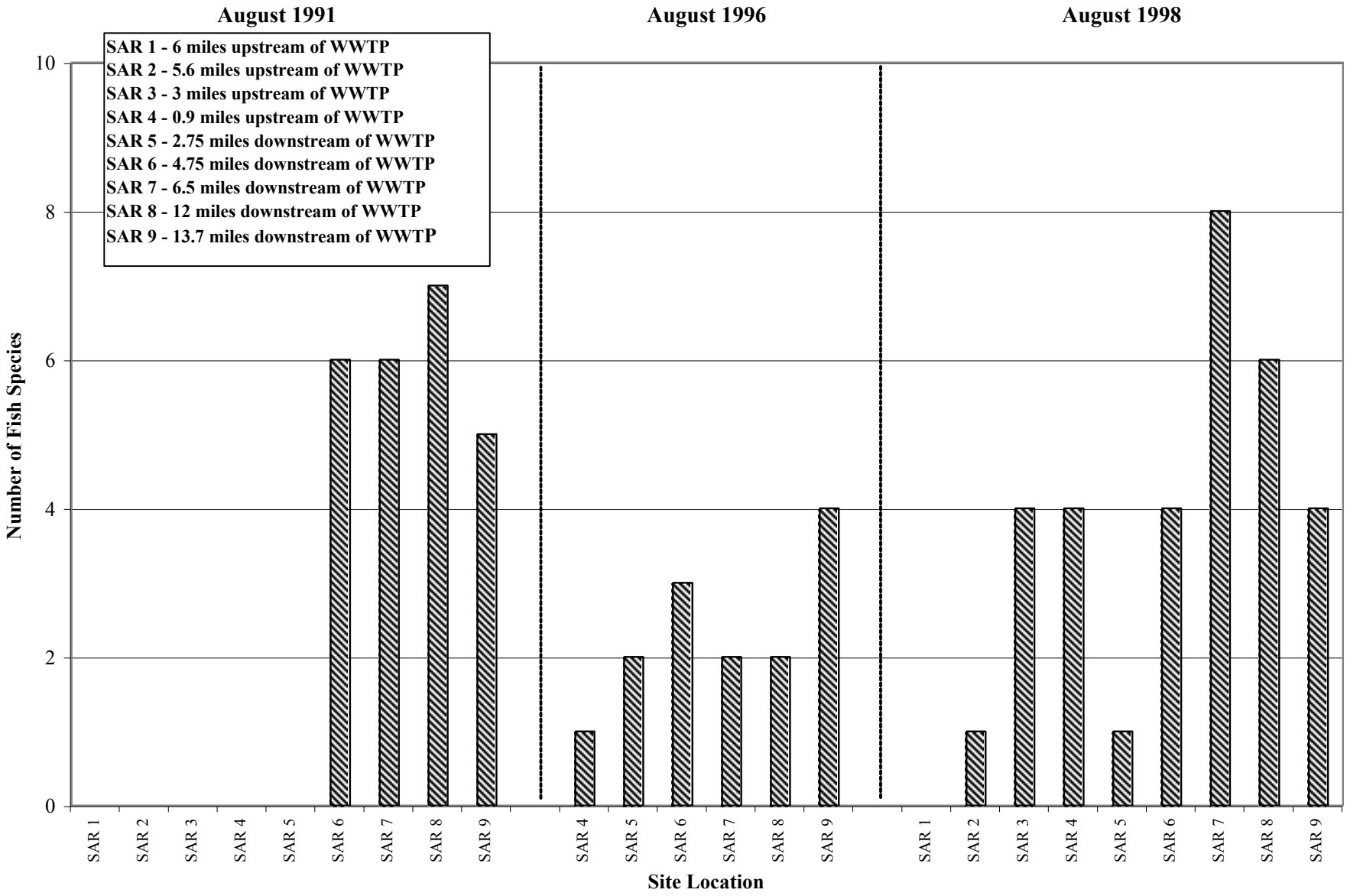


Santa Ana River - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): August 1991, August 1998 (CEC) vs. Site Reconnaissance 2000

