3.0 STREAM ECOSYSTEMS

Stream ecosystem structure and function is collectively defined by a number of physical, chemical, and biological attributes. These attributes result in various classifications based on different schemes. Hydrologically, stream ecosystems may be characterized as perennial, intermittent, or ephemeral depending upon the permanence of natural baseflow. Or, with regards to temperature effects on fish communities, streams may be classified as warmwater or coldwater. Another often-defined stream type in the arid West is the effluent-dependent water, a flowing surface water created by the discharge of effluent to an otherwise dry or intermittent riverbed. This chapter first describes the effluent-dependent water ecosystem in the context of natural stream ecosystems, and second, provides an overview of effluent-dependent water characteristics identified by the Habitat Characterization Study.

3.1 CONCEPTUAL MODEL OF AN UNDISTURBED STREAM ECOSYSTEM

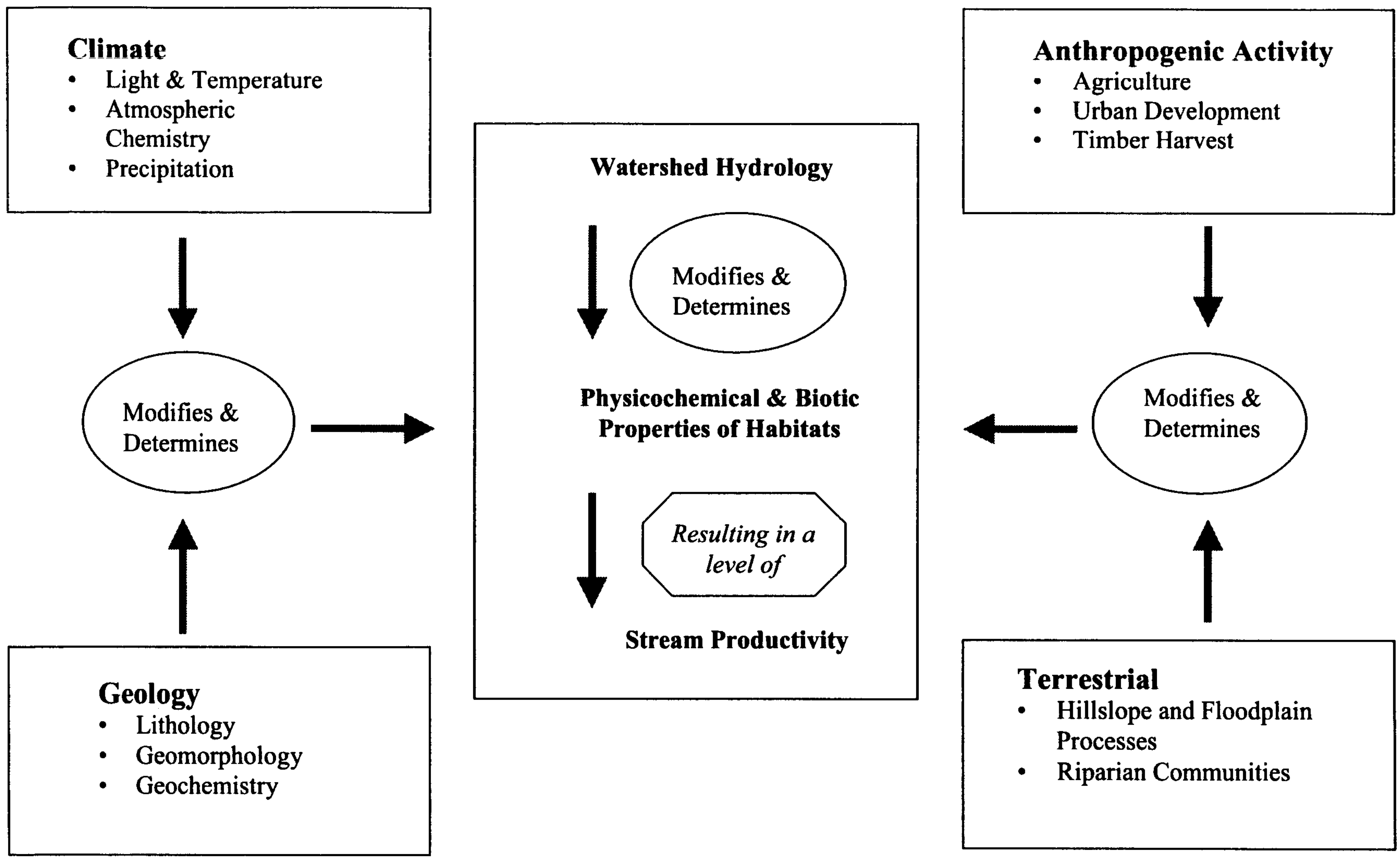
The movement of water through a stream ecosystem results in the establishment of pathways for the movement of energy and nutrients through the system. For natural streams, these pathways have been variously described in the stream ecology literature (see for example Giller and Malmqvist 1998; Resh and Rosenberg 1984). Of particular interest to the Habitat Characterization Study was how effluent-dependent waters as a distinct type of stream ecosystem compare to natural stream ecosystems. The following sections provide an overview of this comparison as well as a summary of observations regarding the physical, chemical, and biological characteristics of the 10 effluent-dependent waters evaluated for this study.

3.1.1 Four-Dimensional Model of Stream Ecosystems

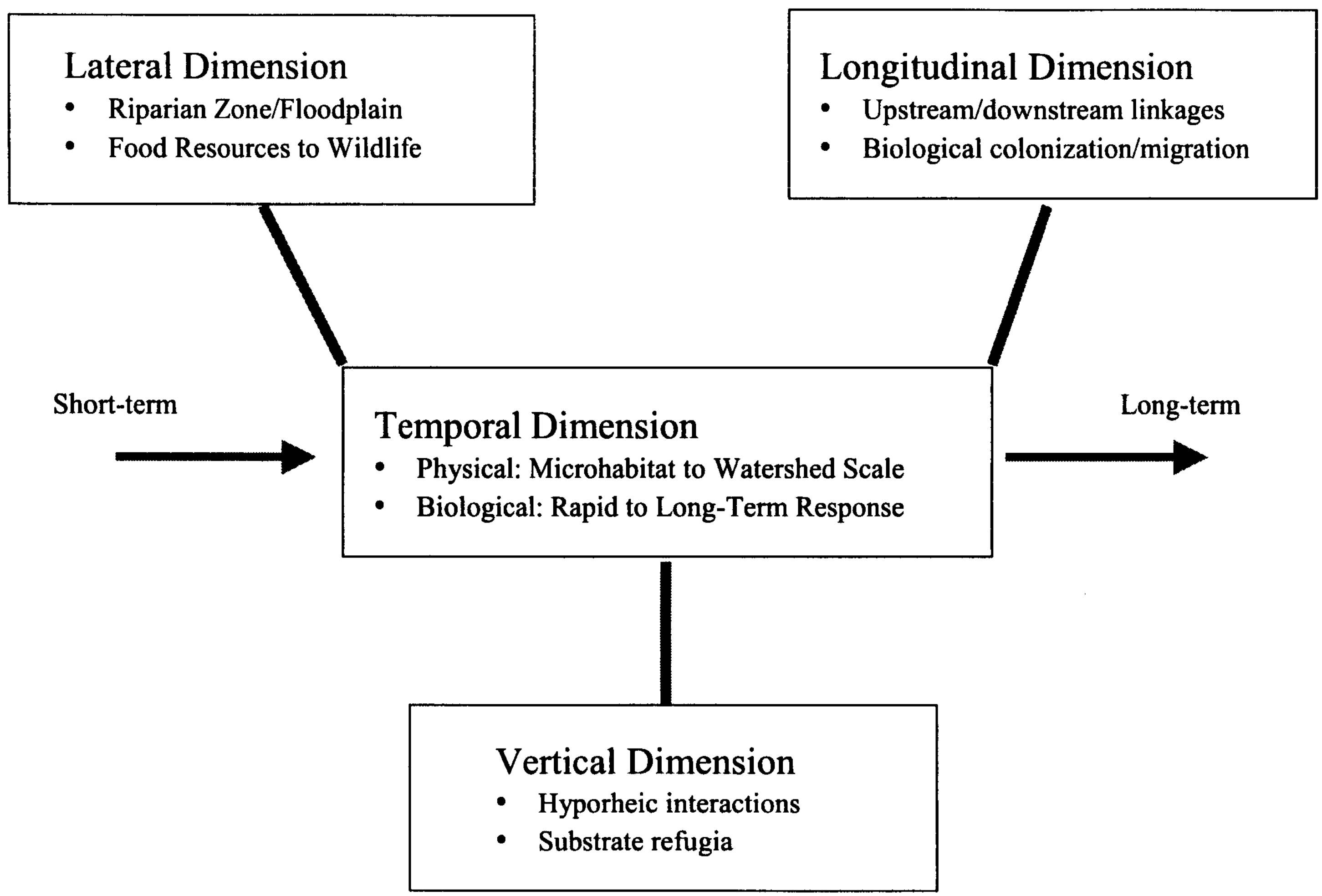
Conceptual models of stream ecosystems have been summarized in the literature in varying ways (for example, **Figure 3-1**; Gregory et al. 1991; Resh and Rosenberg 1984). In contrast, conceptual models for effluent-dependent waters, especially those created in drainages that have only intermittent or ephemeral streamflow, have not been developed. The four-dimensional template published by Ward (1989) provides a particularly useful approach upon which to describe characteristics of both natural streams (i.e., naturally ephemeral, intermittent, and perennial waters), and effluent-dependent waters (**Figure 3-2**). This template becomes particularly useful because it includes both spatial and temporal components, elements with important meaning in an arid environment. The following section provides an overview of the characteristics of a typical natural stream ecosystem, summarizes observed differences in stream ecosystems created by the discharge of wastewater effluent, and provides a discussion of the expected implications of these changes, especially in terms of biological communities.

3.1.1.1 Longitudinal Dimension

Vannote et al. (1980) observed that most streams gradually changed in hydraulic properties in a downstream direction and that, within individual reaches, these gradual changes could be related to an equivalent change in ecological communities. In this way, the river formed a continuum of biological change directly dependent upon downstream physical changes (Osterkamp et al. 2001).



Biotic and Abiotic Factors Affecting Stream Productivity.
Adapted from Resh et al. (1988).
Figure 3-1



Four-Dimensional Nature of Stream and River Ecosystems. Adapted from Ward (1989).
Figure 3-2

A geological framework should be presumed to influence longitudinal patterns. This framework includes the lithology of the valley, structural features (folds and faults), and a regional groundwater system. In most cases, a stream channel uncontrolled by anthropogenic structures or external dynamics will adopt a concave-upward longitudinal profile (Langbein 1964), meaning that the slope of the channel will decrease in the downstream direction. The most common longitudinal change will be a transition from bedrock-controlled channels to alluvial, or sediment-filled, valleys. This transition is most commonly observed in the arid West when headwater or mountain streams cross a range front and enter a valley.

Non-physical longitudinal patterns also occur in stream ecosystems. Examples include: (1) longitudinal patterns in nutrient dynamics as nutrients are taken up, processed, excreted, or leached (Newbold et al. 1981, 1982a, 1982b); (2) fish migration along a river continuum (e.g., see for example Lowe-McConnell 1987); and (3) the upstream-downstream linkage that exists between the downstream drift of immature aquatic insects, often as larvae, and the dispersal of adult insects upstream following emergence (see for example Müller 1954).

3.1.1.2 Lateral Dimension

The lateral dimension consists of the relationship between the stream channel and the adjacent landscape for some distance perpendicular to the channel. This relationship occurs at three levels; subsurface, surface, and above surface. The size of this lateral dimension is dependent on a number of factors, most significantly watershed area (i.e., with increasing watershed area, the width of the floodplain typically increases). Floodplain characteristics (e.g., vegetation, biological productivity, and morphology) are a reflection of the nature of the flood regime in a river basin. The relationship between floodplain characteristics and flood regime is most pronounced in low-gradient rivers where high water events can spread out onto the floodplain.

The predictability of flood events in the floodplain is somewhat related to the weather patterns of the region in which the river is found. Where this flood event still occurs with relative predictability (i.e., where the cycle has not been modified or lost because of the presence of a dam or diversion), an exchange of nutrients can occur between the river channel and riparian/floodplain system (Junk et al. 1989). In addition, active or passive movements of organisms occur between the channel and the adjacent riparian and floodplain system (Junk et al. 1989).

The form of any channel, braided versus incised, can alter the distribution of both nutrients and soil moisture across the floodplain. This can have an important impact on the individual species and structure of the floodplain plant community. Studies have shown that the selection of cottonwoods over salt cedar, for example, and other habitat complexity, can be driven by the distribution of flow across the floodplain (Asplund and Gooch 1988; Cuomo 1992; Howe and Knopf 1991; Levine and Stromberg 2000; Osterkamp et al. in press; Szaro 1990;).

As flood crests pass downstream, water that infiltrates from the banks and inundated surfaces can remain temporarily as a kind of perched aquifer. Eventually, this water can drain back to the river under gravity, percolate downward to the water table, or be taken up by plants as soil moisture. The ability of the adjacent soil to store and transmit water is a function of its hydrogeologic function, which itself is related to the permeability of the channel substrate and adjacent overbank deposits. The lateral distribution of surface or subsurface water serves to create soil moisture and irrigate plant life.

The subsurface saturated region under the floodplain, termed the parafluvial zone, has been found to have potentially important implications to the functioning of the river ecosystem (Boulton et al. 1998). For example, Stanford and Ward (1988) have demonstrated the lateral migration of riverine aquatic insects up to several miles from the main channel in a glacial Montana river. The presence of large cobble substrates with substantial interstitial space across the wide floodplain assisted the lateral spread of riverine water and associated aquatic organisms. While the lateral migration found in this Montana river may not be reflective of typical lateral migration distances, it illustrates that what constitutes the river ecosystem can include much more than what is seen at the surface.

Finally, another lateral path associated with stream ecosystems is the transfer of energy from the stream to the terrestrial community adjacent to the stream. While some of this transfer can occur in a longitudinal manner, the majority of transfer is lateral. The transfer of energy along this dimension is in the form of insect biomass. As aquatic insects complete the immature portion of their life cycles, many of these insects emerge from the stream as winged adults to mate and lay eggs in or adjacent to the river. Jackson and Fisher (1986) showed that less than 4 percent of the biomass of emerging insects returned to the stream (in the form of dead insects or newly lain eggs). The remaining 96 percent of the biomass resulted in a net export of 22.4 grams/square meter/year of biomass or food to terrestrial insectivores (e.g., birds and bats).

3.1.1.3 Vertical Dimension

The vertical dimension primarily describes the interaction between river waters and any underlying groundwater. The transition region between the two water layers has been termed the hyporheic zone (see review by Boulton et al. 1998). The extent or size of the hyporheic zone is temporally dynamic and dependent on factors such as porosity and relative volume of water recharging groundwater from the surface or surface water from the aquifer. It is assumed that at least a minimal hyporheic zone exists in all rivers from headwaters to confluence. However, the size and shape of this zone varies along longitudinal and lateral dimensions as a function of the porosity and permeability of the floodplain deposits.

The influence that the hyporheic zone has on biological communities at the surface is complex and depends on the interaction of several factors, especially flow, water chemistry, and substrate characteristics. For example, the degree to which upwelling subsurface water is nutrient-rich or nutrient-poor may vary in response to changes in flow, (e.g., flooding, drying, and seasonal flow patterns) (Dent et al. 2001; Jones et al. 1995; Stanley and Boulton 1995). Sediments that are coarse and well oxygenated favor nitrification processes and are more likely to be nutrient sources (Hendricks and White 1995; Jones et al. 1995; Valett et al. 1994). However, sediments that are fine and organic-rich are more likely to be nutrient sinks because the environment favors denitrification (Duff and Triska 1990). Where upwelling water is nutrient-rich, algae and associated invertebrates may respond positively. In contrast, if the upwelling water is nutrient-poor, a different biological response may be observed.

In addition to the role that surface/subsurface interactions have on nutrient characteristics in streams, the hyporheic zones influences the stream ecosystem in other ways. For example, in streams with coarse substrates, the hyporheic zone can provide the aquatic fauna with a refugium during high water events (Stanford and Ward 1993).

3.1.1.4 Temporal Dimension

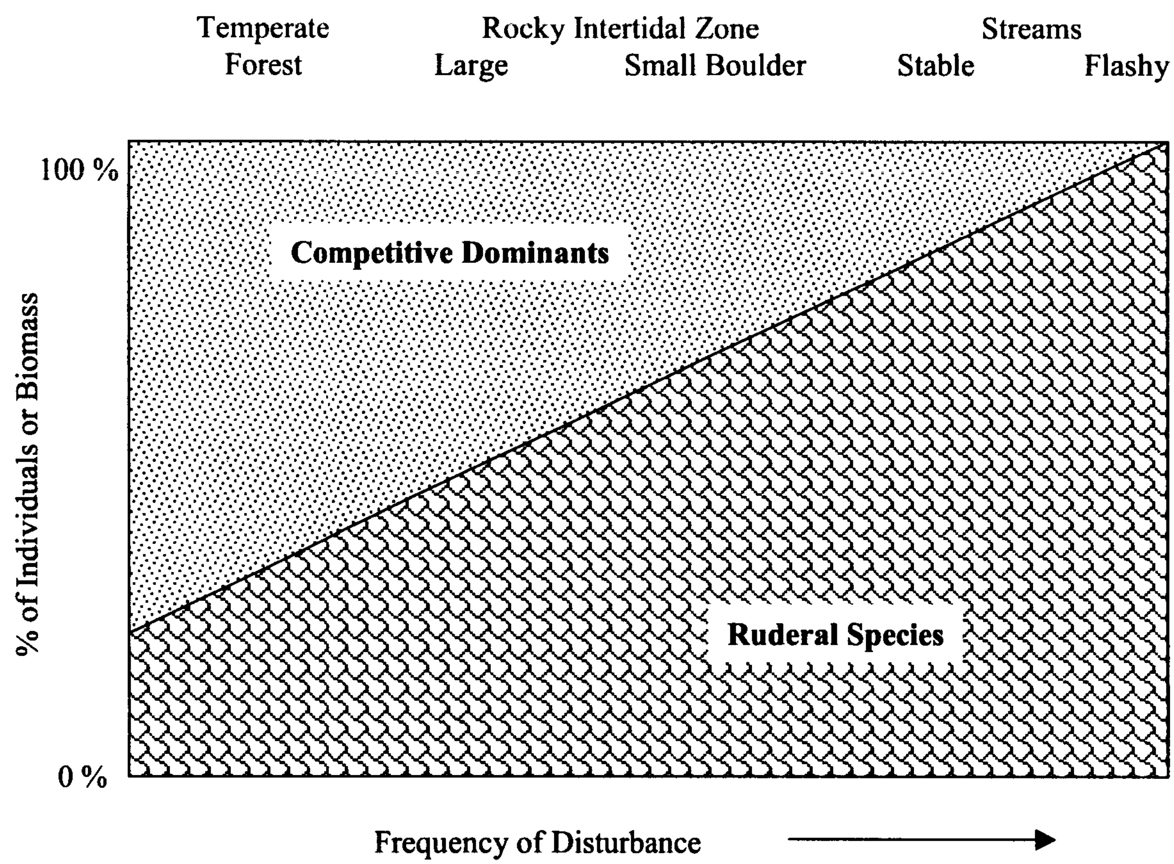
Because the physical, chemical, and biological characteristics of the stream ecosystem are a function of the watershed through which it flows, it would be natural to expect these characteristics in a given stream, especially the biological characteristics, to change in varying degrees over time. Temporal events, such as floods, climatic change, and changes in land use or vegetation of the watershed can all influence the biological nature of the stream (Giller and Malmqvist 1998).