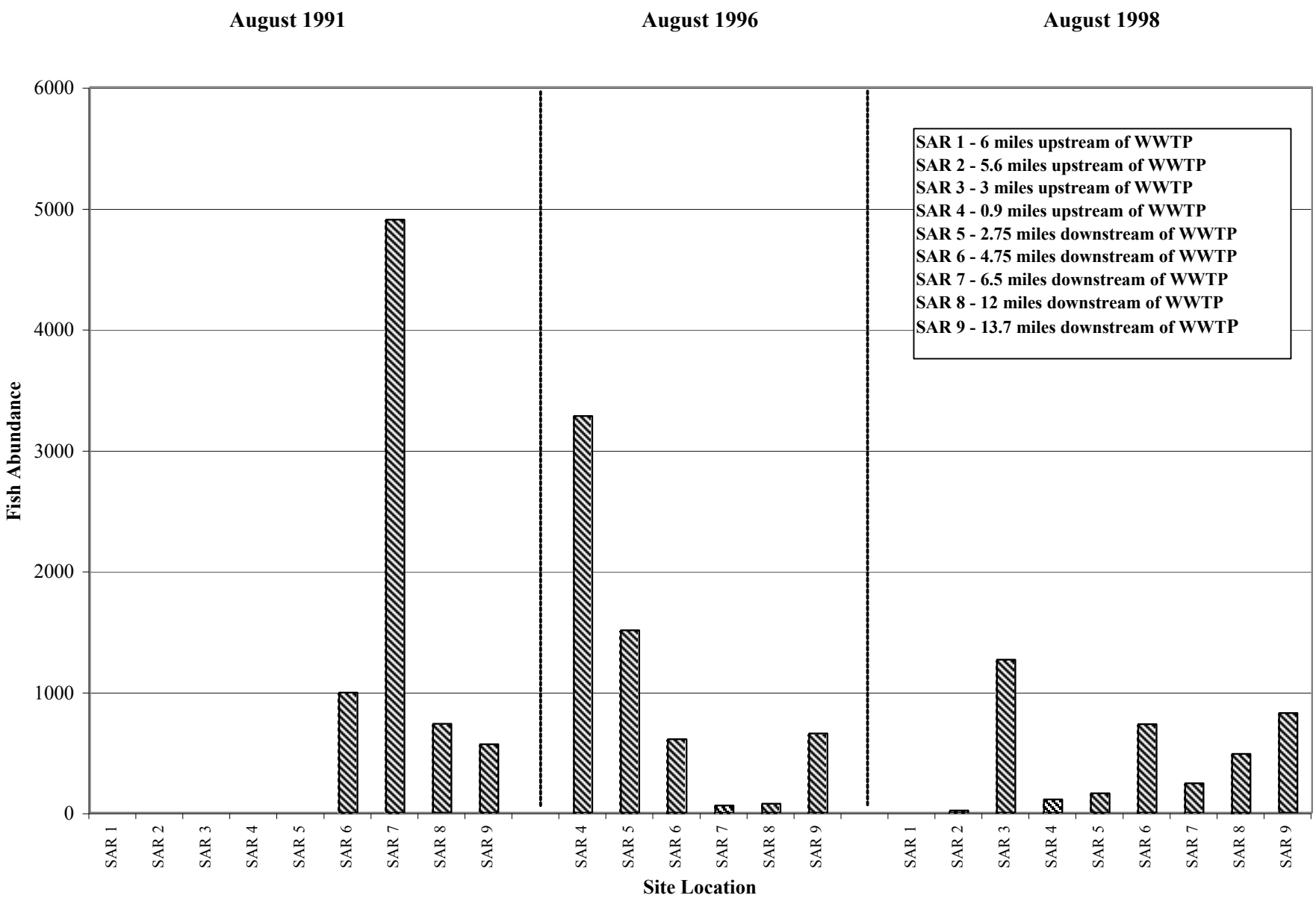
Figure D-49



Santa Ana River - Number of Mayfly, Stonefly, and Caddisfly Taxa (EPT Taxa):
1991 vs. 1998 (Taxa Identified to Lowest Practical Level)
Figure D-50



Santa Ana River - Changes in Fish Species Richness: 1991-1998
Figure D-51



Santa Ana River - Changes in Fish Abundance: 1991-1998
Figure D-52

richness at the family/order/class level was similar in 2000 to what was typically found from 1988 to 1993 (**Figure D-53**). The relative proportion of EPT taxa over these same years has also remained similar (**Figure D-54**).

Fish taxanomic richness is variable from year to year, but it appears that overall richness at three sites downstream of the effluent discharge has not changed since 1988, especially at 124th and 160th Avenues (refer to **Figure D-35**). Taxanomic richness at 88th Avenue, site nearest the discharge, appears to have generally declined after 1990 (refer to **Figure D-35**). However, there appears to have been a general decline in species richness at 31st Avenue, upstream of the discharge, from 1992 to 1997 and further downstream at 124th Avenue after 1991.

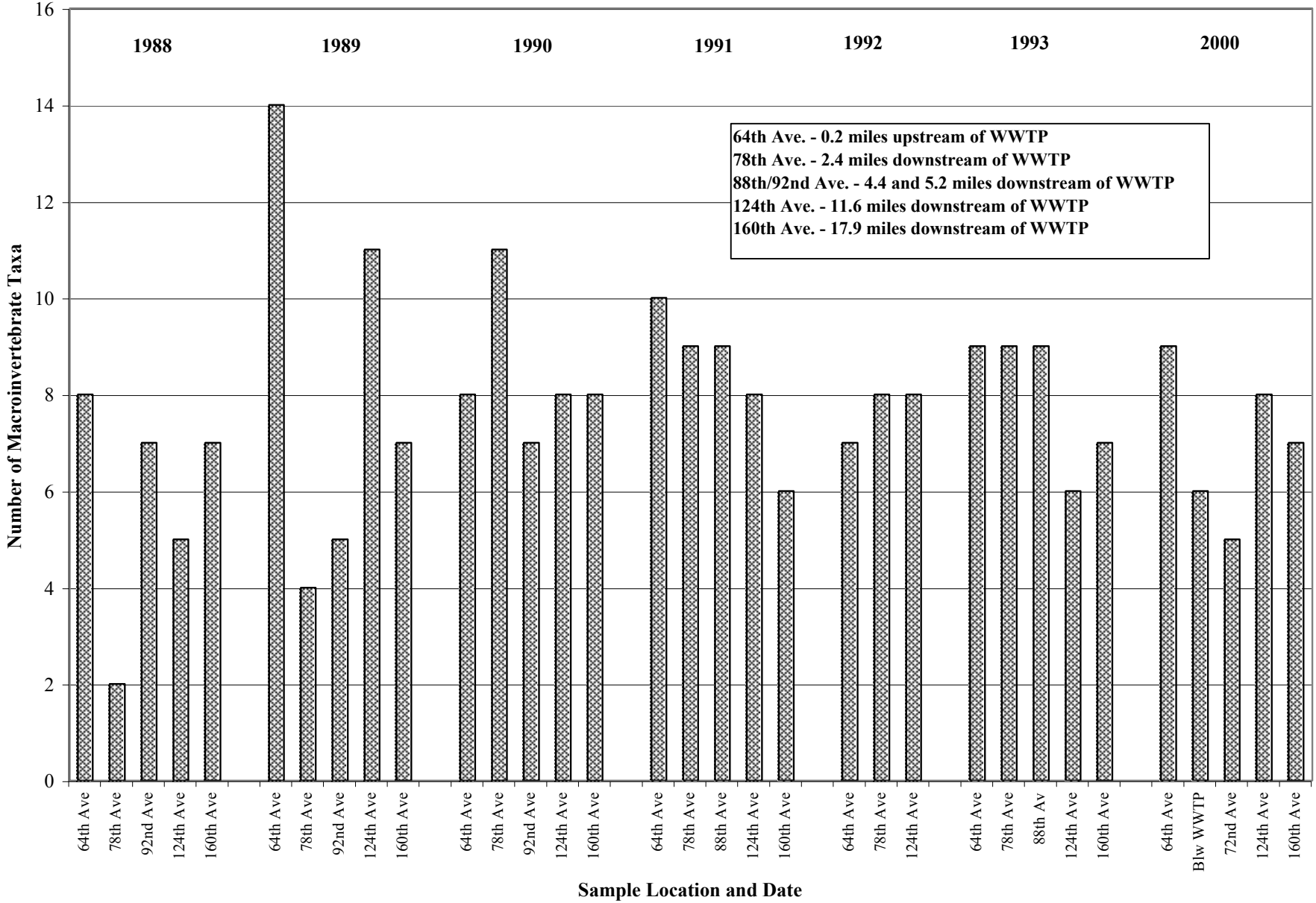
The results from the South Platte River would suggest that improvements in wastewater treatment have had no impact on macroinvertebrate or fish taxanomic richness.

Fountain Creek. Fountain Creek has been sampled for macroinvertebrates since 1980, providing a 20-year record of changes in taxanomic richness and macroinvertebrate community structure. Two key treatment upgrades have been implemented at the Las Vegas Avenue wastewater treatment facility since 1980: (1) dechlorination was added in the mid 1980s; and (2) nitrification and denitrification was added in 1996.

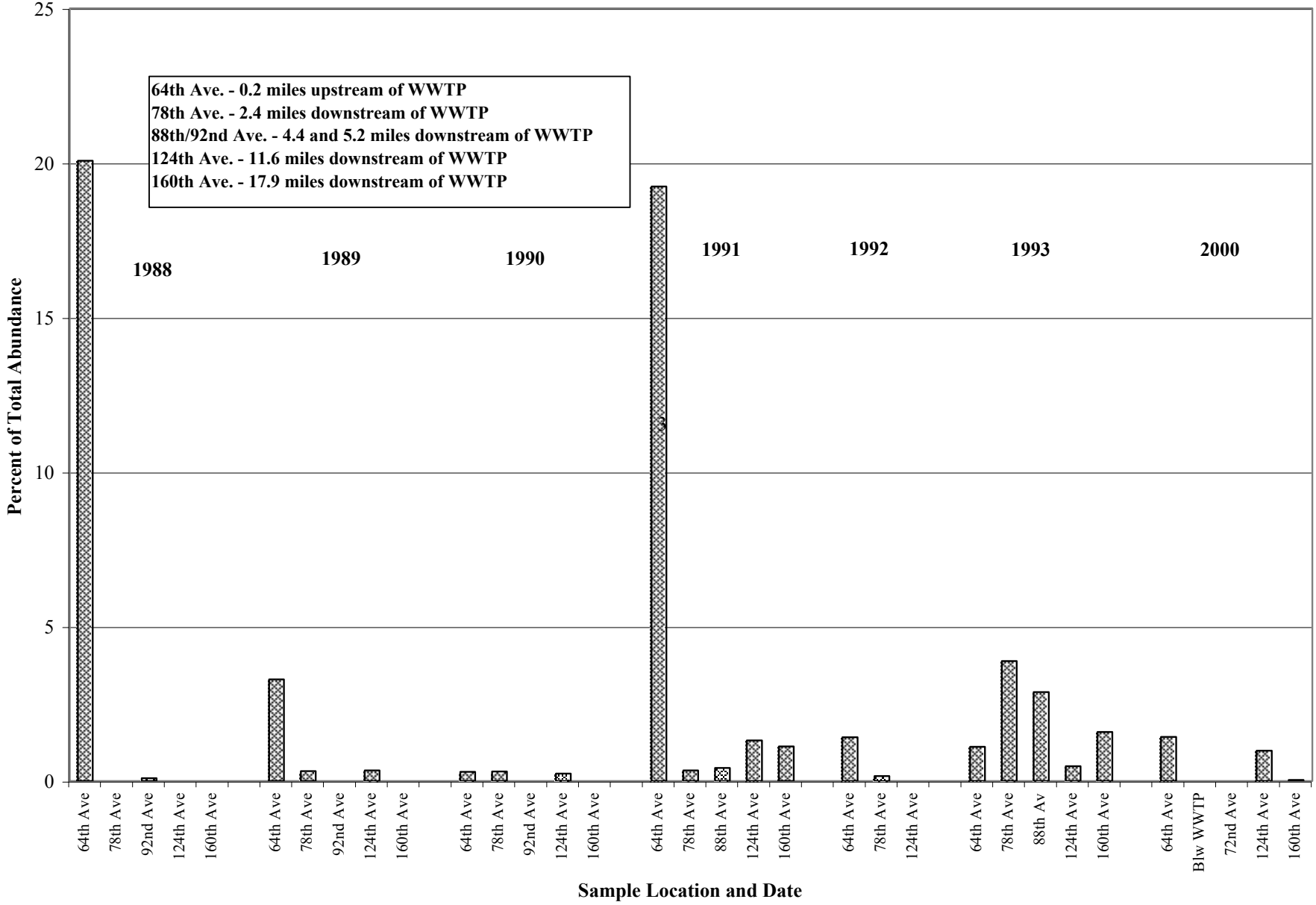
Taxanomic richness declined markedly in Fountain Creek between 1980 and 1989 (refer to **Figure D-36**). This change occurred both upstream and downstream of the effluent discharge, suggesting that stressors on the macroinvertebrate community other than or in addition to the Las Vegas Avenue wastewater treatment facility were impacting the aquatic community. Between 1989 and 1998/1999 taxanomic richness increased significantly to levels similar to what was typical downstream of the effluent discharge in 1980 (20 to 25 taxa) (refer to **Figure D-37**). However, the 1998/1999 results show little difference upstream and downstream of the discharge; moreover, taxanomic richness of the macroinvertebrate community upstream of the effluent discharge has not returned to levels found in 1980 when 30 to 35 taxa appear to have been the norm.