Physically speaking, in the unmanaged environment, surface water channels owe their geometry not to the "100-year flood" or even the "10-year flood" but to the more common flood events with sufficient tractive energy to move and redeposit the dominant (e.g., D_{50}) particle size of their channel. These "channel-forming" flows (Leopold 1994) represent the optimization of energy and likelihood of occurrence. For streams in non-arid climates, the stream discharge that forms these floods is called bankfull flow and can be determined in the field from geomorphological measurements and inferences. For ephemeral streams the bankfull flow recurrence interval is between 1.1 and 1.8 years for channel-maintaining events (Moody et al., in press). These estimates are similar to the range reported by Leopold (1994).

The types of aquatic species expected at a given site can be influenced by the frequency of disturbance. Townsend (1989) hypothesized a general biological relationship between the frequency of disturbance and the relative abundance or biomass of competitive dominant species and species with life history adaptations that allow them to respond quickly to disturbance (e.g., continuous reproduction, rapid development, and lack of dormant life stages). With increasing frequency of disturbance, the percentage of abundance or biomass contributed by competitive dominants is expected to decline (**Figure 3-3**).

Evaluating the impact of flood events can be considered in the context of spatial scales. As a general rule, the greater the impact to the watershed (e.g., loss of riparian vegetation, change in the active channel, or sedimentation), the longer the response time of the aquatic and terrestrial fauna following disturbance. Recovery from flood disturbances is a dynamic and variable process and the endpoint used to measure recovery (e.g., function or taxonomic) may result in different expectations with regard to the length of the time required for the biological community to recover (Wallace 1990).

A review of the literature by Wallace (1990) and Chadwick Ecological Consultants (2000) (**Appendix I**) suggests that recovery in frequently flooded systems in arid regions requires about 2 months, while recovery in more temperate regions requires 4 to 12 months. The shortest recovery periods have been observed in arid Southwest streams in Arizona (e.g., Sycamore and Aravaipa creeks). Longer recovery periods have been observed in more temperate areas. For example, during a flood on Tesuque Creek (tributary to Rio Grande River in northern New Mexico), up to 3 feet of streambed was eroded resulting in a modified habitat (Molles 1985).



Hypothetical Relationship Between Frequency of Disturbance and Relative Representation of Species Adapted from Townsend (1989).

Figure 3-3

Because the habitat changes differentially affected different aquatic insect groups, Molles (1985) observed a variable recovery period of 60 days to two years depending on the aquatic insect species.

These observations on recovery periods were obtained from studies conducted on streams where the flood disturbance was generally limited to the active channel. Other studies have shown that in disturbances where the channel, as well as the adjacent floodplain, is impacted, response times can be somewhat longer. For example, Minshall et al. (1983) monitored the recovery of aquatic invertebrates in a river following the failure of a dam. Estimates of recovery time ranged from 1 to 3.5 years depending on the aquatic group. Meyerhoff (1991) studied the recovery of aquatic invertebrates on a stream impacted by the eruption of Mt. St. Helens. After 10 years, the aquatic invertebrate community still had not recovered to its expected potential. However, in this latter example, the entire watershed was significantly impacted, thus stream recovery would be expected to be relatively slow.

Evaluating the impact of a natural or anthropogenic event should consider both temporal and spatial scales. Understanding the relationship between these scales can be important for evaluating how an activity or event could impact elements of the aquatic community. For example, an evaluation of deforestation impacts on stream channel morphology should occur at a different scale than an evaluation of the impacts from constructing a road crossing (Ward 1989).

Other spatio-temporal scales can be described. For example, lateral migration of river flows during a flood event (spatial) can be a seasonal event as well (temporal). In addition, migratory patterns of aquatic organisms can be closely tied to seasonal flood patterns, especially if such flooding is a relatively predictable event (Ward 1989).

3.1.1.5 Four-Dimensional Complexity

The discussion of each of the four dimensions described by Ward (1989) was presented to provide a means to organize the typical patterns and processes observed in natural stream channels. While useful in concept, stream ecosystems do not function along single dimensions independently, but along all four dimensions simultaneously. This natural complexity creates multiple pathways for interactions among physical, chemical, and biological processes.

Stanford et al. (1996) applied this complexity to an evaluation of habitats of valley bottoms, referring biophysical gradients of fluvial systems to systematic landform and habitat change resulting from differences in hydrologic processes. The biophysical continuum of Stanford et al. (1996) recognizes gradients (gradational changes through space and time) of all habitat types; thus, all biota from headwaters to a stream mouth result from and respond to down-valley change in geomorphic processes. This three-dimensional model includes ecological variation dependent on elevation of a geomorphic surface above mean stream level. The fourth gradient, time, is superimposed on this model relating biological processes that respond to events with a frequency component such as flood events, or long-term events such as climate change.

3.1.1.6 Discontinuity in the Landscape

It is important to note that within any of the dimensions suggested by Ward (1989) and discussed further by Stanford et al. (1996), discontinuities occur that can create a boundary between distinct regions, especially along longitudinal and lateral dimensions. Naiman et al. (1988) defined such a boundary as a "zone of transition between adjacent ecological systems, having a set of unique characteristics uniquely defined by space and time scales and by the strength of the interactions between adjacent ecological systems, i.e., an ecotone (Holland 1988)." Naiman et al. (1988) went further, stating, "In general, a boundary may be thought of as analogous to a semi-permeable membrane regulating the flow of energy and material between adjacent resource patches."

Within a natural landscape, distinct longitudinal and lateral boundaries can exist within a natural stream ecosystem (e.g., changes in geology). Examples of longitudinal boundaries within stream ecosystems include substantial changes in topographic relief, which result in significant changes in gradient, and locations where tributaries enter a main channel. Laterally, boundaries tend to be more distinct, for example the boundary between early successional vegetation along the active channel and mature vegetation on the floodplain but farther away from the active channel. An additional obvious boundary is the gradation between riparian and upland vegetation, which can be gradual or distinct depending on the location of the waterbody.

The distinctness of the boundary between riparian and upland vegetation can be sharp in arid regions where water availability limits the establishment of riparian vegetation. Often, riparian zones exist as distinct ribbons immediately adjacent to the active flowing channel. Within a short distance laterally of the channel it is common for riparian vegetation to quickly give way to vegetation adapted to hot, arid conditions. This condition is in contrast to non-arid regions where the gradation from riparian vegetation to upland vegetation can be marked by a comparatively large transition zone. Coincidental with the marked vegetation boundary in arid West streams between riparian and upland areas is the marked change in other parameters such as temperature, cover, food resources, nesting habitat, and other factors.

Exacerbating the ribbon effect is the interplay of geomorphology and hydrology that creates even more linear patchiness in riparian ecosystems. Understanding the interdependence of riparian structure and fluvial geomorphology began with several pioneering studies (Hupp 1983; Harris 1986; Osterkamp and Hupp 1984, 1985; Sigafoos 1961; Turner 1934). Harris (1986) provides a good review of the dependence of riparian microhabitats on floodplain structure.

3.2 EFFLUENT-DEPENDENT WATER STREAM ECOSYSTEM

The input of wastewater at one or more discrete points along a river continuum is an unnatural event in the context of a stream ecosystem. Consequently, it is not surprising that the discharge of effluent has the potential to fundamentally change the physical, chemical, and biological processes expected in a natural stream. This change is analogous to other anthropogenic flow regulation activities that result in a significant disruption of natural longitudinal patterns. Ward and Stanford (1983) and Stanford et al. (1988) conceptually described this disruption in their model called the Serial Discontinuity Concept. Not only does the sudden addition of effluent create a discontinuity in the natural flow of a stream system, but addition of effluent also creates a boundary between two longitudinal stream segments. The result of this anthropogenically created boundary is the establishment of a distinct ecotone with physical, chemical, and biological characteristics different from the adjacent upstream reach.

The following section discusses the types of changes observed as a result of the review of historical and site reconnaissance data gathered for the Habitat Characterization Study. These sections document the changes that were observed to occur across the discontinuity created by the addition of effluent. It can be assumed that the types of effects observed would apply to other effluent-dependent waters where the stream prior to discharge was located in an arid or semi-arid region and had an alluvial channel where flow was intermittent or ephemeral. Some of these observations are obvious, but they are still discussed given their implications with regard to how the observation potentially impacts expectations for aquatic and terrestrial communities.

3.2.1 Physical Observations

To some extent, each of the 10 study areas examined by the Habitat Characterization Study exhibits some departure from physical equilibrium. Each stream is in the process of changing its shape and drainage pattern to adjust to some sort of disturbance. In some cases, the introduction of effluent has had an impact on this adjustment, while in other cases the effects are minimal. Before these observations are discussed, a general concept of physical equilibrium must be agreed upon.

As Stanford et al. (1996) point out, the human regulation of flooding periodicity can importantly change downstream biodiversity. Other modifications to the river continuum can provide further impacts on the distribution and abundance of biota, including the introduction of thermal energy, pollutants, and non-native species; however, the regulation of flow is the most important physical modification that occurs in effluent-dependent water.

Wastewater is typically sediment free, resulting in energy available for sediment transport. This disequilibrium between sediment and flow results in an environment favorable for incision of the alluvium (Note: on systems where the stream has cut down to the local bedrock, such vertical incision is limited). Some incision may be offset by vegetative stream bank stabilization. Regardless, if the stream sediment regime is responding to human or natural disruption and a geomorphic threshold has been exceeded, no amount of vegetation can preserve the channel morphology (Schumm et al. 1984).

The physical effects of effluent discharge are attenuated as the flow progresses downstream. This attenuation occurs as the flow and sediment carried by the system move toward some new equilibrium. Seepage additions or losses occurring in the downstream direction influence surface flow and sediment carried as suspended load or bed load gradually changes.

Many of the streams examined in the Habitat Characterization Study were incised into their floodplain. For example at the WWTP outfall, the Santa Fe River was incised several inches right at the point of discharge. The clear water of the treated effluent became turbid less than a mile downstream.

The creation of a constant flow in an otherwise dry or intermittently flowing channel creates a saturated zone below the channel. This saturated zone can extend laterally from the channel edge to the edge of or beyond the floodplain, if the water is available in the stream and can be transported laterally at sufficient velocity to make up for losses and evapotranspiration. A confining, or low-permeability, zone under coarser stream deposits can extend this effect.

The nature of the saturated zone dictates to a large degree the extent to which riparian vegetation can become established adjacent to the channel. While the development of riparian vegetation and associated wildlife habitat along an effluent-dependent channel is an important benefit of discharge to an intermittent or ephemeral stream, the growth of riparian vegetation also serves to stabilize the channel downstream of the point of discharge. Where erosion and channel incision are great or where management activities prevent the establishment of a natural flow pattern (e.g., flood control activities), riparian vegetation may be slow to establish, resulting in a less stable channel.

Urban characteristics, often associated with the effluent-dependent water, add layers of complexity to physical changes expected as a result of the discharge of effluent into an otherwise dry channel. For example, flood control activities such as channelization, cementing of banks, and grade control structures, designed to facilitate the transport of storm flows, can affect the natural tendency of the effluent-dependent water to find a new equilibrium, thus limiting normal expectations for in-stream habitat and establishment of riparian vegetation. At most sites anthropogenic activities other than effluent discharge also have influenced the physical characteristics of the stream system.

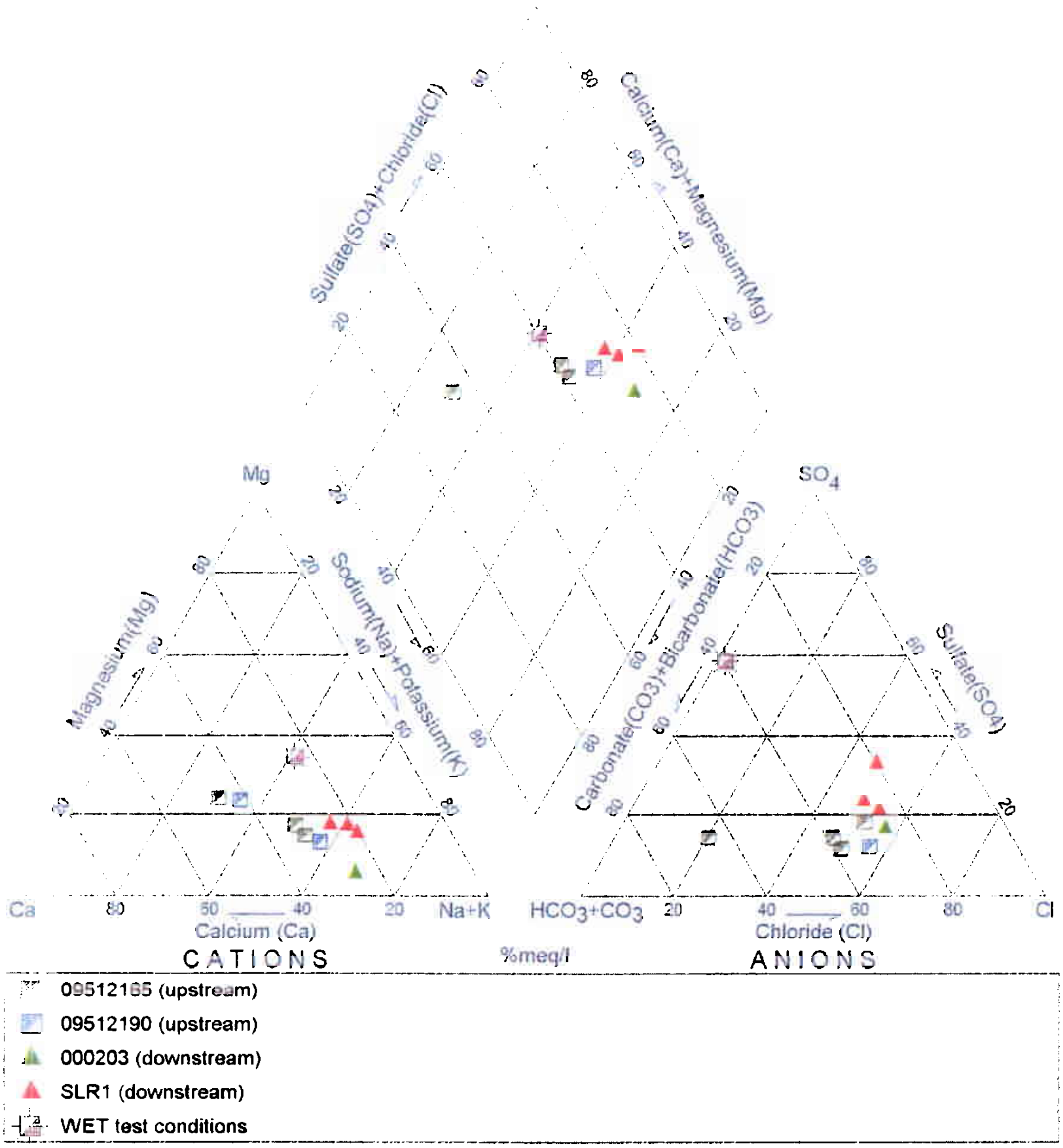
Physical changes caused by the addition of effluent are transient, both spatially and temporally, as long as no additional changes are imposed on the system (e.g., additional effluent discharges or flood control activities). Left to itself and given sufficient time and space, the stream channel, even with the addition of effluent, will return to a state that is in equilibrium with the physical characteristics of the channel (e.g., gradient, bed load). However, given that effluent-dependent waters naturally tend to be associated with urbanized environments, the likelihood of finding an effluent-dependent water in equilibrium with its channel is low.

Despite the additional discharge imposed upon it, the stream will still need to convey stormwater, and if all else is unchanged, these flows will be similar in magnitude and frequency to the pre-effluent state. Obviously, any structures constructed in the channel will need to withstand the enormous and rapid changes in discharge between effluent flows and storm flows. The channel morphology also will be modified by both flows. There are many streams in the arid West in which small perennial flows co-exist with occasional and large stormwater discharges. However, predicting the exact behavior of an effluent-dependent floodplain under an extreme discharge is complex and probably site-dependent.

3.2.2 Water Quality Observations

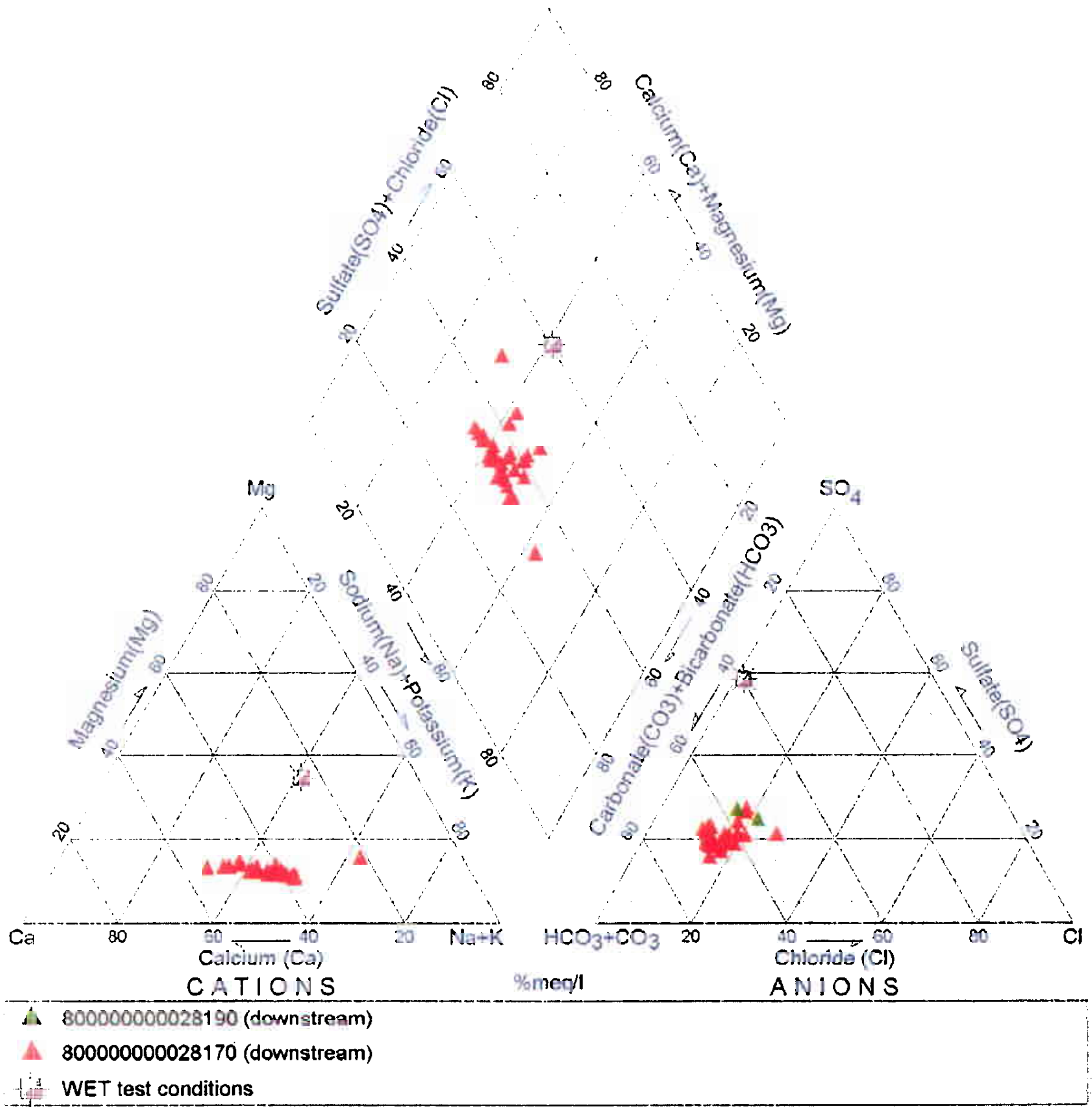
The chemical characteristics of each of the 10 study areas was documented to the extent possible from historical data (**Figures 3-4 through 3-18; Table 3-1**). A review of these data suggests that the chemical nature of flows in effluent-dependent waters is for the most part dependent on the characteristics of the effluent discharged to the stream channel. The degree to which in-stream