With the exception of 1999 results, EPT organisms have constituted less than 10 percent of the macroinvertebrates at all sites downstream of the effluent discharge since 1980 (**Figures D-55 and D-56**). This percentage contrasts with the high percentages of EPT organisms at upstream sites, especially in 1998 and 1999. The 1999 data show higher percentages at some sites, especially in fall 1999. Whether this increase will remain in subsequent years is unknown (however, we note that the site reconnaissance results from Spring 2000 were somewhat similar to the Spring 1999 results).

The number of EPT taxa varied from 2 to 4 taxa downstream of the effluent discharge and between 5, immediately upstream of the discharge, to 10 taxa farther upstream (**Figure D-57**). Few EPT taxa were observed at any site in 1989, but in 1998/1999, the mean number of EPT taxa varied from 8 to 11 upstream of the discharge to approximately 2 to almost 8 taxa downstream of the discharge. The highest number of EPT taxa downstream of the discharge was observed immediately downstream of the discharge and the number of EPT taxa declined with

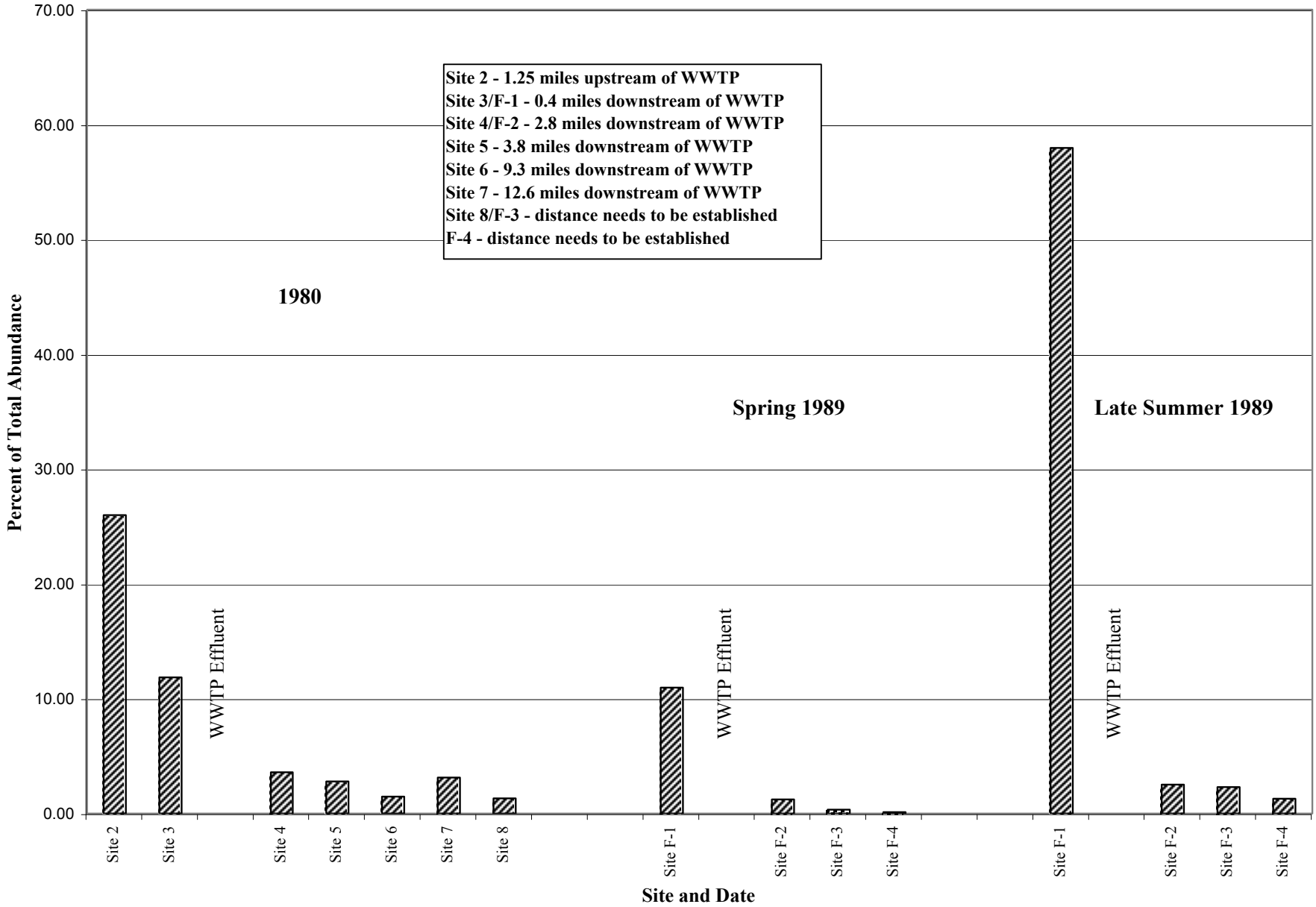


South Platte River - Number of Macroinvertebrate Taxa:
Denver Metro 1988-1993 Data vs. Site Reconnaissance 2000
(Taxa Identified to Family/Order Level)
Figure D-53



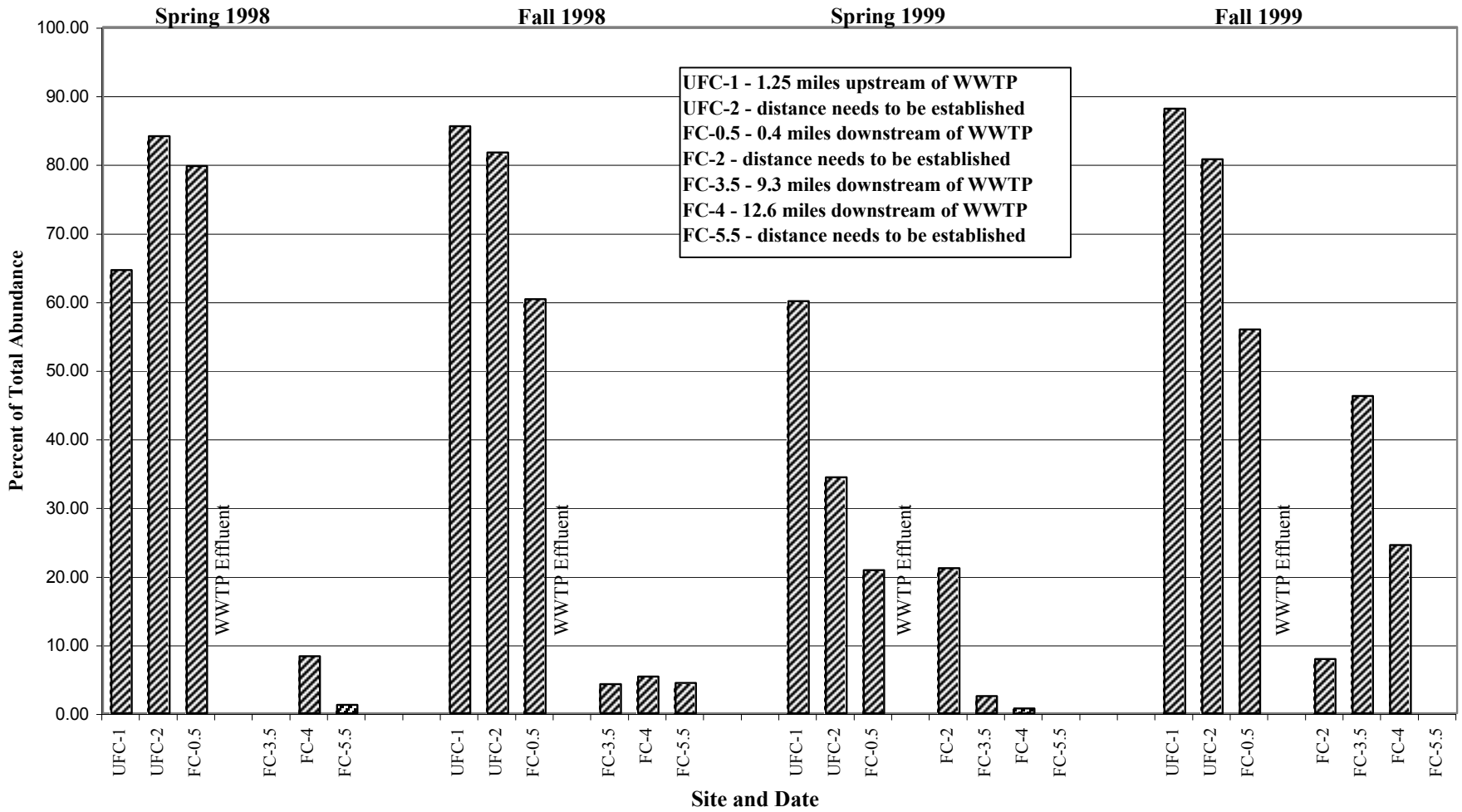
South Platte River - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): Denver Metro 1988-1993 Data vs. Site Reconnaissance 2000