Salt River near Phoenix, Arizona



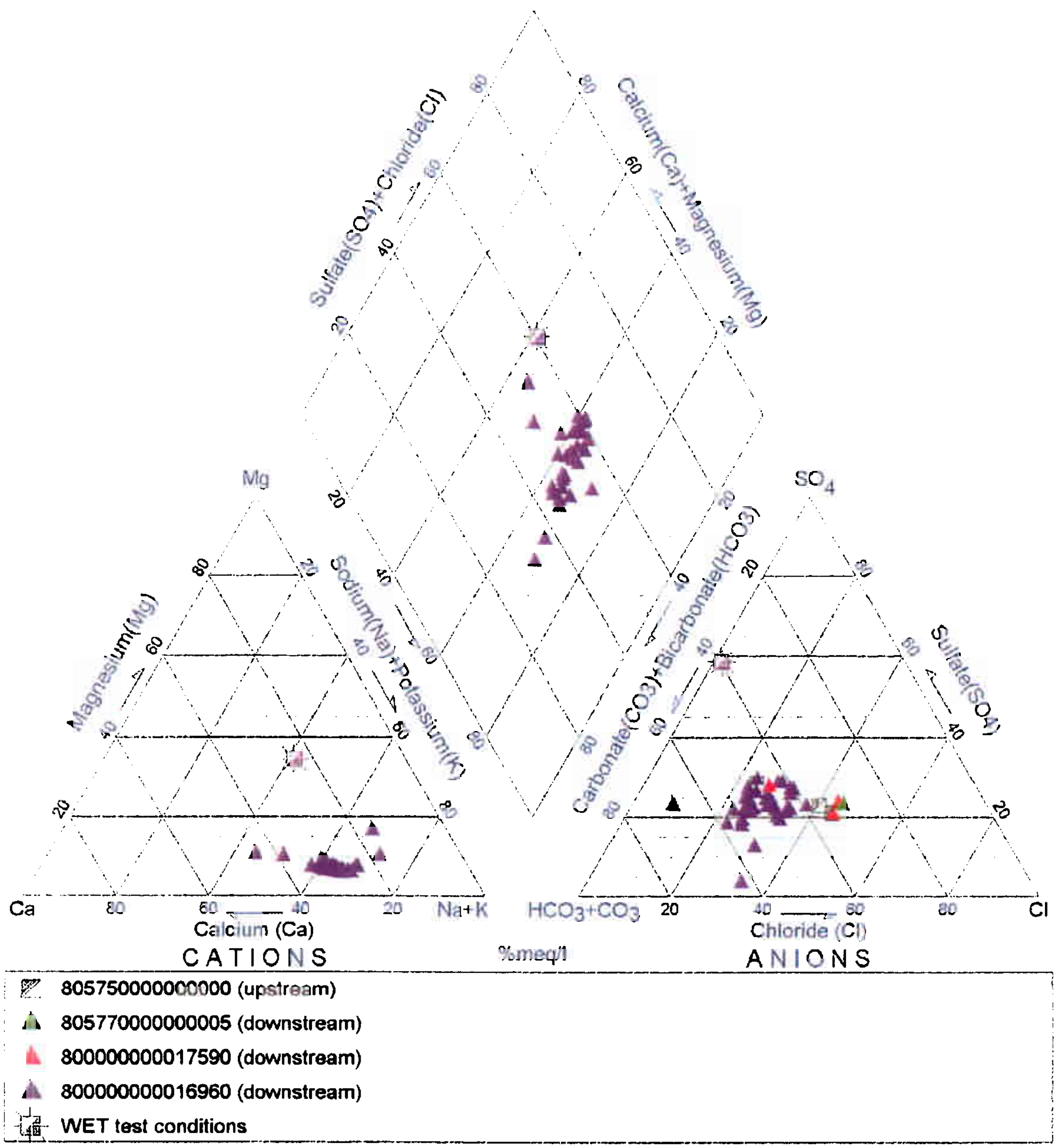
Piper Diagram for the Phoenix Study Area
Figure 3-4

Santa Cruz River near Nogales, Arizona



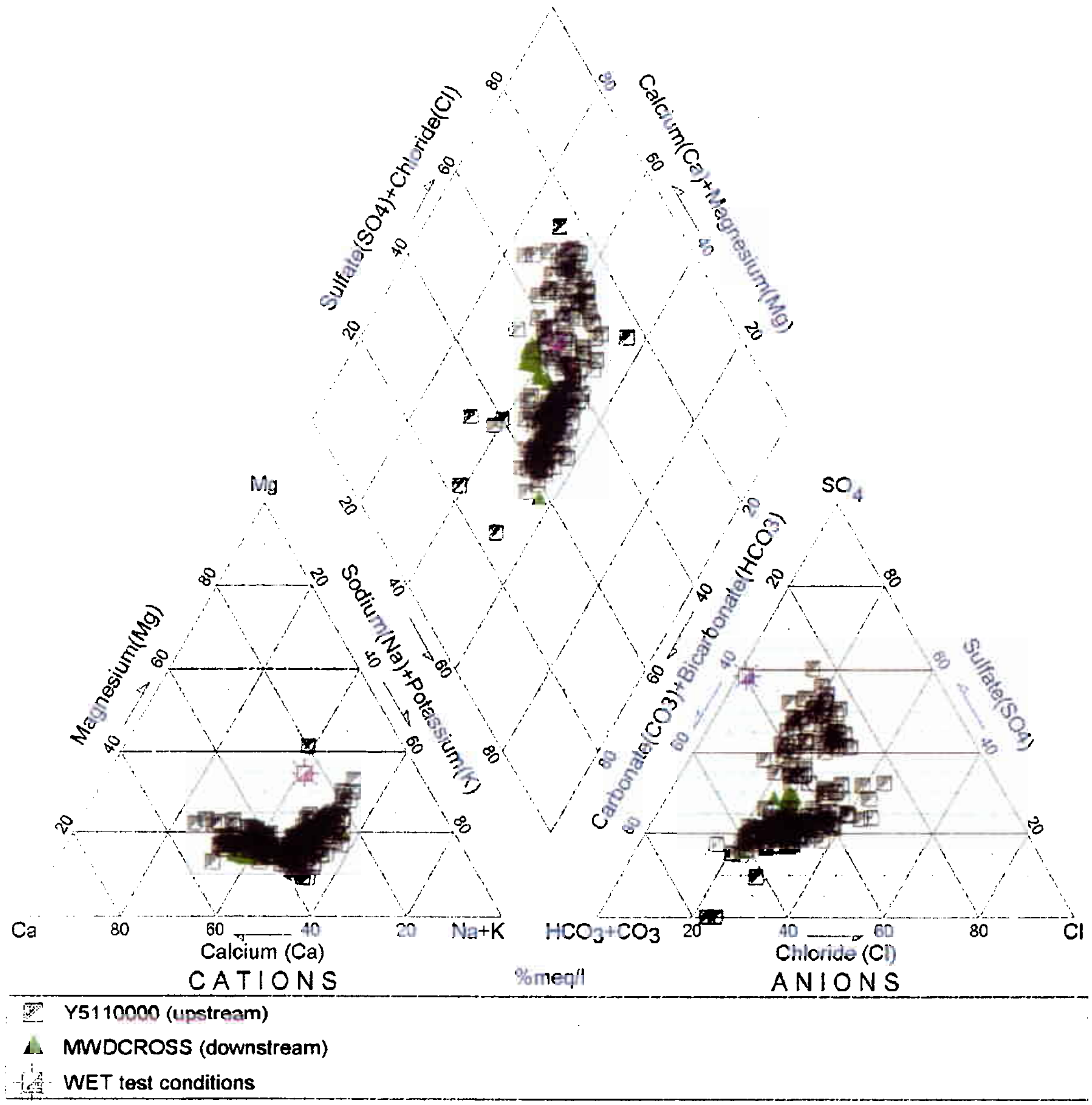
Piper Diagram for the Nogales Study Area
Figure 3-5

Santa Cruz River near Tucson, Arizona



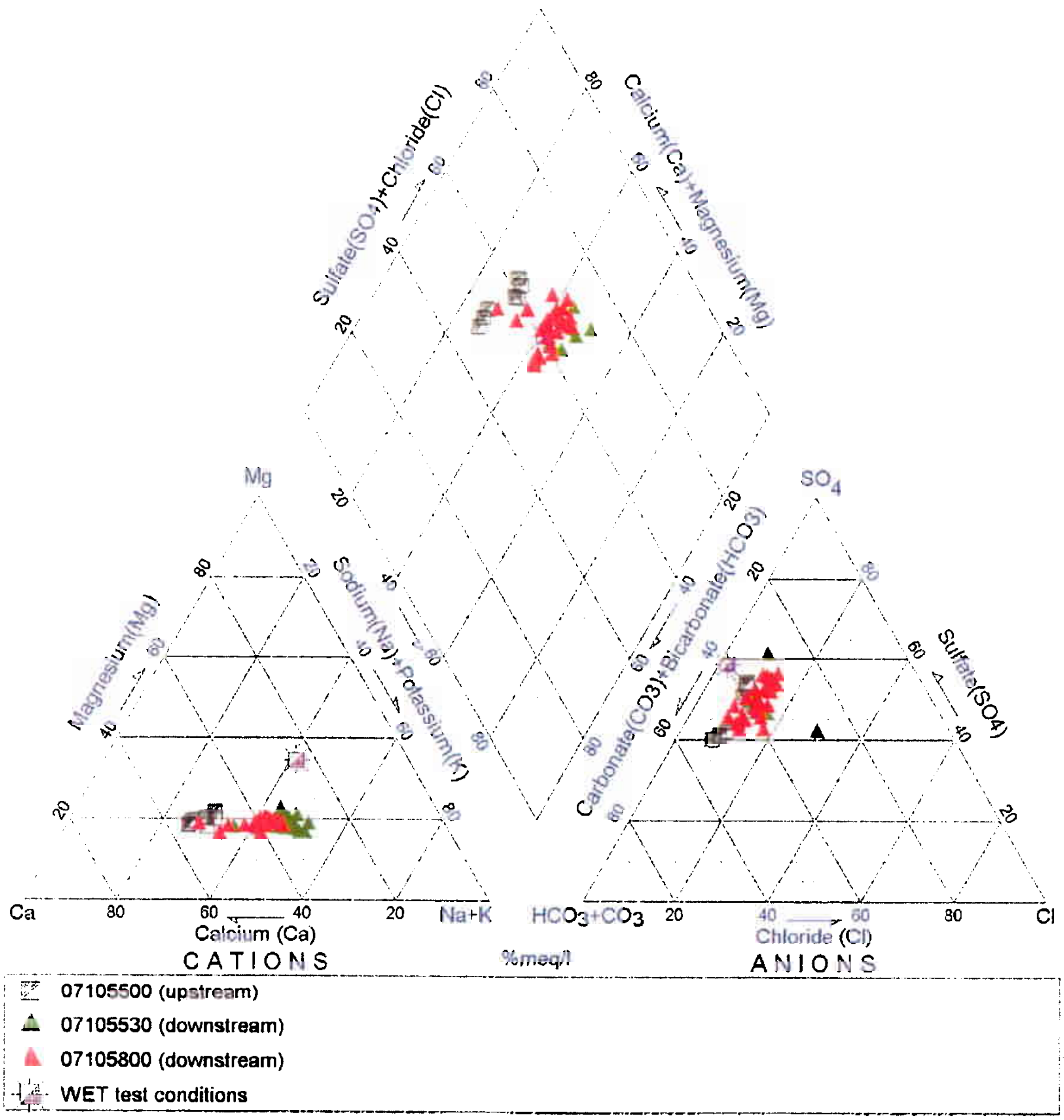
Piper Diagram for the Tucson Study Area
Figure 3-6

Santa Ana River near San Bernardino, California



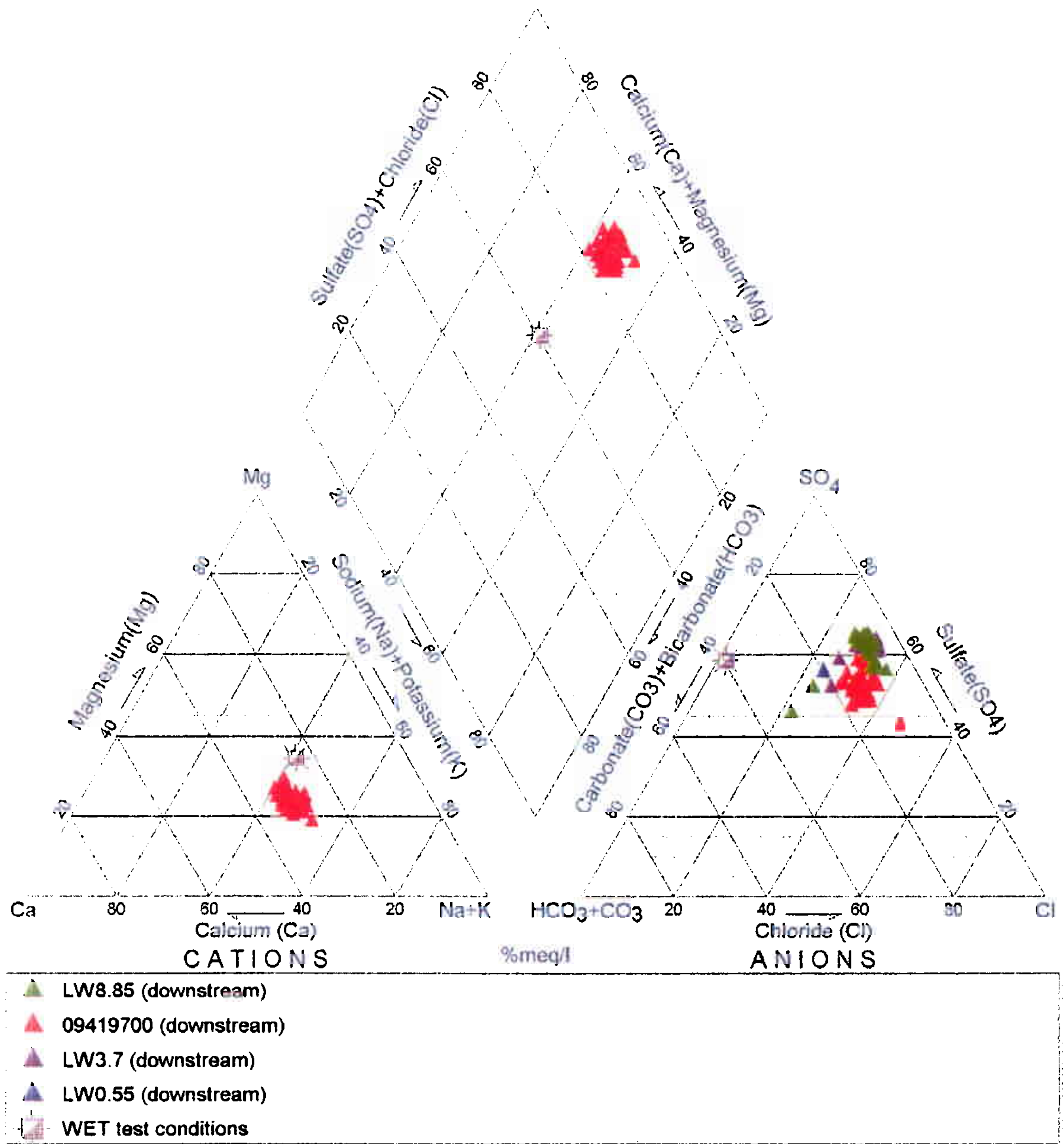
Piper Diagram for the Santa Ana Study Area
Figure 3-7

Fountain Creek near Colorado Springs, Colorado



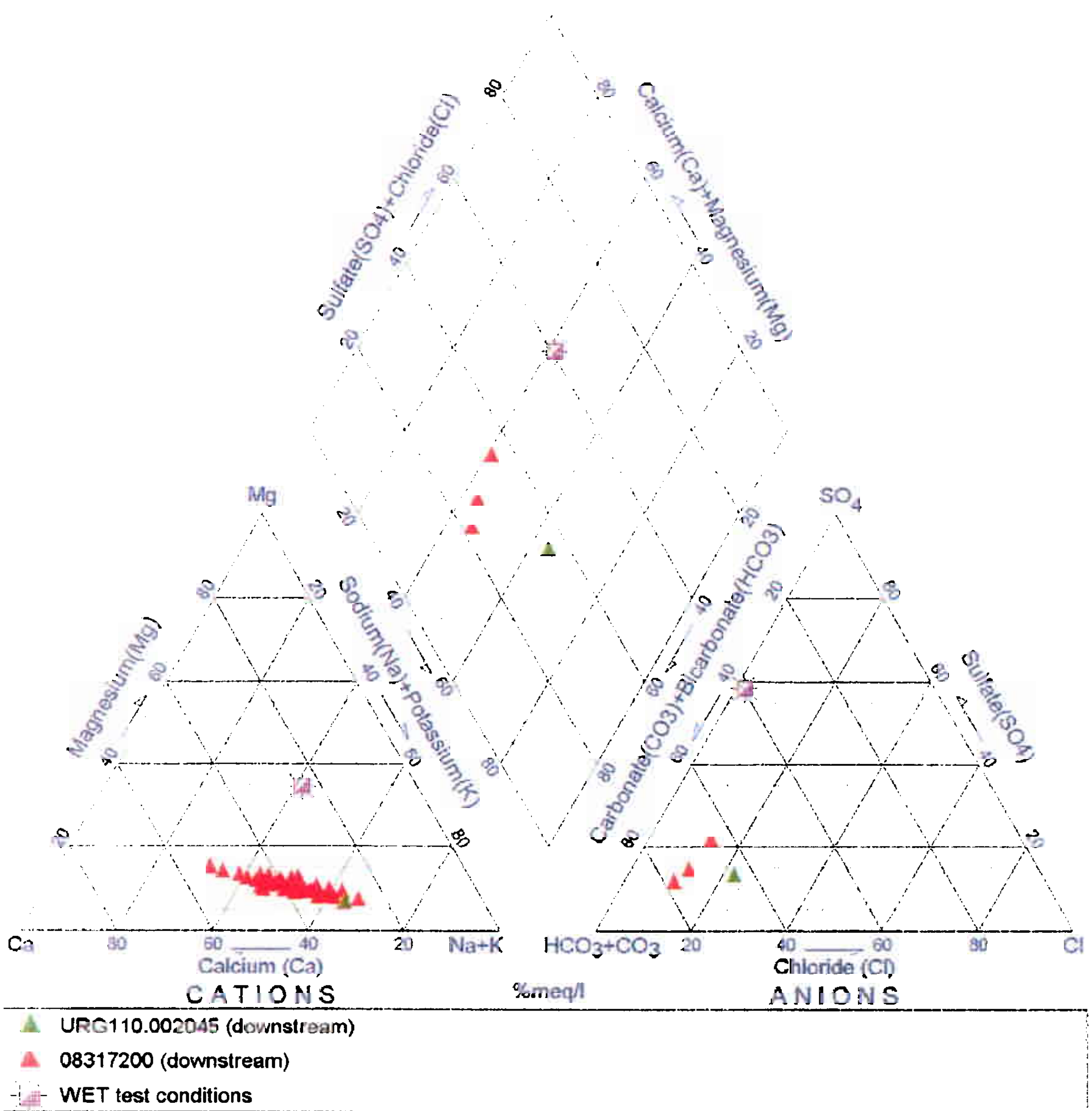
Piper Diagram for the Fountain Creek Study Area
Figure 3-8