Figure D-54



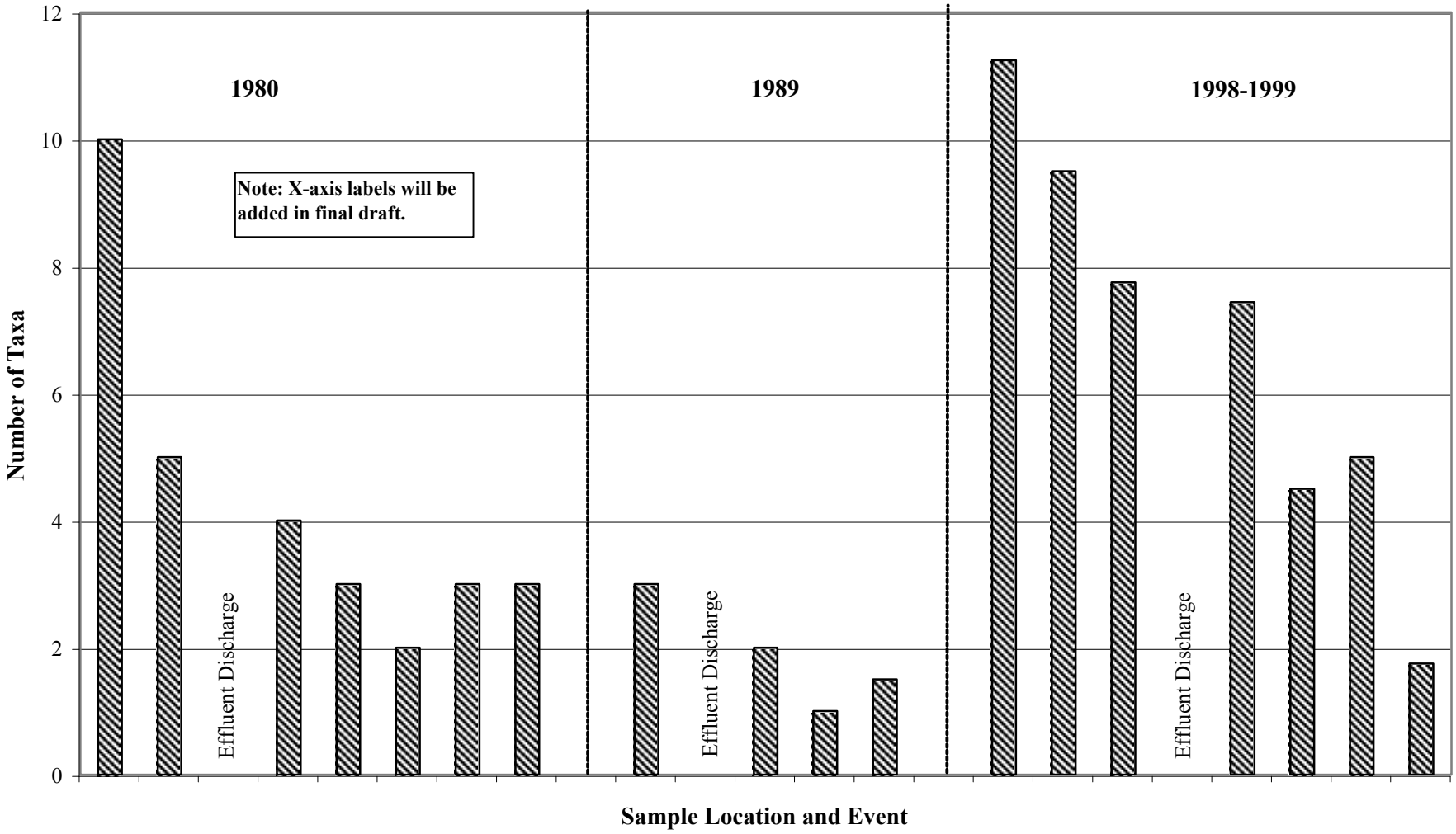
Fountain Creek - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa) 1980 vs. 1989 (Colorado Springs Wastewater Division,Taxa Identified to the Lowest Practical Level)

Figure D-55



Fountain Creek - Percent Mayflies, Stoneflies, and Caddisflies (EPT Taxa): 1998 vs. 1999 (Colorado Springs Wastewater Division, Taxa Identified to the Lowest Practical Level)

Figure D-56



Fountain Creek, CO - Mean Number of Mayfly, Stonefly, and Caddisfly Taxa (EPT Taxa), Upstream vs. Downstream of Effluent Discharge, 1980-1999 (Taxa Identified to Lowest Practical Level)

Figure D-57

increased distance downstream of the discharge. Although treatment levels increased over this 20-year period, the change in EPT taxa was from about three per downstream site in 1980 to about five per site in 1998/1999. Moreover, comparable gains in EPT taxa occurred upstream of the discharge suggesting that improvements in the aquatic community cannot be attributed solely to treatment upgrades.

More fish taxa were typically found at sample sites in 1989 than in 1980. This increased fish community richness contrasts with the decreased macroinvertebrate richness observed during the same period. This improvement in fish taxanomic richness between 1980 and 1989 was attributed to improvements in effluent treatment (addition of dechlorination) and improved instream habitat (stream was formerly diverted into a side channel) (CSWD 1989).

Summary. The results from these four study areas provide mixed results for evaluation of this issue. The aquatic community of the Santa Fe River appears to have increased taxanomic richness, which could be attributed to improvements in wastewater treatment. EPT taxa, which would be expected to respond to improved effluent quality, have increased in proportion as well, supporting the suggestion that improvements in wastewater treatment have impacted the aquatic community.

Some improvements have been observed in the aquatic communities of the Santa Ana River, but some of these improvements have occurred both upstream and downstream of effluent discharges. This mixed result suggests that factors other than wastewater treatment improvements have influenced changes in the aquatic community.

Macroinvertebrate and fish taxanomic richness in the South Platte River, while variable from year to year, appears to have changed little over a 10-year period. EPT taxa have also changed little in terms of their relative abundance in the aquatic community. These results would suggest that improvements in wastewater treatment have had little impact on the aquatic community.

Between 1980 and 1999, invertebrate taxanomic richness declined and then improved in Fountain Creek. This might suggest that increases in wastewater treatment were the cause, but it is notable that taxanomic richness at sites upstream of the effluent discharge also declined and improved during the same period of time. Moreover, there has been little difference observed in taxanomic richness above and below the discharge since 1989, suggesting that stressors other than the effluent are acting upon Fountain Creek. The proportion of EPT taxa is typically much higher upstream of the effluent discharge, especially in 1998 and 1999. At the same time, the proportion of EPT taxa has changed little downstream of the effluent discharge. These results coupled with the taxanomic richness data make interpretation difficult for the purposes of this issue. It would appear that improvements in wastewater treatment have had some positive impact on the macroinvertebrate community, but clearly because changes also have occurred upstream of the discharge, impacts from stressors other than effluent have also acted on the aquatic community. In contrast to the invertebrate community, data collected during the 1980s suggest that fish taxanomic richness did improve as a result of wastewater treatment improvements.