Las Vegas Wash near Las Vegas, Nevada

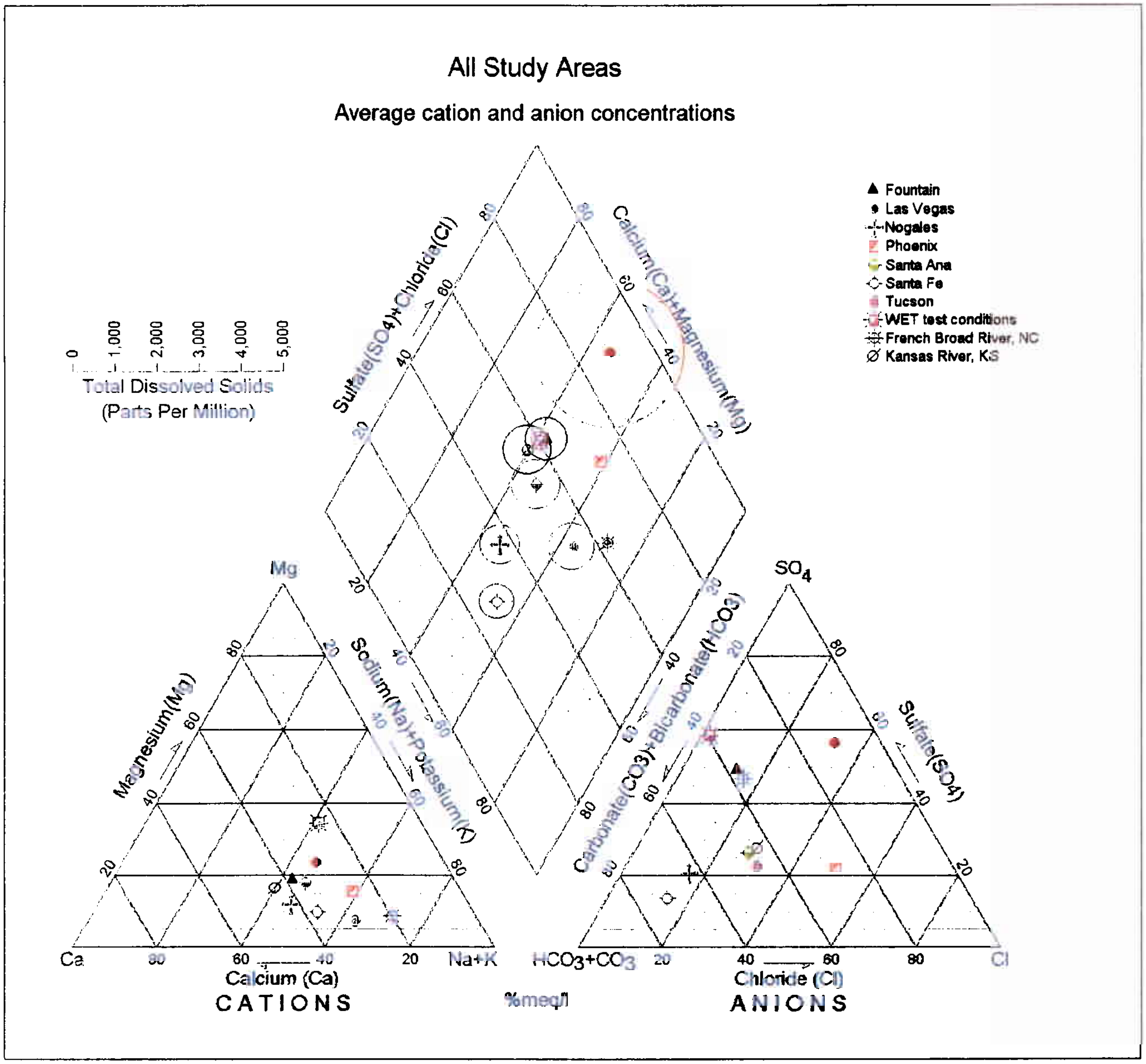


Piper Diagram for the Las Vegas Study Area
Figure 3-9

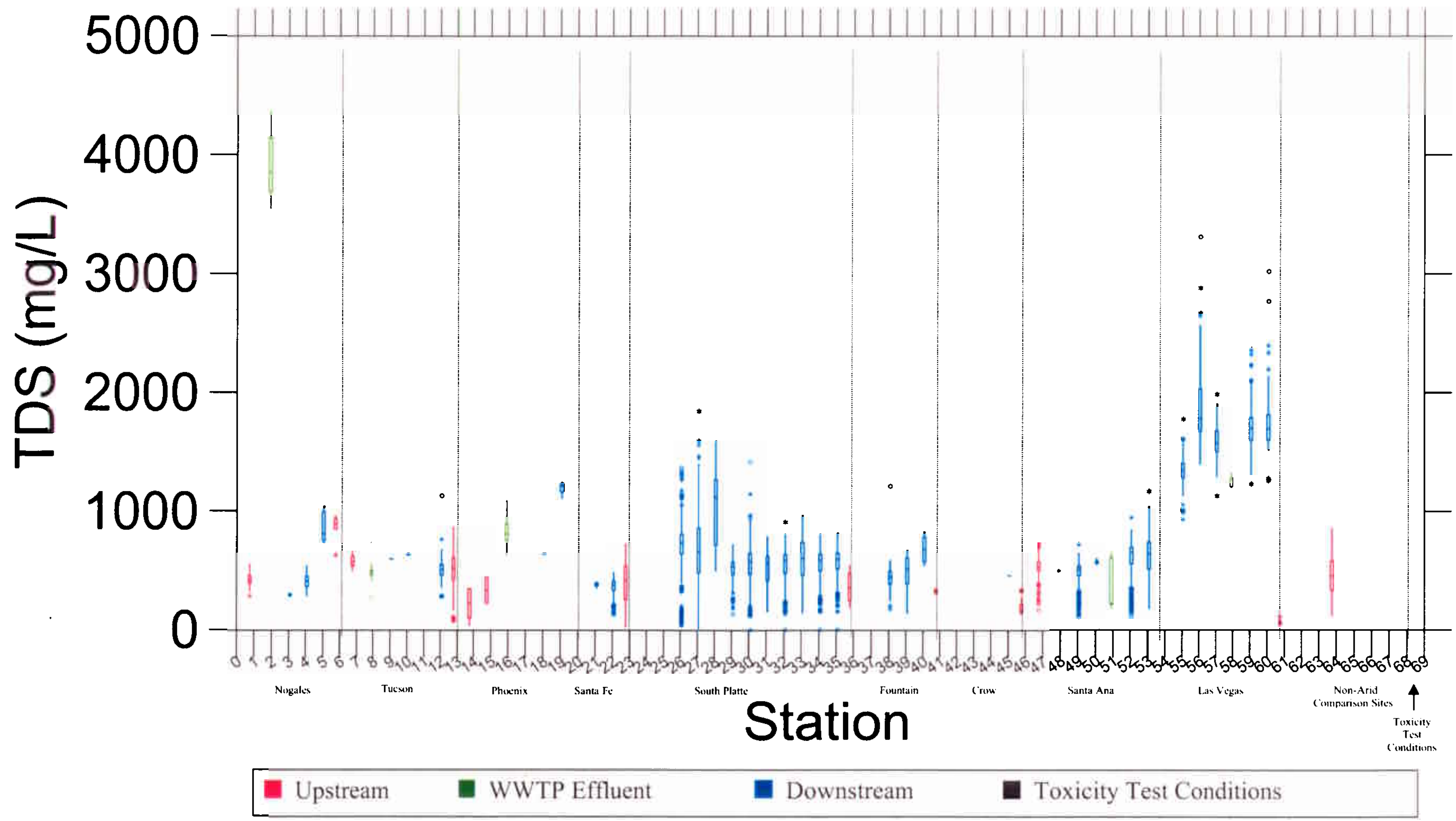
Santa Fe River near Santa Fe, New Mexico



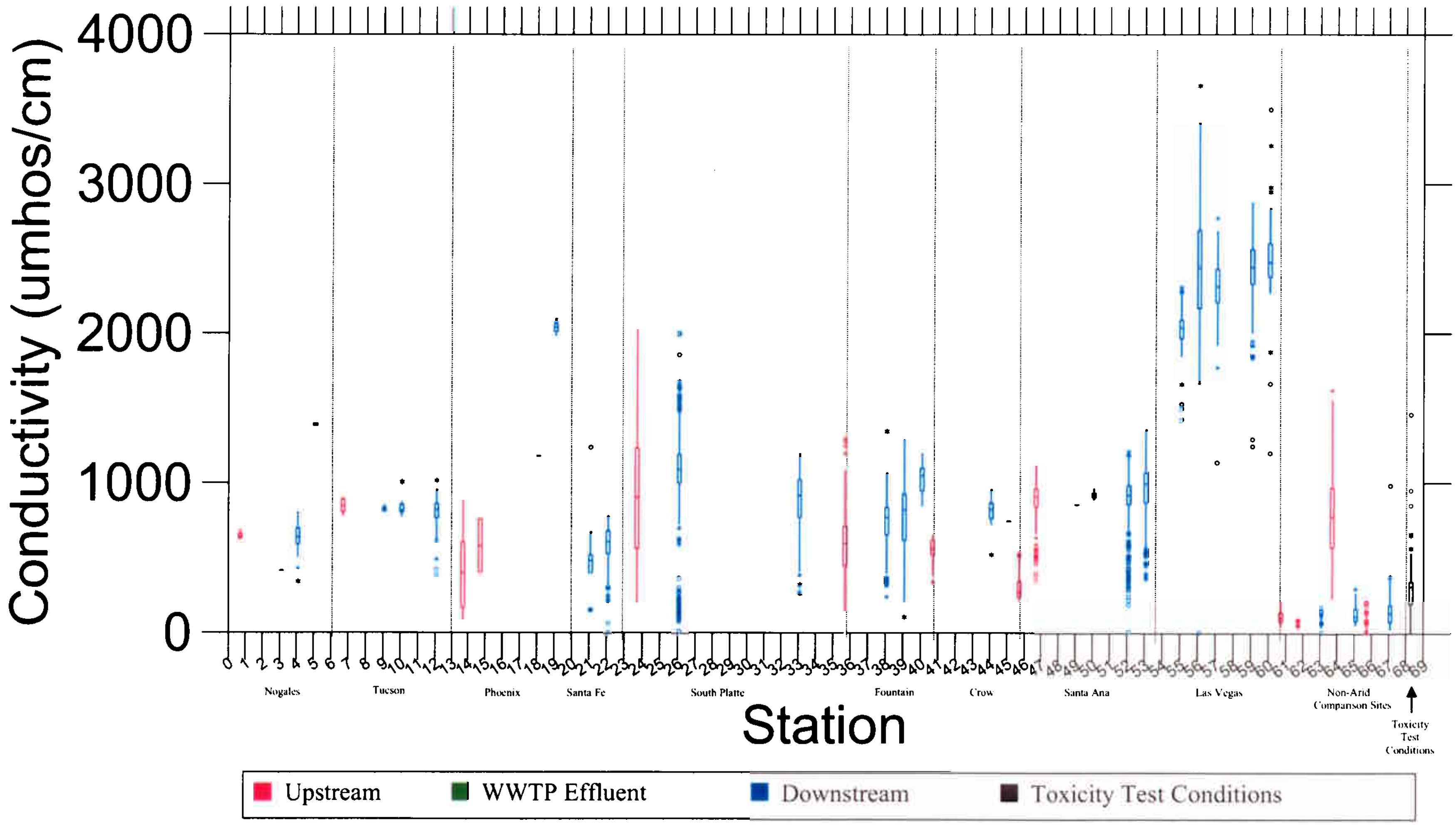
Piper Diagram for the Santa Fe Study Area
Figure 3-10



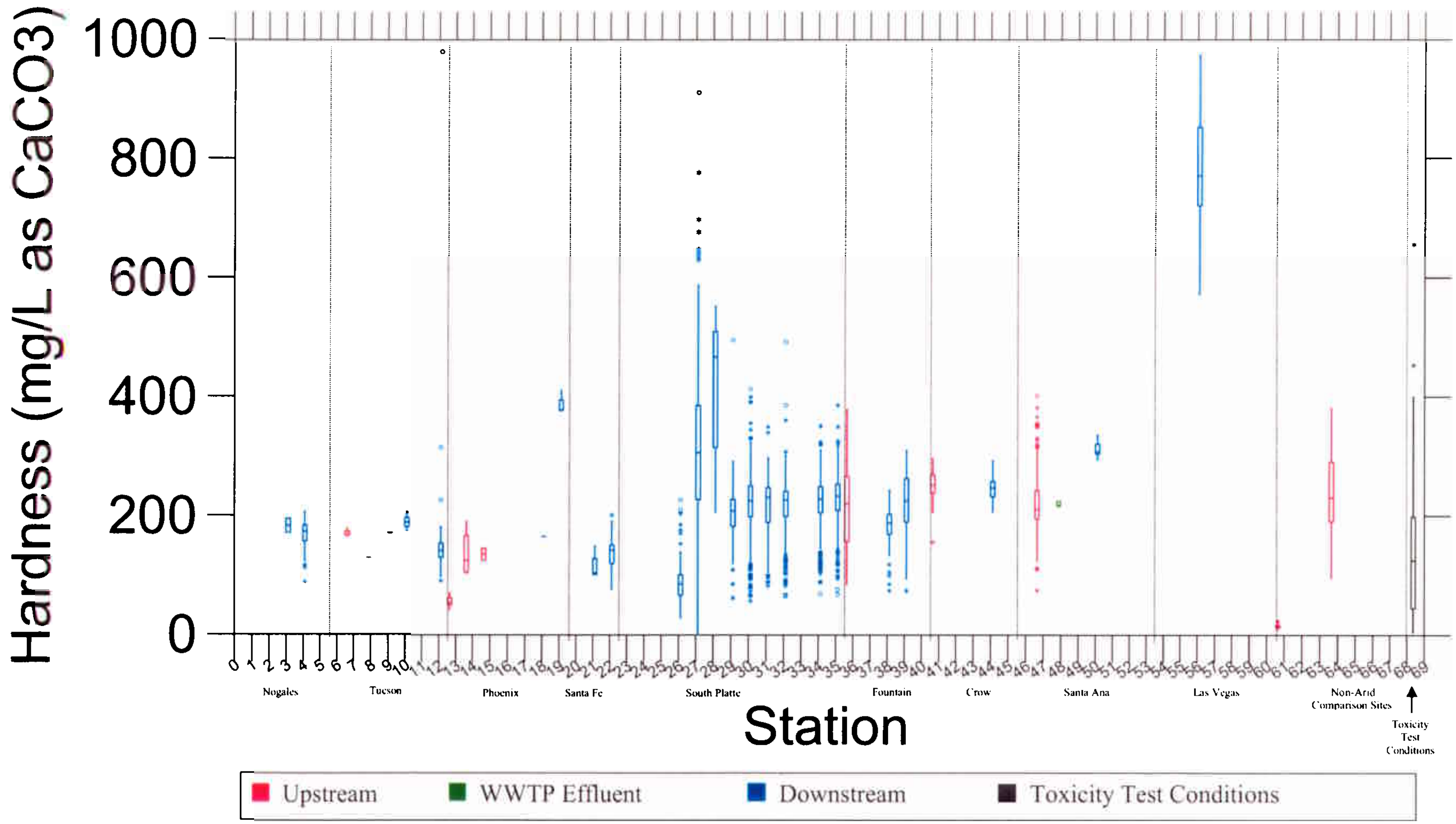
Piper Diagram Showing Average Cation and Anion Concentrations for 7 of the 10 Study Areas and Non-Arid Streams
Figure 3-11



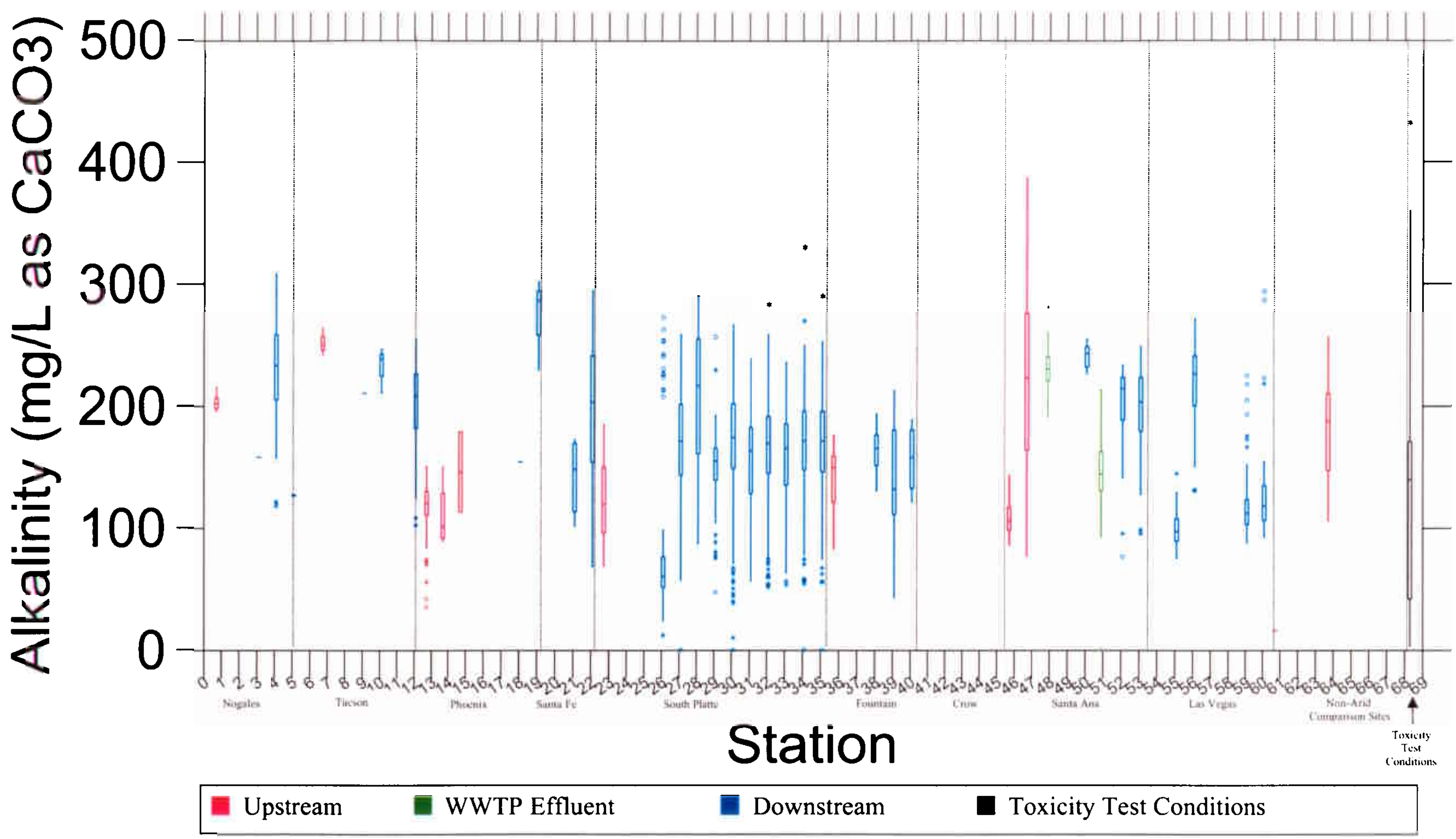
TDS Box Plots
Figure 3-12



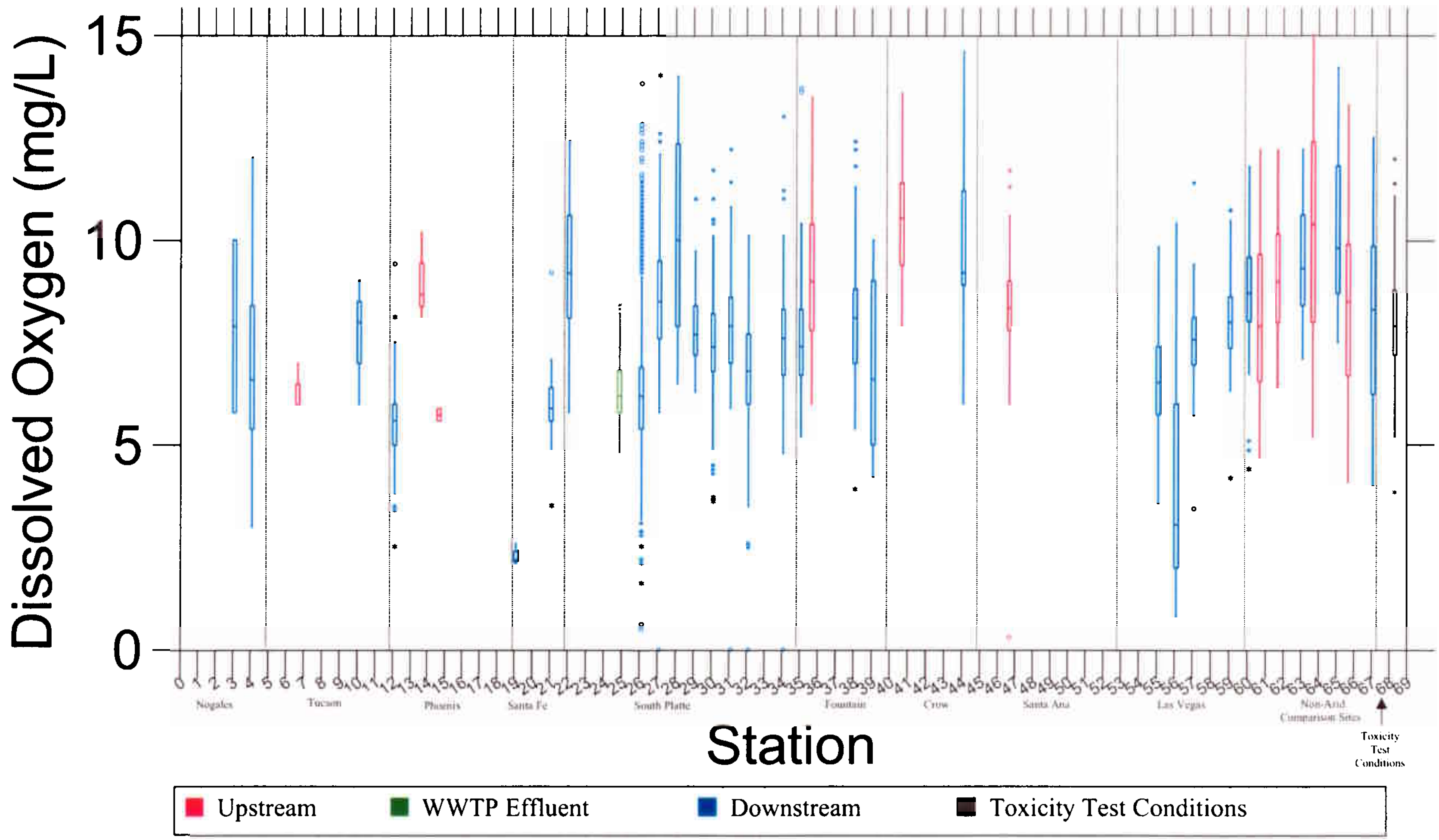
Conductivity Box Plots
Figure 3-13



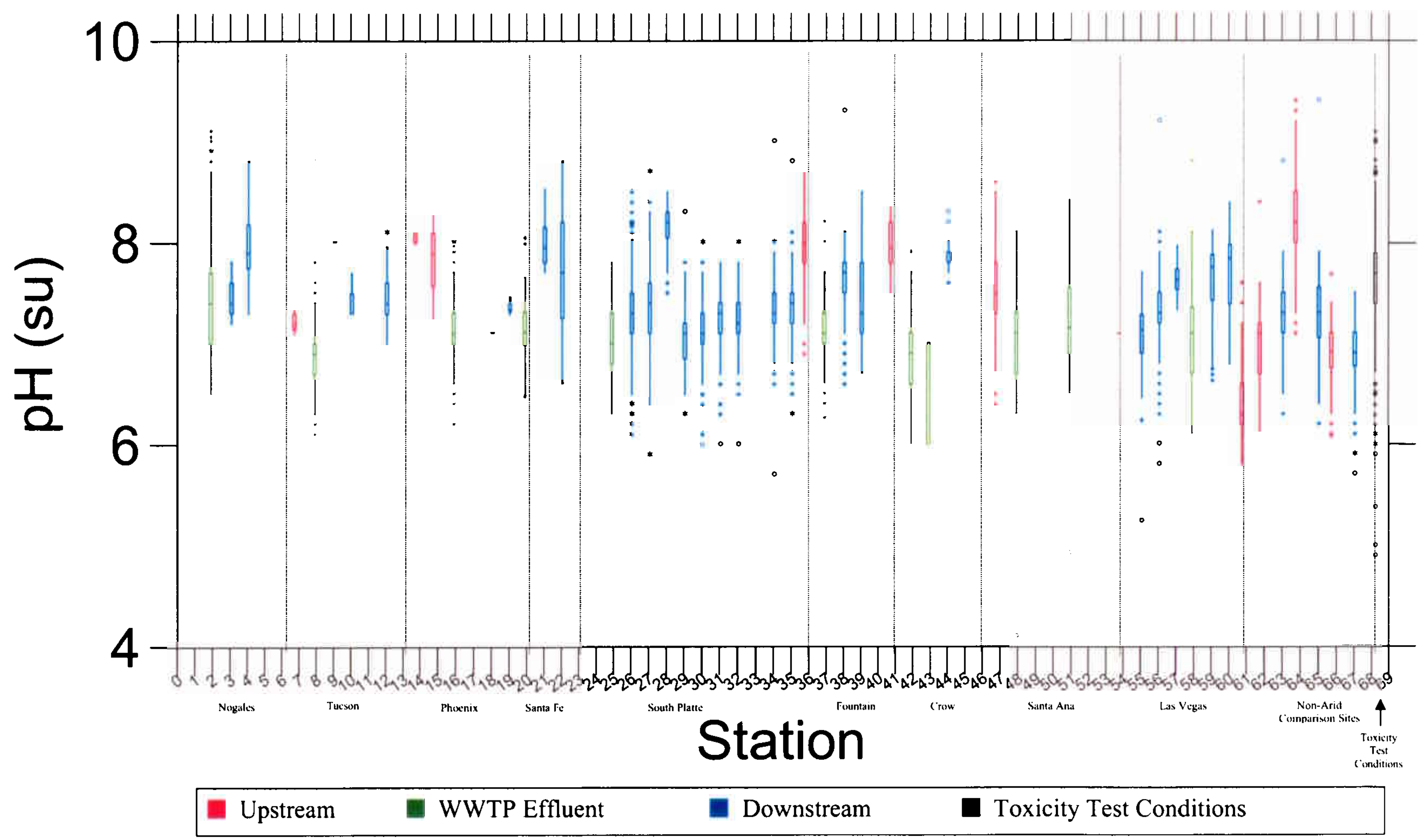
Hardness Box Plots
Figure 3-14



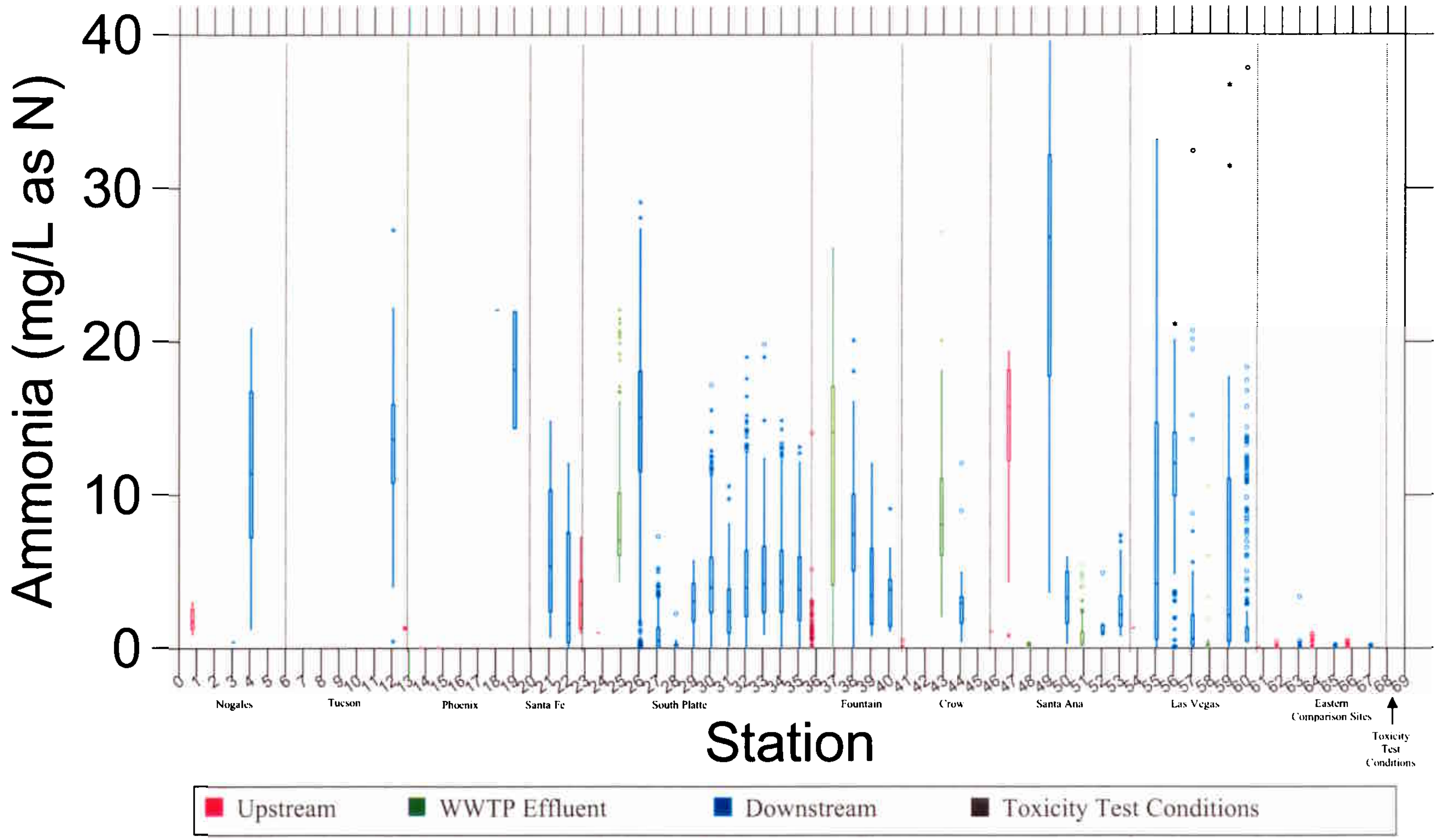
Alkalinity Box Plots
Figure 3-15



Dissolved Oxygen Box Plots
Figure 3-16



pH Box Plots
Figure 3-17



Ammonia Box Plots
Figure 3-18

quality is the same as effluent quality depends on how much in-stream flow is available for mixing. Given that the stream ecosystem without effluent is intermittent or ephemeral, mixing is typically limited or non-existent for most of the year.

The quality of the effluent is directly related to the types of treatment processes. Effluent quality often remains somewhat constant over a long, continuous period, but it is possible to have variable quality, which is dependent on diurnal or seasonal patterns associated with the influent entering the WWTP. If upstream flow is present, the quality of the effluent typically will be significantly different from the quality of the upstream flow. In some cases (e.g., Fountain Creek or the South Platte River), seasonal flow may briefly mix with the effluent, temporally changing in-stream water quality.

Table 3-1
Station ID Key for Box Plots

Box Plot No.	Station ID	Box Plot No.	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)

Table 3-1
Station ID Key for Box Plots

Box Plot No.	Station ID	Box Plot No.	Station ID
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)		

Although the chemical and physical composition of the effluent is fairly constant at the point of discharge, these characteristics often change with distance downstream of the discharge as in-stream physical, chemical, and biological processes modify the chemistry. This is especially true for water quality parameters such as temperature, dissolved oxygen, pH, nitrogen, and phosphorus. For example, some degree of oxygen depletion can occur for some distance below the discharge point because of the high biochemical oxygen demand that can be associated with certain effluents, especially where there is minimal removal of organic matter prior to discharge. The degree to which oxygen depletion occurs, then, is dependent on the wastewater treatment process. However, because no dissolved oxygen data were collected as part of this study, the degree to which the different treatment processes associated with the 10 study areas specifically influenced dissolved oxygen levels downstream of the discharge is unknown. With regards to water temperature, it is believed that temperature likely will be relatively stable near the point of discharge, but with increased distance downstream, water temperature likely will tend towards a more typical diurnal cycle.

3.2.3 Aquatic Biology Observations

The input of a continuous stream of effluent at a discrete point along an intermittent or ephemeral stream significantly changes the aquatic habitat of the natural system. Thus, expectations associated with the natural community of the natural system are different from expectations associated with the effluent-derived system. Naturally intermittent or ephemeral streams have a biota that is adapted to the harsh, unpredictable flow regime associated with these streams. The extremes of lack of water and too much water during storm events limits the kinds of aquatic organisms that normally colonize this environment. In contrast, the effluent-dependent water is stable in terms of flow being present on a continuous basis. This stable flow naturally would be expected to cause a shift toward an aquatic community indicative of such stability. However, additional factors influence what actually will colonize the effluent-dependent water, including both factors linked to habitat stability and effluent quality and the proximity and connectivity of the site to colonization sources.

As indicated above under the discussion of physical characteristics, the discharge of effluent into an otherwise dry or intermittent channel creates disequilibrium between flow and sediment transport, often resulting in an erosive environment. This habitat limitation, coupled with the chemical characteristics associated with certain effluent types (e.g., those resulting in high biochemical oxygen demand in the receiving water), is not particularly supportive of a diverse community of aquatic organisms, whether plant or animal. With increased distance downstream of the discharge, both habitat and chemical limitations are ameliorated as the stream system reaches a new equilibrium. The distance between the point of discharge and equilibrium is somewhat unpredictable, depending on many factors including local geology, effluent volume and quality, presence or absence of additional anthropogenic structures, or activities that impose additional constraints on the stream channel.

Superimposed on expected habitat and chemical limitations is the natural flow regime of the watershed. Unpredictable, flashy flood events are a natural component of streams in an arid environment. Therefore, even without the potential limitations imposed on aquatic communities from the discharge of effluent, the naturally existing aquatic community would be somewhat limited in species richness and have varying abundance. The influence of unpredictable, flashy flood events remains a potential stressor on the system with or without the discharge of effluent.

The discharge of effluent fundamentally changes the aquatic system by providing a somewhat stable source of water for some distance downstream. The stability of this water source, however, will be variable from site to site, depending on seasonal cycles in wastewater discharge and competing uses for the water (i.e., water discharged to a riverbed may be diverted for other uses downstream).

Observations of effluent-dependent waters reveal that the aquatic community has the types of characteristics expected in a system with a number of limitations imposed upon it. At or near the point of discharge, species richness is typically low, but abundance can be high. With increasing distance downstream of the discharge, two processes occur: the stream tends toward a state of physical equilibrium based on the new flow conditions, resulting in improved habitat conditions, and limitations imposed by water chemistry are reduced. The expected and often observed result is increasing species richness. However, other limitations can be present that prevent this improvement from occurring. These limitations may be imposed by anthropogenic activities, especially in flood control activities in urban environments, or even naturally (e.g., naturally limiting habitat factors such as fine sandy substrates).

Because of the influence from factors creating instability, it is difficult to evaluate the role that a relatively stable source of flow might have on the aquatic biological community. Intuitively, we might predict that the effect would be one of increased richness, especially in an environment where the natural flow regime is unpredictable. However, two factors would appear to negate the positive benefits of stable flow: habitat instability and the lack of variability in the physical characteristics of the effluent. For example, in a recent review of factors limiting biodiversity in streams and rivers, Vinson and Hawkins (1998) identified a consistent negative relationship between species richness and the annual temperature range. Effluent typically discharges at a fairly constant temperature, thus creating an aquatic environment near the point of discharge with little temperature variability.

The Habitat Characterization Study did not measure biomass or biomass production in the effluent-dependent waters studied. However, the data suggest that even though taxonomic richness may be low, biomass production still could be quite high in effluent-dependent waters, especially among aquatic groups such as oligochaete worms and chironomid midges, which can have rapid life cycles and biomass production rates (Giller and Malmqvist 1998). This fact has important implications to the terrestrial community that develops adjacent to the effluent-dependent water and benefits from the transfer of energy (e.g., insect dispersal), from the aquatic system. Although oligochaete worms do not emerge, and therefore do not provide a source of food to terrestrial vertebrate species, midges do emerge and riparian plants do host a wide variety of insect and other invertebrate species that can be used by terrestrial vertebrates as a food source.

3.2.4 Terrestrial Biology Observations

The introduction of water into normally dry stream channels can have profound effects on the terrestrial vegetative systems that ultimately occupy the stream banks. Most notable is the vegetative response to water that is manifest in the development of emergent and riparian vegetation. Emergent vegetation is not a true component of the terrestrial vegetation system inasmuch as emergent plants generally have their root systems submerged in water while most of the photosynthetic vegetative portions of the plant "emerge" from the aquatic system and persist above the water line. However, depending on seasonal fluctuations in hydrology, some of the smaller wetland obligate (e.g., *Juncus* sp., *Eleocharis* sp., and *Scirpus* sp.) may form mosaics on the shore adjacent to marshlands (Brown 1994).

Riparian systems that develop as a result of wastewater discharge into normally dry channels may stand in stark contrast to the adjacent upland vegetation that is not influenced by discharge. Lowe's (1961) definition of riparian vegetation is probably the best. He defines riparian vegetation as that which occurs in or adjacent to drainageways and/or their floodplains, and that differs in species and/or lifeforms from that of the immediately surrounding vegetation. In the current study, sites at which riparian habitats are radically different in lifeform and species composition from the adjacent uplands include the Santa Cruz River at Tucson and Nogales; Salt River in Phoenix, Arizona; Las Vegas Wash in Nevada; Santa Ana River in California; and, to a lesser extent, the Santa Fe River in New Mexico. Vegetation along Carrizo Creek in Texas is also very different from the surrounding Chihuahuan Desert scrublands. Scattered willow and cottonwood stands along Crow Creek in Wyoming contrast with the surrounding Great Plains grasslands near Cheyenne. Likewise, Siberian elm, box elder, and cottonwood/willow associations along the South Platte River and Fountain Creek are different from the surrounding urban and grassland habitats through which the streams pass.

Riparian habitats, especially those in arid zones, are noted for the contribution they make to local, regional, and national wildlife populations. Vertebrate, especially bird, species diversity has been shown to be positively correlated with the complexity of vegetation structure in riparian systems (Anderson and Ohmart 1974; Carothers et al. 1974; Carothers and Johnson 1975; Knopf 1985). Lizard densities also have been shown to be higher in riparian areas than in adjacent nonriparian habitats (Warren and Schwalbe 1985).

Not only is the riparian vegetation that develops and is supported by the addition of a continuous effluent stream at a discrete point different from that of the adjacent non-watered uplands, it also may be radically different from streamside vegetation upstream of the effluent discharge point.

In the present study, the differences between upstream and downstream vegetation was striking at Tucson, Nogales, Phoenix, Las Vegas, and Santa Fe; notable at Santa Ana; and less pronounced at Carrizo Springs, Cheyenne, Denver, and Colorado Springs. Of all sites, the upstream and downstream differences in streamside vegetation were less pronounced at the Colorado Springs (Fountain Creek) and Denver (South Platte River) study areas. The degree of contrast is dependent on the volume and seasonality of upstream flow wherein little or no upstream flow (e.g., Santa Cruz River at Tucson) results in sharp contrasts compared with sites with relatively continuous upstream flow (e.g., Fountain Creek and South Platte River).

Farther downstream from the effluent discharge point, the vegetation associated with the wastewater stream may be very similar to naturally occurring vegetation on perennial rivers in the same vicinity. In cases where the discharge point is in a river that has been dried out by past human actions, the vegetation may mimic what was in the same location prior to those actions. The width of the riparian zone associated with the effluent stream will be related to the quantity of water available and to the geomorphologic characteristics of the stream channel. Generally, these zones are wider than the upstream riparian zones, and the downstream areas have more vigorous plant growth because of the greater availability of water. Differences in vegetation downstream from discharge points generally are related to increased channel width and/or braiding compared with discharge points, which are most often relatively narrow and confined.