1.2.8 Adequacy of Available Habitat Characterization Protocols

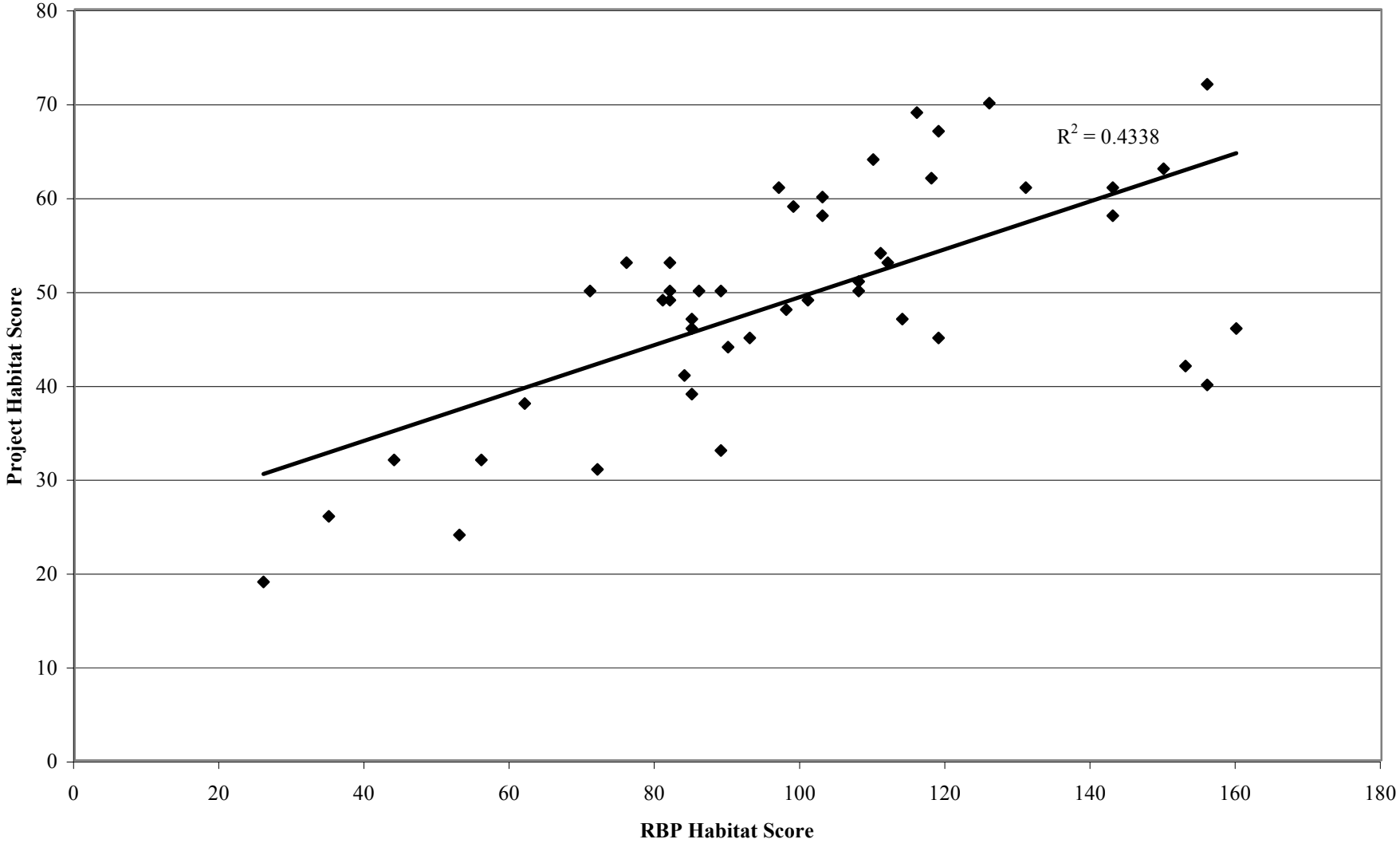
Instream bioassessment methods are based on the principle that selected habitat or biological parameters evaluated in the field will be compared to expected, optimum conditions. Based on those comparisons, the site of interest is rated as to whether or not it provides optimum conditions. The EPA's Rapid Bioassessment Protocols (RBPs) are an example of this type of procedure.

However, assessment of habitat in streams of the arid Southwest may not be accurately portrayed by direct application of the RBP procedures. The original RBP habitat procedures were actually based on habitat assessment techniques developed in Wisconsin and Idaho, and especially trout streams in these states. Processes that create optimum habitat in those regions may not be as important, or even relevant, in the arid southwest. The RBP habitat assessment field sheets have condition categories for 10 habitat parameters, rating these parameters as poor, marginal, suboptimal, and optimal as defined in the current RBP protocols. However, the ephemeral or intermittent nature of many southwestern streams may lead to poor ratings for an entire study area, even though this is a naturally occurring condition and within this condition there exists a range of habitat quality not discernable with current RBP ratings. It may be more appropriate to use a habitat assessment procedure developed specifically for the southwestern United States.

As an example, one of the RBP habitat parameters for low gradient streams is designated "epifaunal substrate/available cover." Its optimal condition is assumed to contain large areas of substrate suitable for benthic invertebrate colonization and fish cover, such as woody snags, submerged logs, and undercut banks. However, many low-gradient southwestern streams do not naturally have the appropriate riparian vegetation to contribute material to form woody snags and submerged logs. And although undercut banks may have formed during high flow events, such as spring runoff or flashy runoff from precipitation events, they do not always function as fish cover during the high percentage of the year when flows are low or absent. A different rating scale for this habitat parameter, scaled toward the more limited potential for many low-gradient southwestern streams, would lead to a more realistic picture of what the optimum condition is for the substrate/cover parameter.

To evaluate the comparability of the results from the RBP and Project Habitat Assessment Methods, the habitat scores from each method were subjected to regression analysis to determine the relationship between the resulting scores. If a high correlation coefficient was obtained from this analysis, it could assumed that the two methods provided similar results. If a low correlation coefficient resulted, then it was likely that the two methods provided different habitat assessments. Further analysis would then be warranted to evaluate which method provided the better habitat assessment.