Plant species alpha diversity in the effluent-dependent riparian zone may or may not be greater than the upstream zone. If the upstream area is normally dry, the vegetation may be limited to drought-tolerant species. A riparian zone downstream from this condition is likely to have greater species diversity because of the greater availability of water. In contrast, if the upstream area has some water, it may support both aquatic and dryland species. In this case, the downstream diversity may be limited by the greater abundance of water. As noted earlier, exotic species may dominate effluent-dominated waterways resulting in low species diversity. In this study, we noted a strong dominance by giant reed (*Arundo donax*) on the Santa Ana River in California. Similar strong dominance by salt cedar (*Tamarix ramosissima*) was noted on the Gila River near Phoenix, Arizona. In these two cases, upstream diversity was undoubtedly greater.

Vegetative structural diversity is usually greater in the effluent-dependent riparian zones. Upstream areas that are dry or have limited water availability are more likely to have an open structure with gaps of varying sizes. The reliable water source of the downstream riparian zone is more likely to support a multi-layered vegetation structure, with vigorous growth and high canopy coverage in the tree, shrub, and herbaceous layers.

Exotic plant species are a potentially serious problem in these effluent-dependent streams. Exotics are often better suited for rapid invasion of new habitats (i.e., the introduction of water into a normally dry streambed) than are native species. Exotic species also may have much higher seed production and germination rates, and be more tolerant of floods, fires, and inhospitable soil conditions (Rosenberg et al. 1991). Salt cedar is a common invasive species in many natural and effluent-dependent streams in the arid West. Russian olive and Siberian elm are relatively common in semi-arid locations. Giant reed is abundant in southern California. These species often grow in dense, single species stands, where they effectively prevent native species from becoming established. In addition to having very low species diversity, these stands also have limited structural diversity because virtually all of the growth is concentrated in a single layer. These stands generally provide relatively little suitable habitat for native vertebrate species.

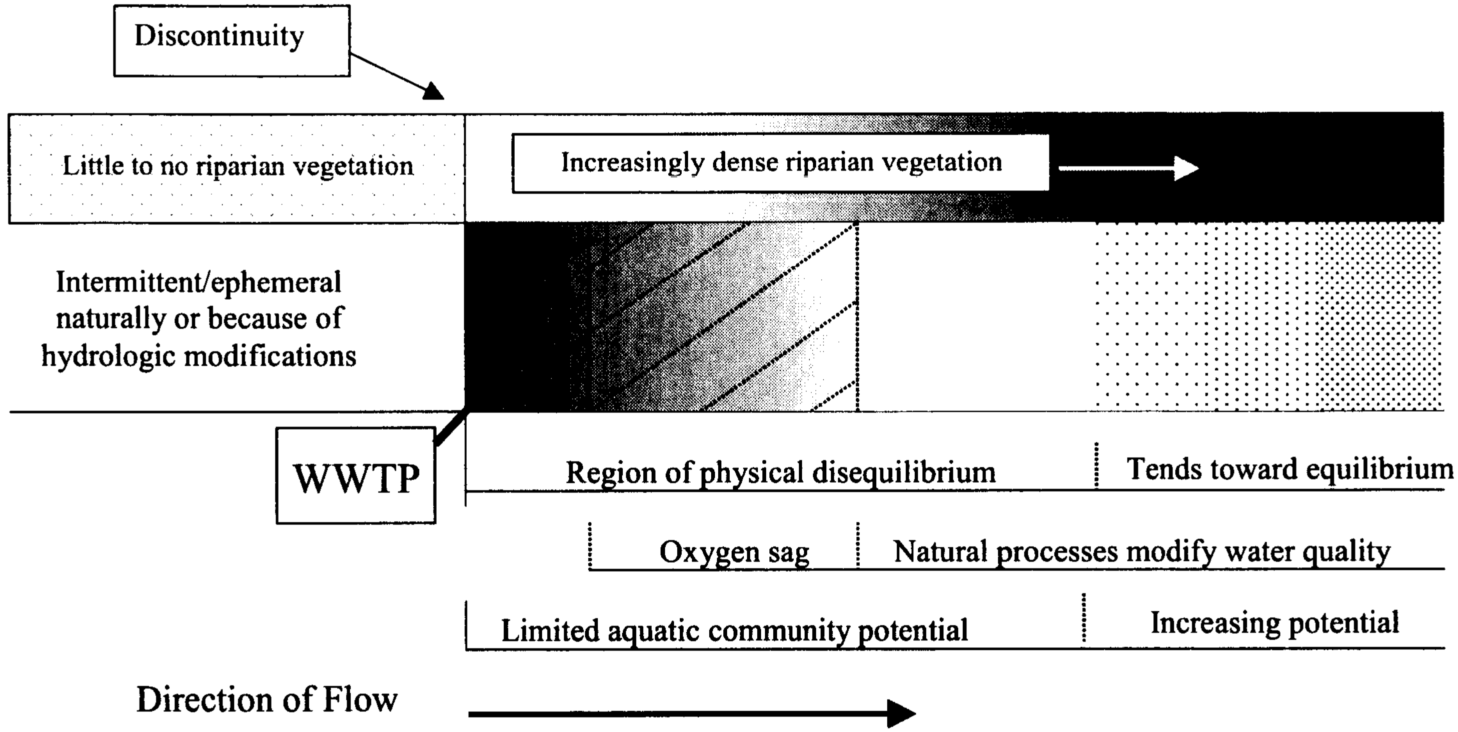
Like other riparian areas, the effluent-dependent riparian areas are particularly important for migratory bird species. The additional plant species diversity and vegetative structural diversity of these areas may provide temporary resting and foraging locations as well as possibly providing movement corridors for some species. Recent research (Skagen et al. 1998) from southeastern Arizona suggests that riparian areas are used by migratory birds for resting and foraging and not so much for migration corridors. This recent study found only three species of neotropical migratory birds consistently associated with riparian corridors during the migration period. Individuals of the three species (yellow-breasted chat, summer tanager, and rough-winged swallow) accounted for less than 10 percent of the migrants passing through southeastern Arizona. All other neotropical migratory species were as likely to be found in isolated oases as they were in continuous riparian corridors. Nevertheless, riparian areas, regardless of their size and connectivity, are viewed as important stopover sites for neotropical migratory birds.

Effluent-dependent streams that are dominated by exotic and invasive plant species usually have limited diversity and abundance of terrestrial vertebrate species. Such areas presumably have lower diversity of potential prey species although Rosenberg et al. (1991) report that insects are abundant in summer in pure salt cedar stands along the lower Colorado River in Arizona. They also report that a very small number of bird species appear to use pure salt cedar stands in summer and such habitats are essentially devoid of birdlife in the winter.

3.3 CONCEPTUAL MODEL OF AN EFFLUENT-DEPENDENT WATER STREAM ECOSYSTEM

Stream ecosystems naturally achieve an equilibrium that is closely linked to the physical characteristics in which they exist. The discharge of effluent to naturally intermittent or ephemeral streams represents a discontinuity resulting in a disruption to the natural equilibrium as it exists at the time the discharge begins (**Figure 3-19**). The natural tendency for the created stream ecosystem is to restructure itself so that a new equilibrium is achieved. This restructuring will take some period of time, the length of which will depend on local factors and whether or not additional stressors are placed on the system (e.g., construction of physical structures in, across, or along the stream or effluent flow is increased).

Figure 3-19 illustrates conceptually the expected characteristics of an effluent-dependent waterway downstream of the effluent discharge. Predicted changes in physical, chemical, and biological attributes are based on information gathered during this study. The relative widths of various zones (e.g., zone of physical disequilibrium) are likely to vary from one site to another depending on multiple factors including local geology, wastewater treatment capabilities, and influence of upstream hydrology. It is also important to note that **Figure 3-19** conceptualizes a created stream ecosystem where the discharge represents the only discontinuity. Based on the sites evaluated for this study, this situation does not represent the norm (i.e., most sites had additional stressors present that would influence expectations for stream ecosystem characteristics downstream of the effluent discharge).



Pre-Effluent Discharge

- Flow Regime: Unpredictable, limited to stormwater or some seasonal flow
- Physical: Determined by flow regime, local geology
- Chemical: Local stormwater quality unless seasonally influenced by local runoff
- Biological (Aquatic): Opportunistic, r-selected species; limited diversity
- Biological (Terrestrial): Minimal terrestrial vegetation

Post-Effluent Discharge

- Flow Regime: Natural flow regime superimposed on stable baseflow from effluent discharge
- Physical: Unstable below discharge, typically erosive environment; extent of disequilibrium zone depends on discharge volume, local geology
- Chemical: Stable zone immediately below discharge but oxygen depletion zone downstream; improving water quality downstream. Width of zones depends on influence of upstream flow and effluent quality
- Biological (Aquatic): Opportunistic, r-selected species; limited diversity; dominance of organic pollution tolerant organisms near discharge. With distance downstream increasing aquatic community potential
- Biological (Terrestrial): Increasing riparian vegetation density as stream tends toward physical equilibrium.

Conceptual Model of an Effluent-Dependent Stream Ecosystem. Tendency Toward State of Physical Equilibrium and Increased Biotic Potential Assumes No Additional Anthropogenic Stressors Influence the Created Ecosystem.
Figure 3-19

Longitudinally, created stream ecosystems are still connected to the upstream watershed, meaning that while the effluent flow defines the new baseflow in the created stream ecosystem, the watershed is still subject to impacts from storm flow events. Prior to the discharge of effluent, the contribution of dissolved and suspended material is generally limited to what is imported to a given location during stormwater runoff events. The discharge of effluent changes this irregular input by providing a constant source of dissolved and suspended materials. The characteristics or quality of these materials are directly related to treatment levels.

Naturally flowing streams interact dynamically with the adjacent floodplain, influencing the characteristics and extent of the terrestrial component of the ecosystem. The addition of effluent can change the nature of this relationship by modifying channel morphology. In some cases the change is significant (e.g., in the Santa Fe River and Las Vegas Wash effluent discharge results in immediate channel incision because the effluent volume exceeds the natural bankfull channel forming flow).

With increasing distance downstream of the discharge, the created ecosystem will naturally tend toward the establishment of a new equilibrium based on the baseflow created by the discharge. As the system stabilizes, biological communities have the potential to respond in a positive manner. However, this potential is difficult to define, simply because additional anthropogenic stressors to the created stream ecosystem may exist and it must be kept in mind that the addition of effluent does not change the natural flow regime of the watershed, which may remain flashy and unpredictable.

3.4 EFFLUENT-DEPENDENT WATERS: COMMONALITIES AND DIFFERENCES

While integrating historical and site reconnaissance data from the 10 study areas, a number of features were identified as common to these effluent-dependent streams. These findings are summarized here, but discussed in more detail as special topics in the following section:

- One of the strongest concepts to emerge was the overwhelming impact of physical limitations on effluent-dependent streams. The simple introduction of running water to a streambed is a profound disturbance, the scale of which depends upon several geomorphic and hydrologic conditions at the point of discharge. For example, channel gradient, substrate type, and complexity are inherited from pre-discharge conditions and may not be appropriate for the flow regime created by effluent discharge.

- Physical limitations of effluent-dependent water habitat are determined not just by the physical dynamics resulting from the interaction between the effluent discharge and receiving channel, but by other existing physical limitations imposed on the system by multiple stressors (e.g., channel modifications, bridges, and other sources of discharge).

- The frequency, duration, and location of natural flow (e.g., the natural frequency of storm flows) in the river system likely influence biological expectations, both aquatic and terrestrial. The impact of ephemeral flow is distributed across the floodplain in a complex manner and is difficult to predict for any single event.

- Treatment levels are not necessarily a good predictor of expectations for aquatic community characteristics.

- Except during stormwater runoff or seasonal flow events, in-stream water quality is primarily or entirely a result of effluent quality.

- Riparian terrestrial characteristics are a reflection of the physical template resulting from in-stream flow characteristics (natural and effluent-driven). The distribution of subsurface water, as either hyporheic zone, soil moisture or shallow ground water can determine the structure of the riparian community.

- Terrestrial vegetation and associated wildlife benefit from the creation of effluent-dependent waters, especially where little or no flow occurs upstream of the discharge.

A particularly important finding from this study is that the establishment of perennial flow by discharge of effluent, where none was present before, either naturally or because of dams or diversions, does not automatically result in a river with all the attributes associated with a naturally flowing surface water. The reason for this disparity is complex but the physical disequilibrium or system perturbation caused by the discharge of effluent is precedent to all else.

Given enough time and if no other physical stressors are imposed on the system, a river composed primarily of effluent will ultimately establish a new physical equilibrium. However, because effluent discharge is typically associated with an urban environment, the changes associated with the discharge rarely occur in isolation. Other stressors include physical structures such as bridges and grade control structures, physical modifications such as channel straightening and widening for flood control, and increased peak flows due to increased imperviousness. As these other stressors are imposed upon the stream, it is driven further from a state of equilibrium.

In addition to the physical changes associated with effluent discharge, the influence effluent water quality will have on in-stream water quality can vary depending on several factors including (1) the frequency, duration, and magnitude of upstream flows; (2) other flow sources (e.g., from tributaries or agricultural return flows); and (3) in-stream chemical processes. The nature and dynamics of these factors will vary from site to site. For this study, a comparison of water quality upstream and downstream of effluent discharges suggests that the influence of effluent quality on in-stream water quality varies depending on site-specific factors. These factors are discussed in Section 3.5.

The aquatic community of any stream is dependent on the physical and chemical characteristics of the surface water. This is true of all surface waters regardless of whether the stream is naturally flowing or created. As indicated above, the physical template established by the discharge of effluent, often coupled with other physical stressors associated with an urban environment, can result in an unstable environment. This finding alone places limitations on aquatic community expectations. Superimposed on this physical limitation are site-specific water quality factors.

Of particular interest to this study was evaluating the assumption that as the quality of effluent "improves" the aquatic community will likewise "improve." This assumption might be valid if all else is equal (i.e., the only factor limiting the aquatic community is effluent quality). However, the results from the 10 case studies strongly suggest that this assumption may be invalid. Other factors, such as habitat, appear to be limiting the aquatic community.

This study also found that the introduction of water into normally dry stream channels can create important terrestrial habitat benefits for wildlife species. Riparian systems that develop as a result of wastewater discharged into normally dry channels may stand in stark contrast to the adjacent upland vegetation that is not influenced by discharge. In addition, the terrestrial community downstream of the discharge point can be distinctly different from the terrestrial community upstream of the discharge. This distinction is greatest where there is little or no flow upstream of the discharge.

The issue of habitat versus water quality as the limiting factor is certainly not new. However, this study suggests that habitat may be a more significant issue than originally believed. The terms "habitat" and "water quality" are broad terms, which encompass numerous specific factors(e.g., substrate type and complexity, sedimentation, dissolved oxygen and temperature), many of which singly or in tandem can influence aquatic community potential. It may be very difficult to determine what specific factor serves as the keystone on which the aquatic community responds. Moreover, it is quite likely that this keystone will be site-specific, differing from stream to stream.

3.5 EFFLUENT-DEPENDENT WATERS: DISCUSSION OF SPECIFIC OBSERVATIONS

The previous sections have provided a description of effluent-dependent waters as a distinct type of stream ecosystem. Physical, chemical and biological observations from the 10 study areas have been presented and summarized. The following discussion focuses on a set of scientific issues that emerged from the Discharger Survey and informal discussions with researchers and stakeholders throughout the arid West. While the investigation did not set out to completely resolve any of these questions, the investigators did attempt to examine the applicability of the 10 study areas to these issues. The questions included the following:

- How much does the hydrological and physical template control the ecological health of an effluent-dependent stream?
- How do hardness and alkalinity vary across the arid West and how do they compare to EPA wastewater toxicity tests?
- Is taxonomic richness low in effluent-dependent streams of the arid West?
- Are there measurable improvements in habitat due to the increased in-stream flows contributed by effluent?
- Is there a relationship between wastewater treatment upgrades and improved habitat in effluent-dependent streams?

- Are existing EPA aquatic habitat assessment protocols appropriate for effluent-dependent streams in the arid West?

The following sections examine aspects of these questions using key observations of the Habitat Characterization Study as illustrations. In **Chapter 8**, specific research recommendations have been developed to support additional studies on these questions.

3.5.1 Hydrological and Physical Template

3.5.1.1 Overview

Because water use by a landscape is controlled by its shape, geology, and the climate (Dunne and Leopold 1978), the environment must work within these limitations to create habitat. These limitations might include the dynamics of all natural and created watercourses, amount and timing of runoff and stream hydraulics, movement of sediment by runoff and stream flow, movement and storage of water in both the deep and shallow subsurface, and ability of the soil to hold moisture for plants. Each of these factors is important to the creation of both aquatic and riparian ecosystems and each is itself a function of arid climatic conditions.

Riparian systems are even more sensitive to climate and other physical limitations than aquatic systems. This is because riparian communities are transition zones, or *ecotones*, between the aquatic ecosystem of the stream and terrestrial ecosystem of the upland banks (Gregory et al. 1991). The size and sharpness of the transition is dependent upon the flow of energy from the linear and rapidly changing stream channel to the broad hill slopes, terraces, and aquifers of the extensive upland terrain (Mitsch and Gosselink 1993).

Plant and animal communities that rely upon the riparian transition zone must adapt to a changing flow of energy and matter. Because of this highly dynamic environment, riparian communities are frequently "patchy" (i.e., not continuous) as favorable habitat grows and contracts with the variability of climate, stream flow, and the other factors listed previously. Patchiness results in a string of riparian zones of varied size and usefulness aligned along the river, a string of refuge points as opposed to a continuous corridor.

Change of both the aquatic and riparian environments can occur quickly in the form of disturbance (flood, fire, human intervention, and other factors) to this physical template. Disturbances that involve rapid swings in physical conditions are called "harsh" and the ecological communities that use these environments are different than the residents of more "benign," or less disturbed systems. Studies of harsh environments report that these environments have lower species richness and simple food chains. Plant and animal life must be resilient, relative to more benign systems. Many taxa will have increased drought or flooding resilience, depending upon the individual ecosystem.

3.5.1.2 Hydrologic Template

Streams of the arid West are generally within this harsh group of aquatic environments. Because the location of precipitation is more influenced by steep mountain ranges and the total rainfall is low, stream flows might be infrequent for many years. When rain finally comes, streams begin flowing suddenly and at abruptly rising velocities and discharges. Channels may become abandoned from one storm to the next and new channels are rapidly cut into the existing floodplain.

There is no universally accepted criterion for the "flashiness" of a stream system. One criterion used by the Habitat Characterization Study is based upon the mean monthly flow (**Figure 3-20**) taken from USGS stream gaging up and downstream of 8 of the 10 WWTPs visited by the study. Data for the Salt and Gila Rivers and Carrizo Creek could not be used for this analysis because of reasons discussed in **Appendix D**. For comparison purposes, four non-arid streams are included on **Figure 3-20**. There are no obvious trends; however, most of the arid streams have a steep summer peak in-stream flow, relative to the four non-arid study streams.

The ratio of the mean annual flow to the mean maximum flow is one indicator of the magnitude of flood discharges experienced by the 10 reaches (**Figure 3-21**; shown individually on **Figures 3-22 through 3-30**). For times that the streams are flowing, the mean annual discharge is many times smaller than the mean annual peak flood for the 10 arid watersheds. Discharge data for the four non-arid study areas do not show as much contrast in peak flows. If the arid streams were more subject to abrupt changes in discharge (more flashy), annual peaks would be expected to be much greater than mean stream flows.

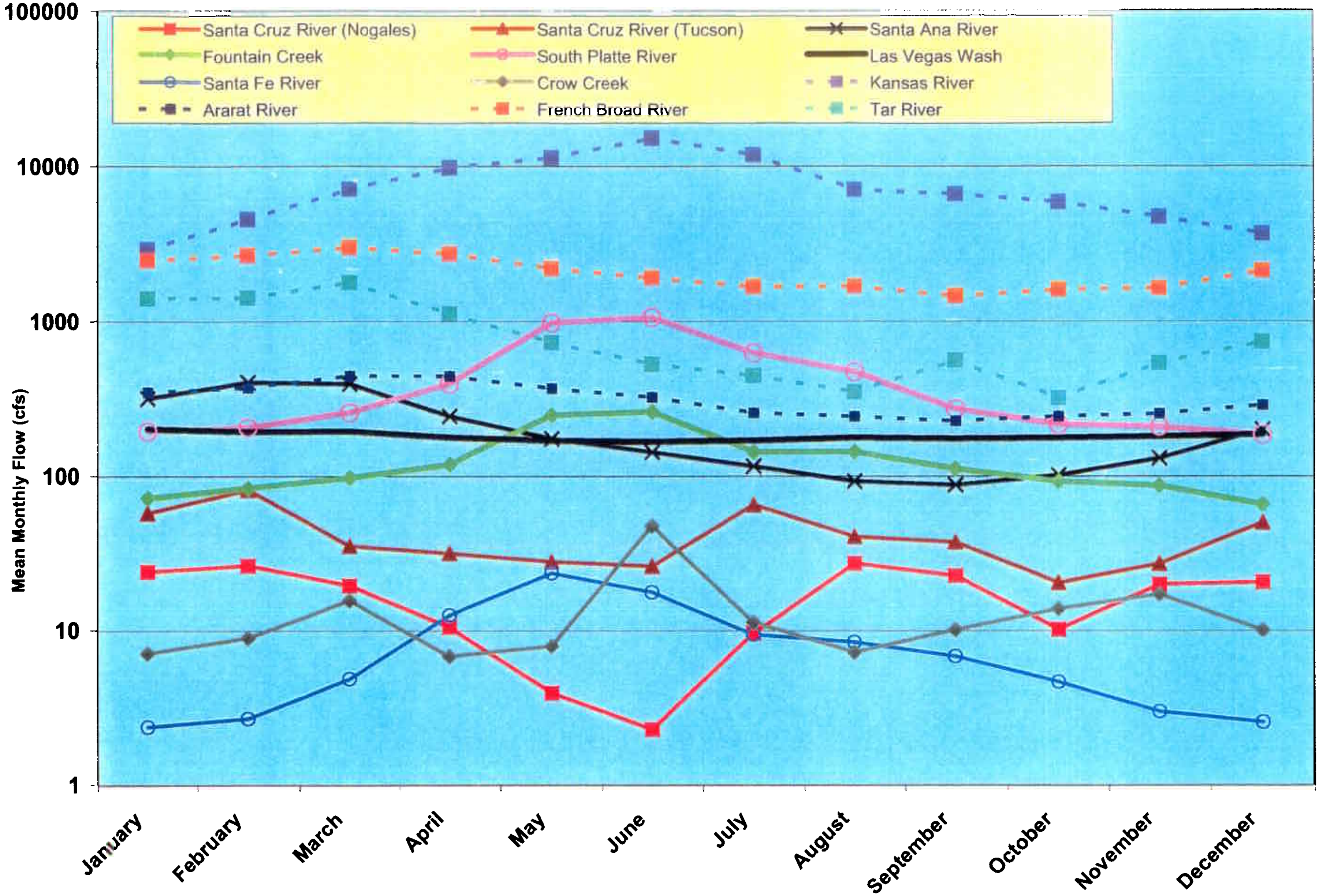
3.5.1.3 Biological Response to Hydrologic Template

Many of the fauna found in arid West streams have adapted to extreme disturbance with changes in behavior or physiology. For example, macroinvertabrate species in flashy streams progress to adulthood more rapidly and have longer reproductive periods than similar species in more benign environments. Research suggests that arid West ecosystems can recover in about two months (**Appendix I**). This recovery period can be from 7 to 52 days, depending upon the frequency and season of the disturbance. If the disturbance is followed by a drought, recovery may be slower.

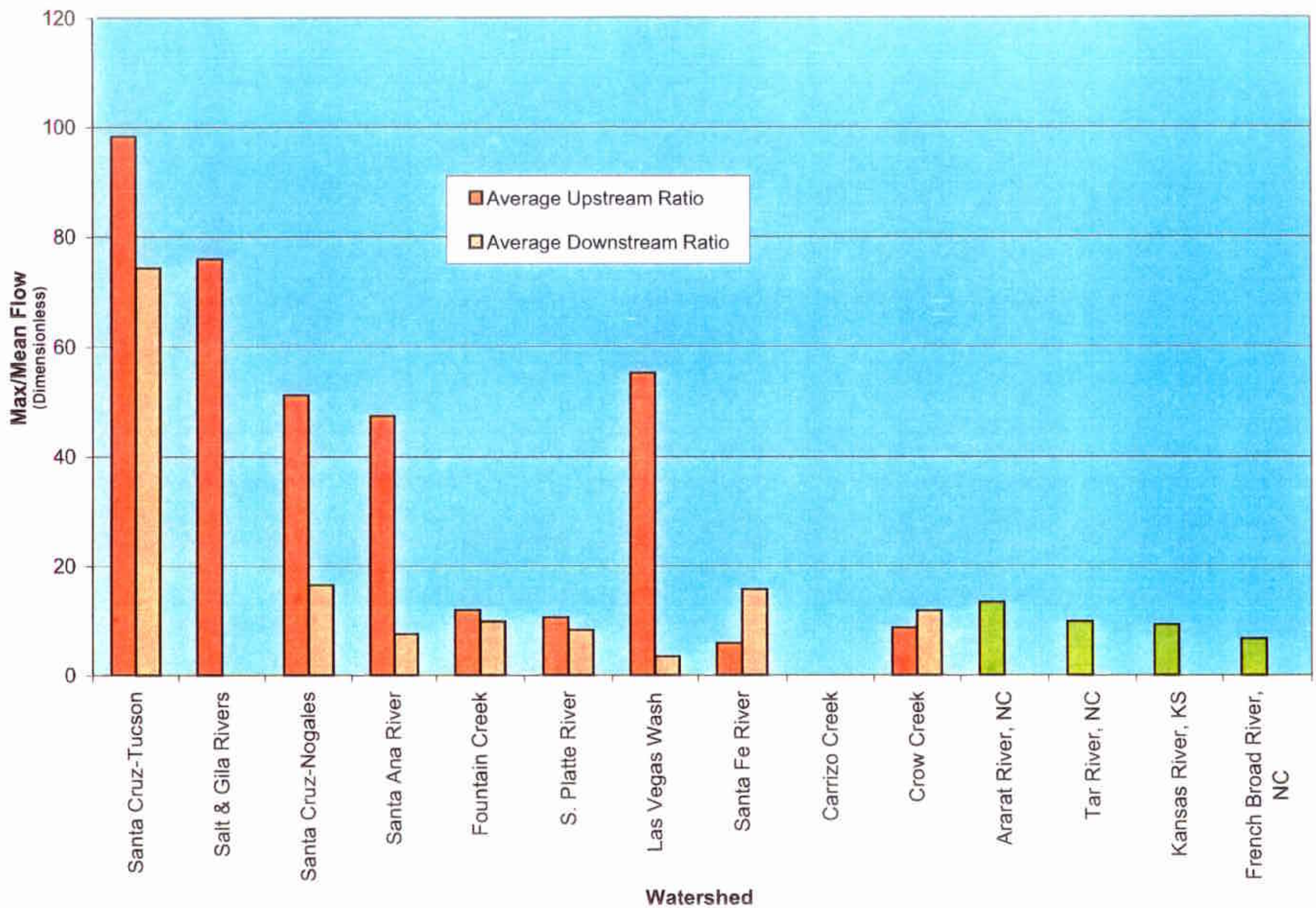
Periphyton (algae and other simple, attached plants) communities appear to be the most easily disturbed by the shear stress and scouring of flood flows. But depending upon the season, recovery times appear to be no greater than a month. Algae have very effective survival strategies, including flexible life histories, with rapid ability to spread and re-generate. Studies in the arid West indicate that recovery can range between five days and up to three months, depending on the season and frequency of flood events (**Appendix I**).

The impact of flooding on riparian plant communities is not as well described in scientific literature. The most complete study on an arid stream was conducted on the Hassayampa River near Wickenburg by Stromberg and co-workers (Stromberg et al. 1991, 1997; Stromberg 1997). This study found that the return of individual species, for example willow, cottonwood, and salt cedar, depended in a complex manner on several factors. These factors included the flood depth and duration, ability of the soil to retain moisture and seasonal timing of the flood.

In addition to these direct effects, floods can also impact riparian zones by simply changing the location of the channel. Many ephemeral streams have channels that shift back and forth across the floodplain each time they flow. The location of the channel at any particular time may only reflect the centerline of the last flood. Over longer time periods, the riparian community may cluster around the most frequent location of flood flows.

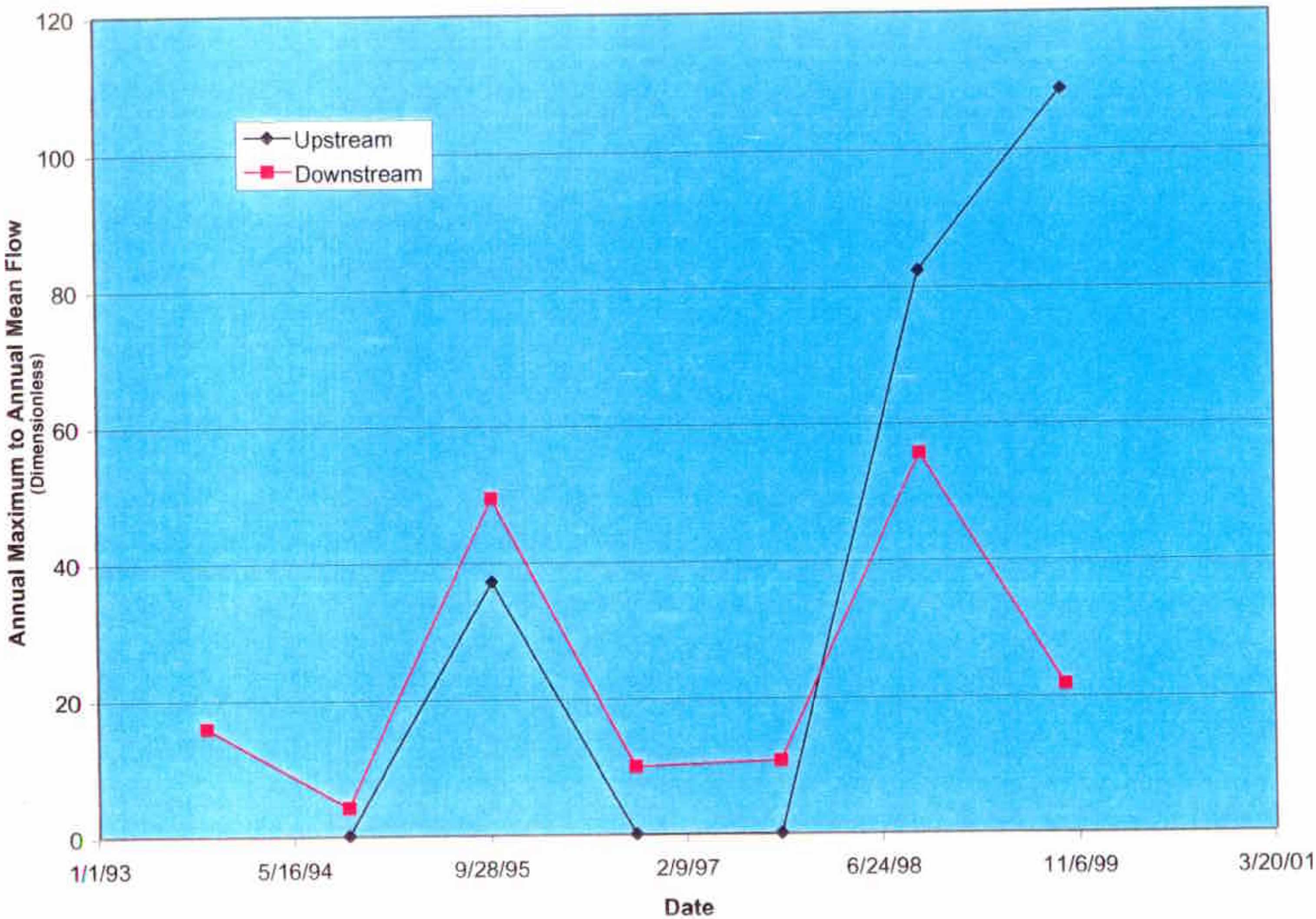


Mean Monthly Streamflow
Downstream of WWTP Outfalls
Figure 3-20

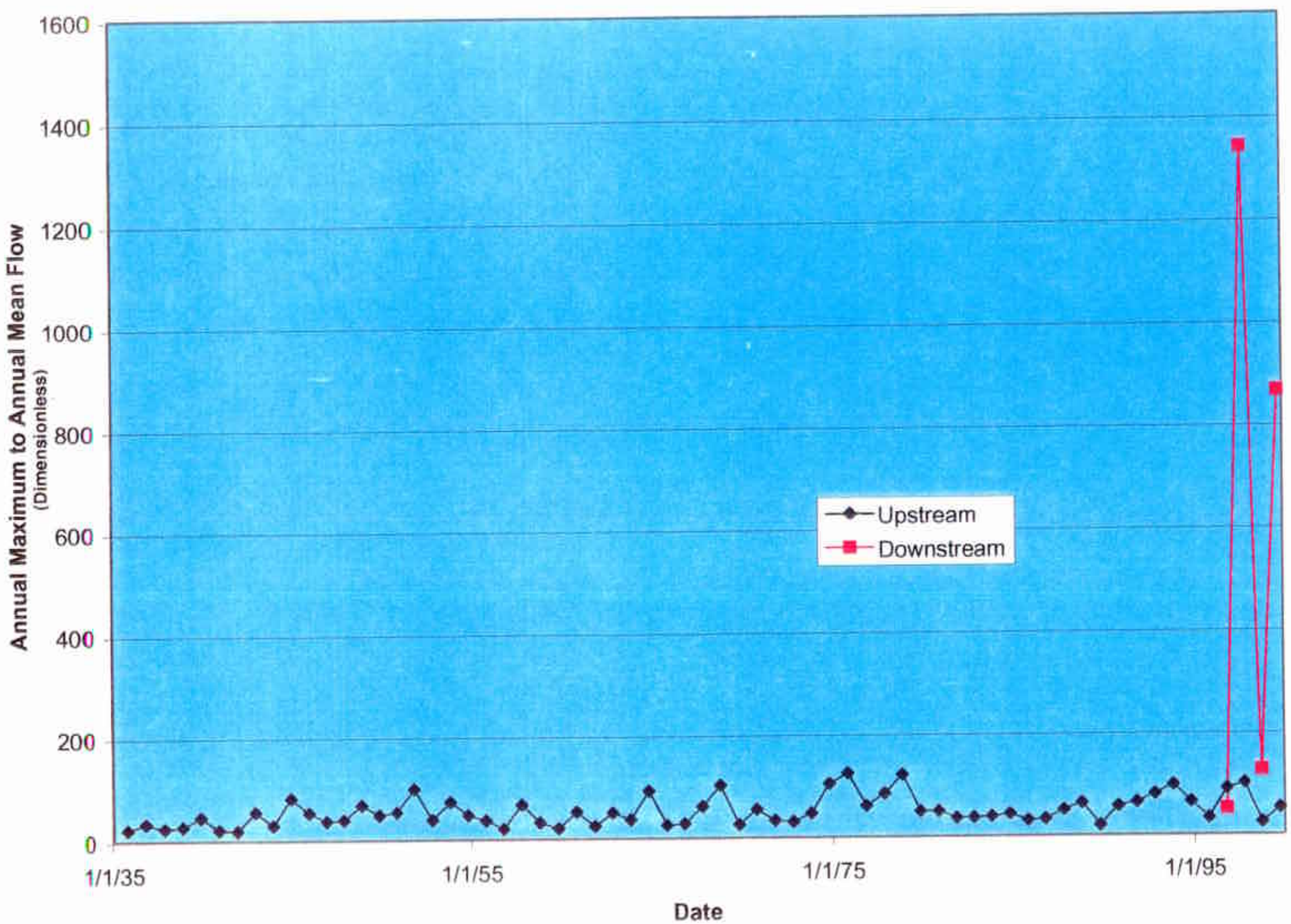


Ratio of Annual Maximum to Annual Mean Stream Flow Upstream and Downstream of WWTP Outfalls