A total of 47 habitat assessment scores were generated during the site reconnaissance. Linear regression results showed a significant positive relationship ($r^2 = 0.4338$) between the scores obtained from the two methods (**Figure D-58**). However, it must be noted that while the relationship using all study areas was positively significant, one study area showed no relationship (Crow Creek) and one study area had a strong but insignificant negative relationship (Carrizo Creek). Removing Carrizo Creek data from the data analysis results in a much greater

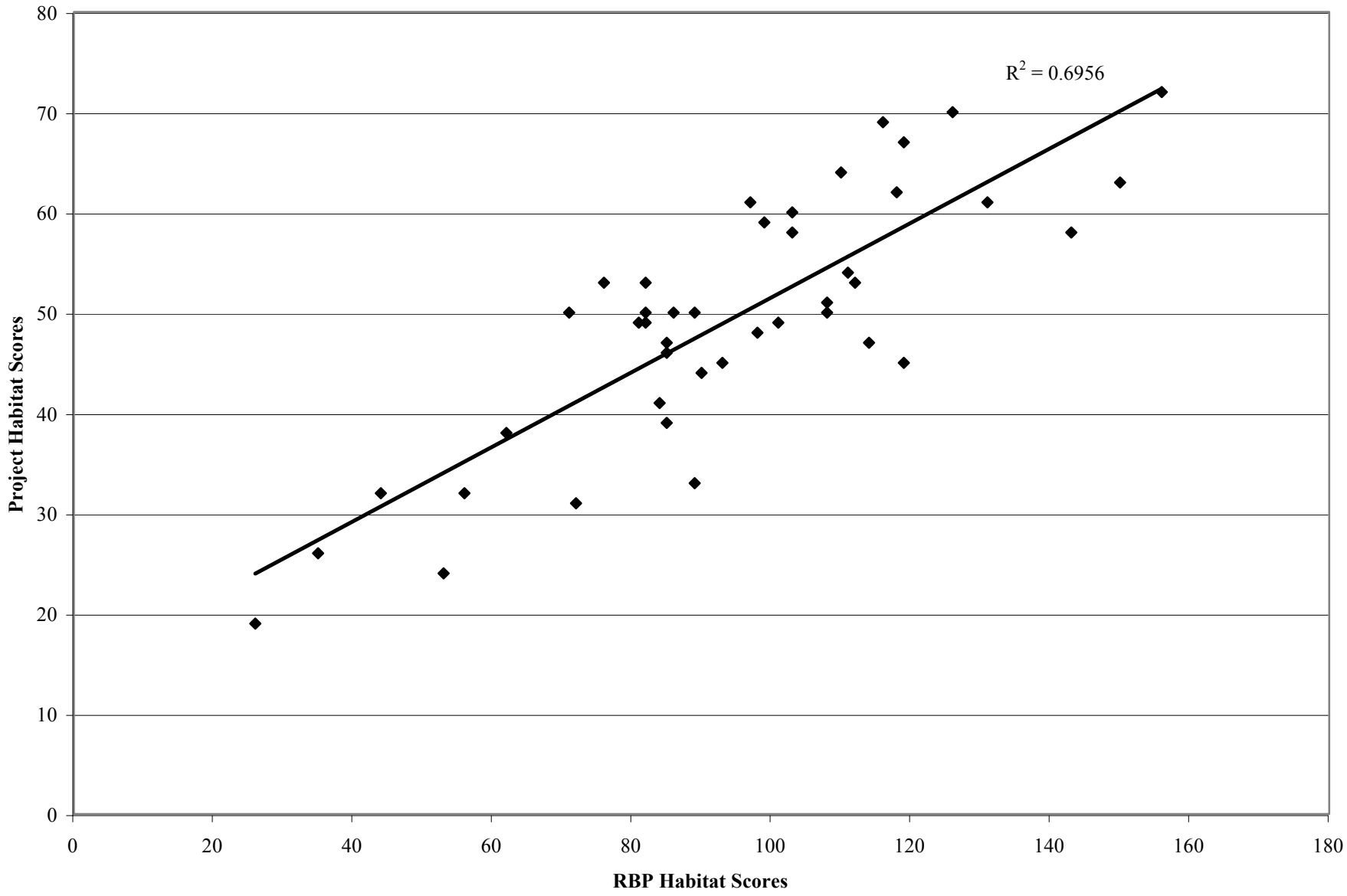


Linear Regression - RBP vs. Project Habitat Assessment Scores
Figure D-58

significant relationship between the two scoring methods (**Figure D-59**; $r^2 = 0.6956$). Although Carrizo Creek and Crow Creek did not follow the pattern observed at the other eight study areas, there is insufficient reason at this time to argue that the EPA's RBP Habitat Assessment Method is inappropriate as a habitat assessment tool for effluent-dependent waters in the arid West. It should also be noted that EPA recommends gathering additional habitat data to supplement the RBP Habitat Assessment Method. Collecting these supplemental data to support the RBP habitat scores should provide additional confidence with the use of the EPA's RBP method in effluent-dependent waters.

1.2.8.1 Terrestrial Habitat Characterization Protocols

There are many methods of measuring riparian habitat parameters such as density, cover, and species diversity. However, most of the methods available require intensive effort and hundreds of hours to obtain statistically reliable information (Anderson and Ohmart 1984). The project team did not find any high-quality, rapid methods for assessing riparian or other terrestrial vegetation. Some methods for collecting very general data have been developed and are included in some manuals on field procedures (EPA 1998; Leonard et al. 1992). For this study, the project team combined features EPA's 1998 field data sheet with features found in other manuals. The project team also developed some of our own field data categories that were used during site reconnaissance visits to the 10 dischargers that constitute our study group.



Linear Regression - RBP vs. Project Habitat Assessment Scores (Carrizo Creek Removed)
Figure D-59