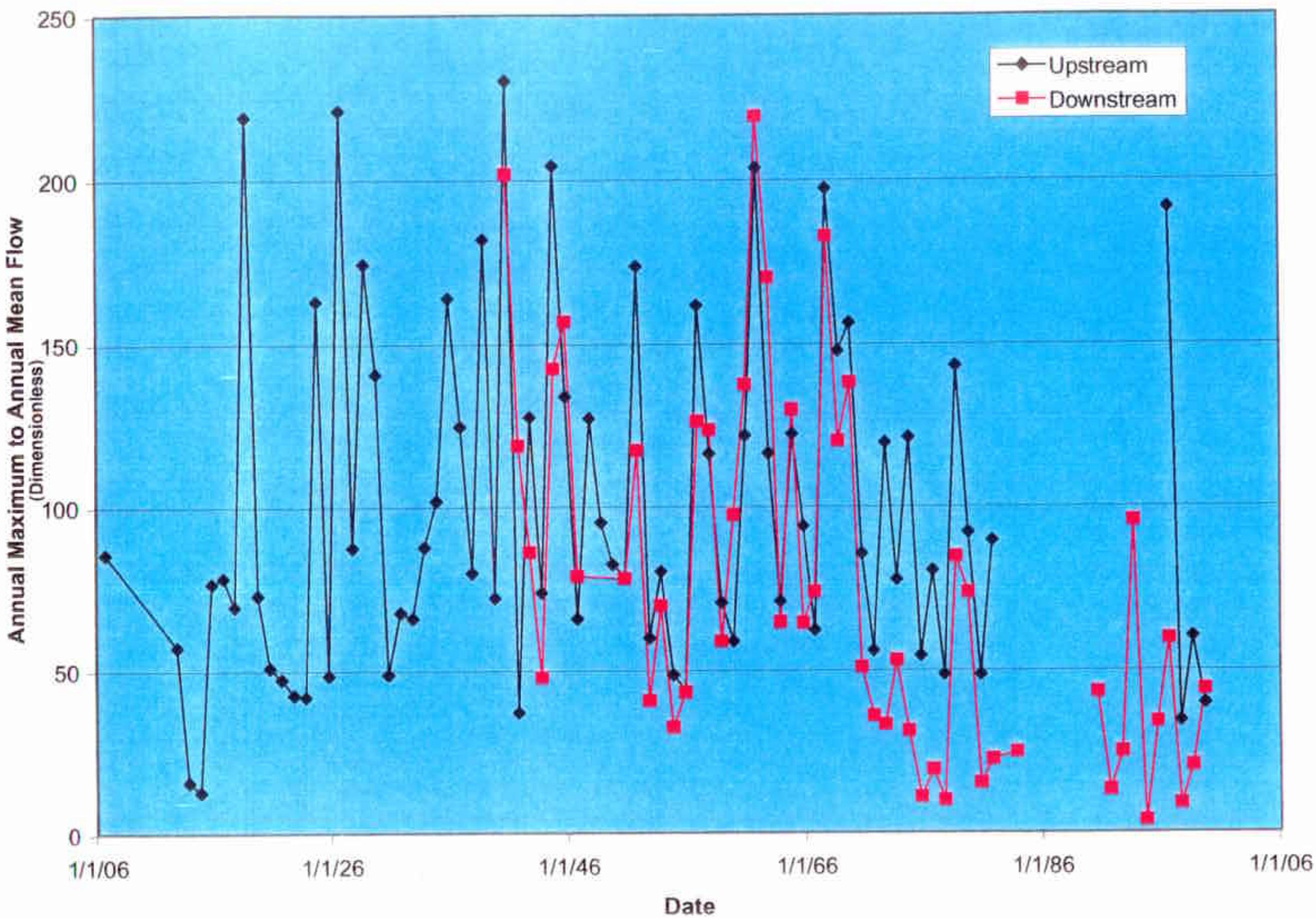
Figure 3-21



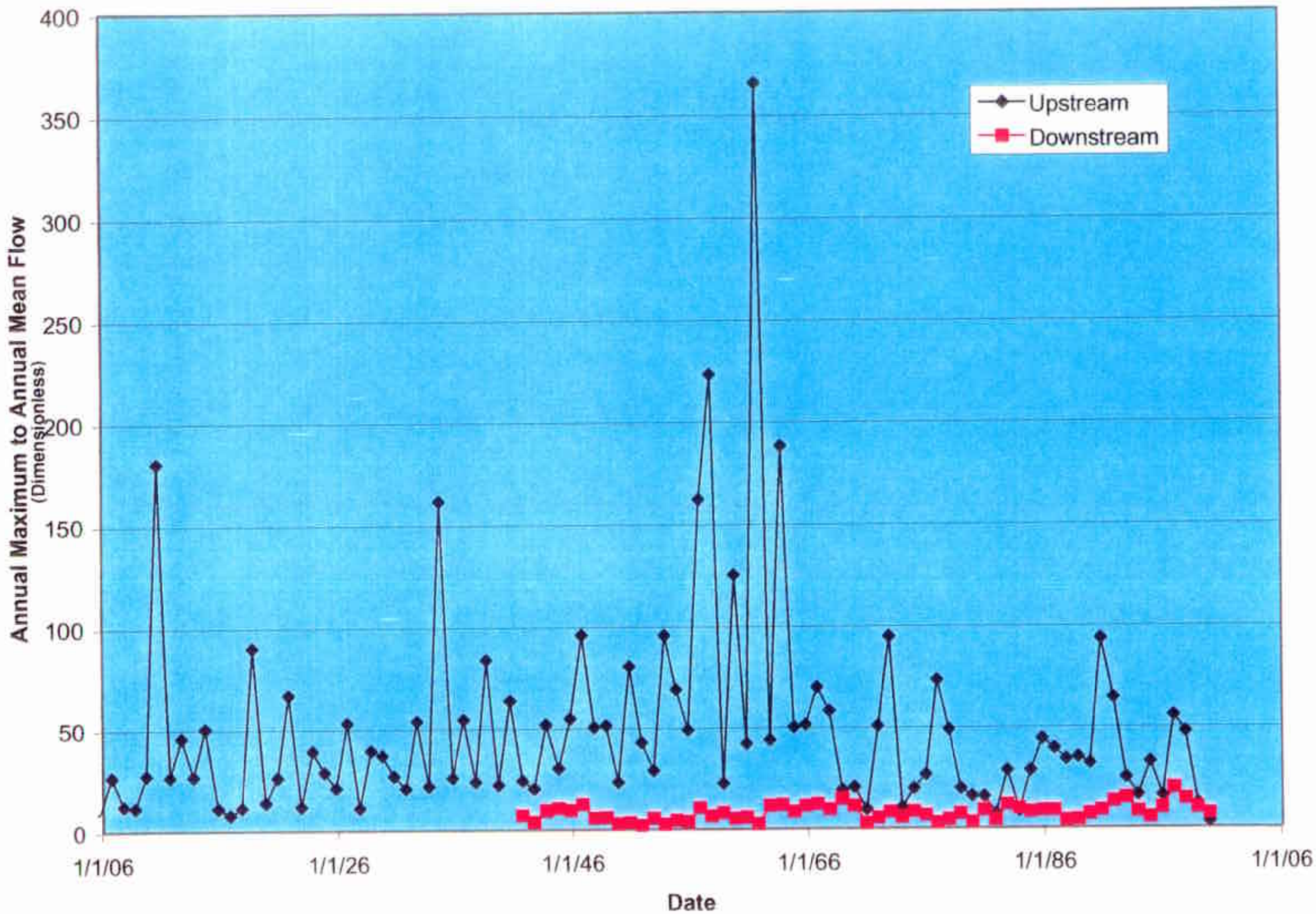
Annual Maximum to Annual Mean Flow
Salt/Gila River near Phoenix, AZ
Figure 3-22



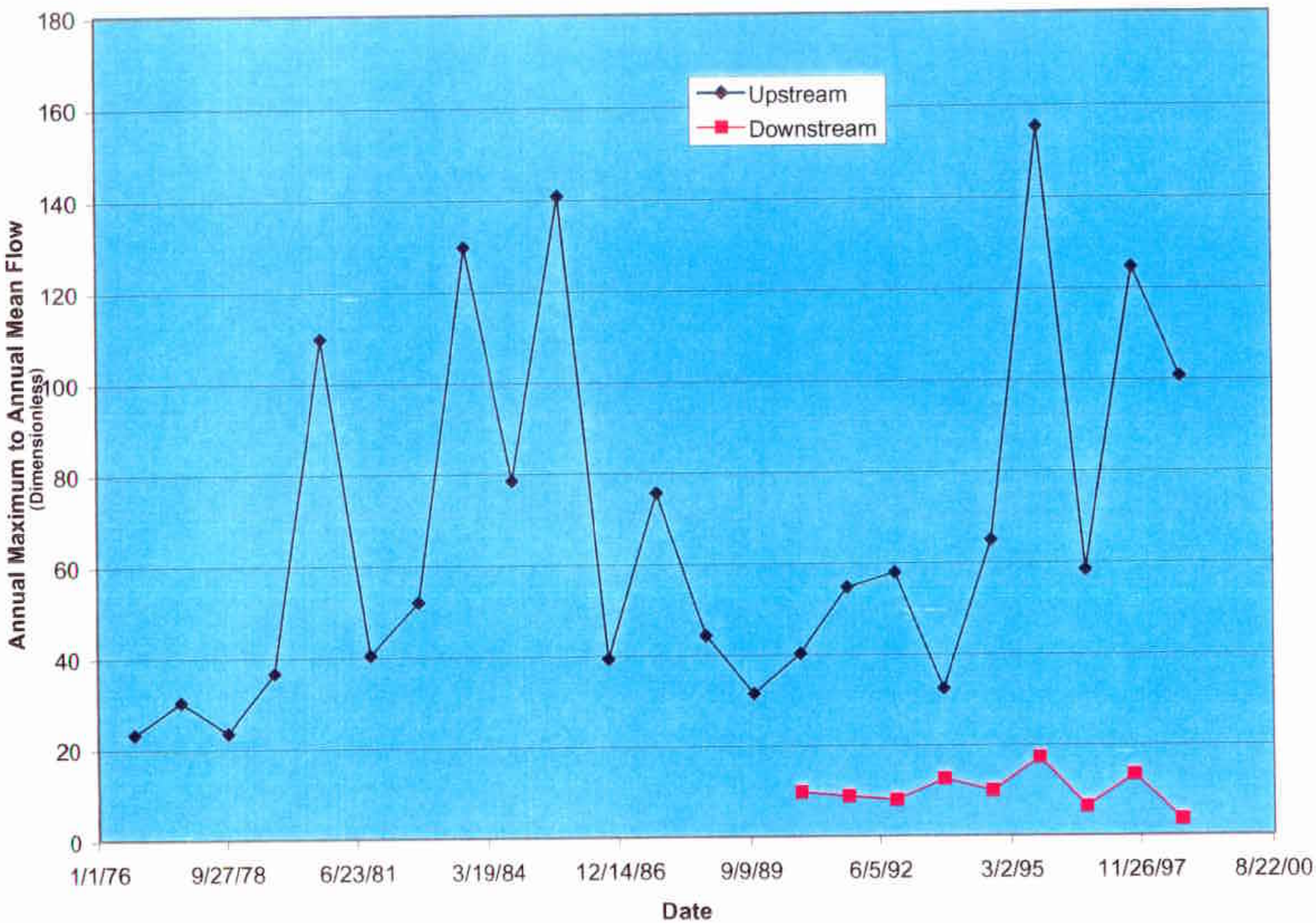
Annual Maximum to Annual Mean Flow
Santa Cruz River near Nogales, AZ
Figure 3-23



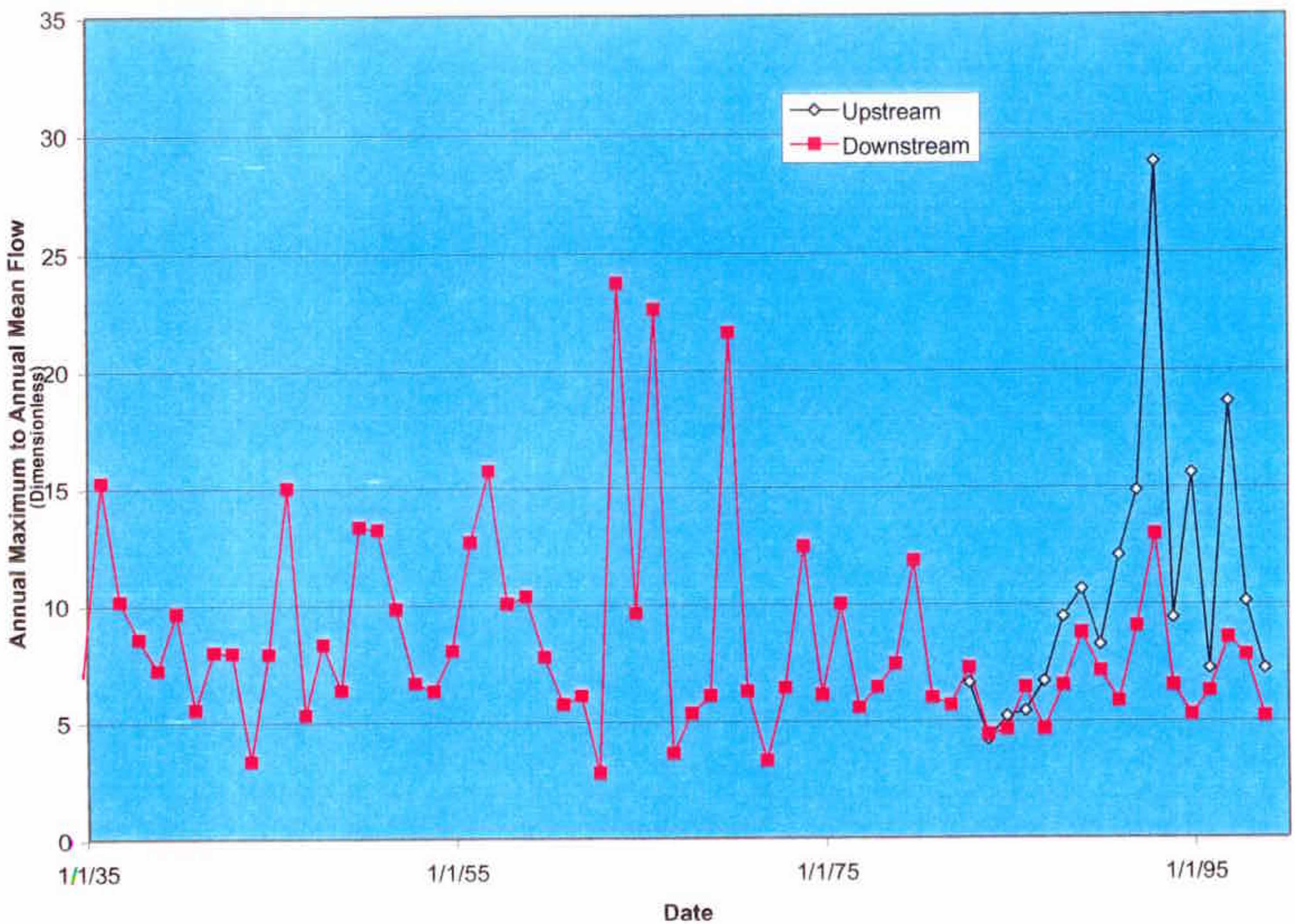
Annual Maximum to Annual Mean Flow
Santa Cruz River near Tucson, AZ
Figure 3-24



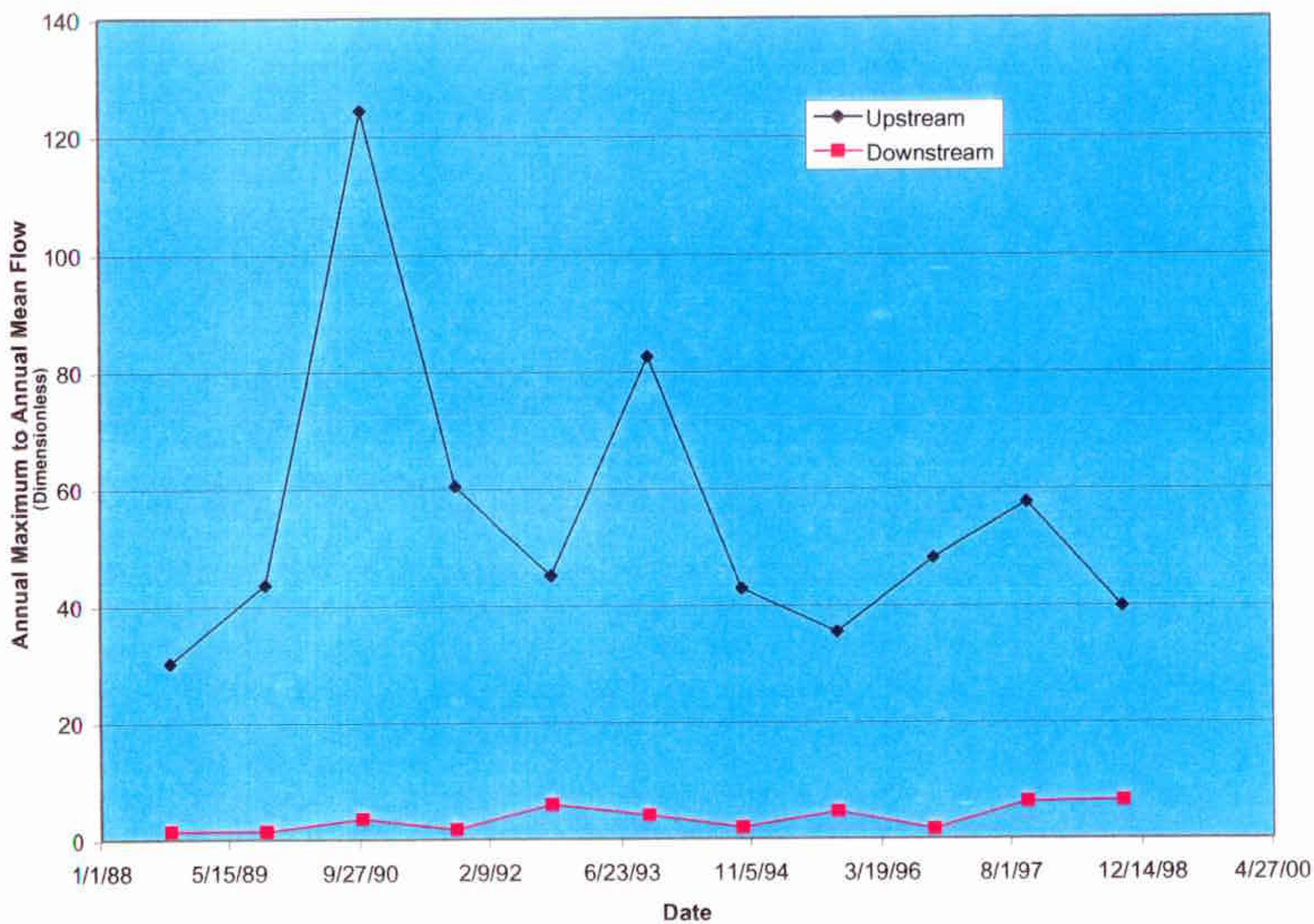
Annual Maximum to Annual Mean Flow
Santa Ana River near San Bernardino, CA
Figure 3-25



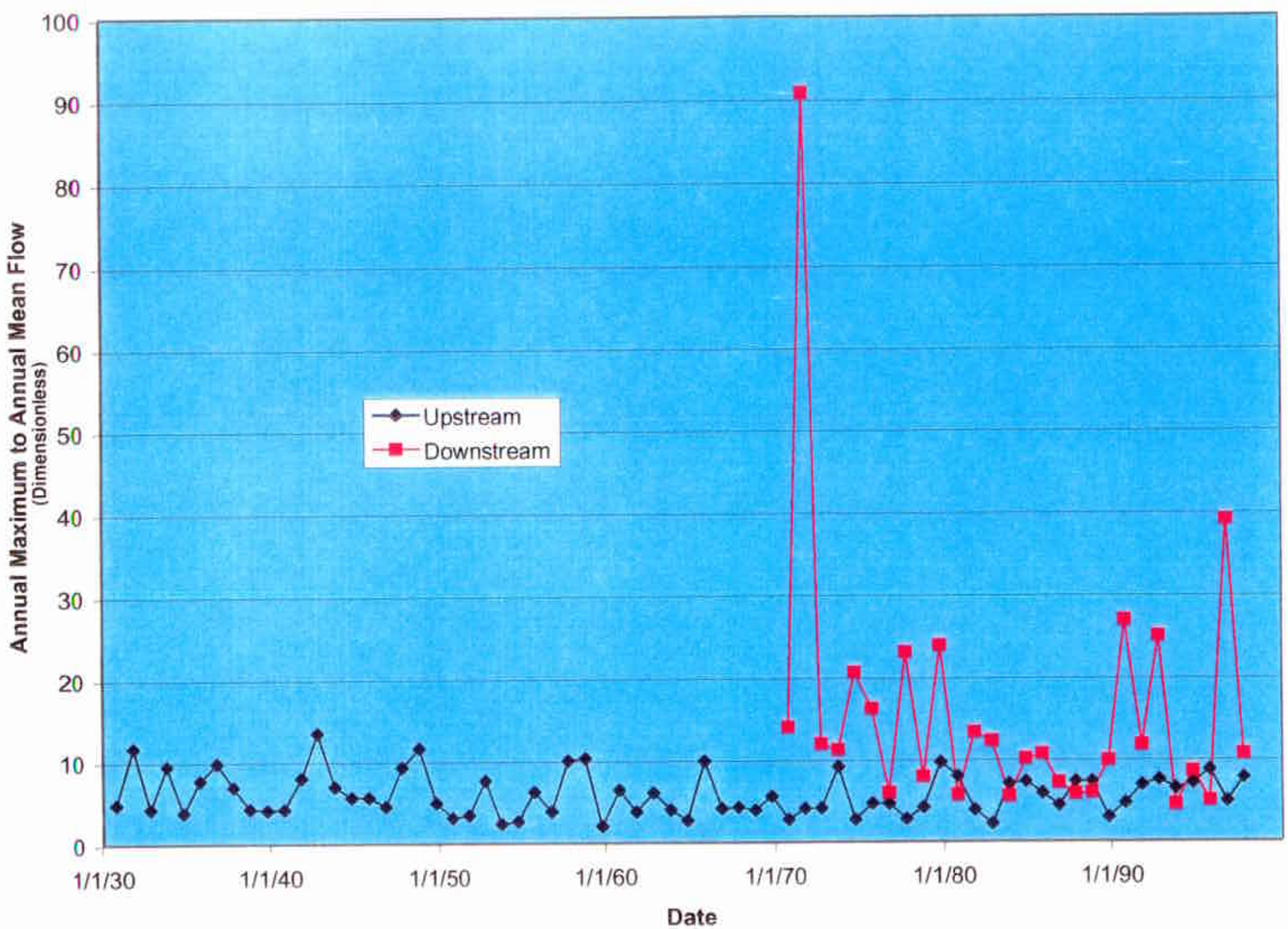
Annual Maximum to Annual Mean Flow
Fountain Creek near Colorado Springs, CO
Figure 3-26



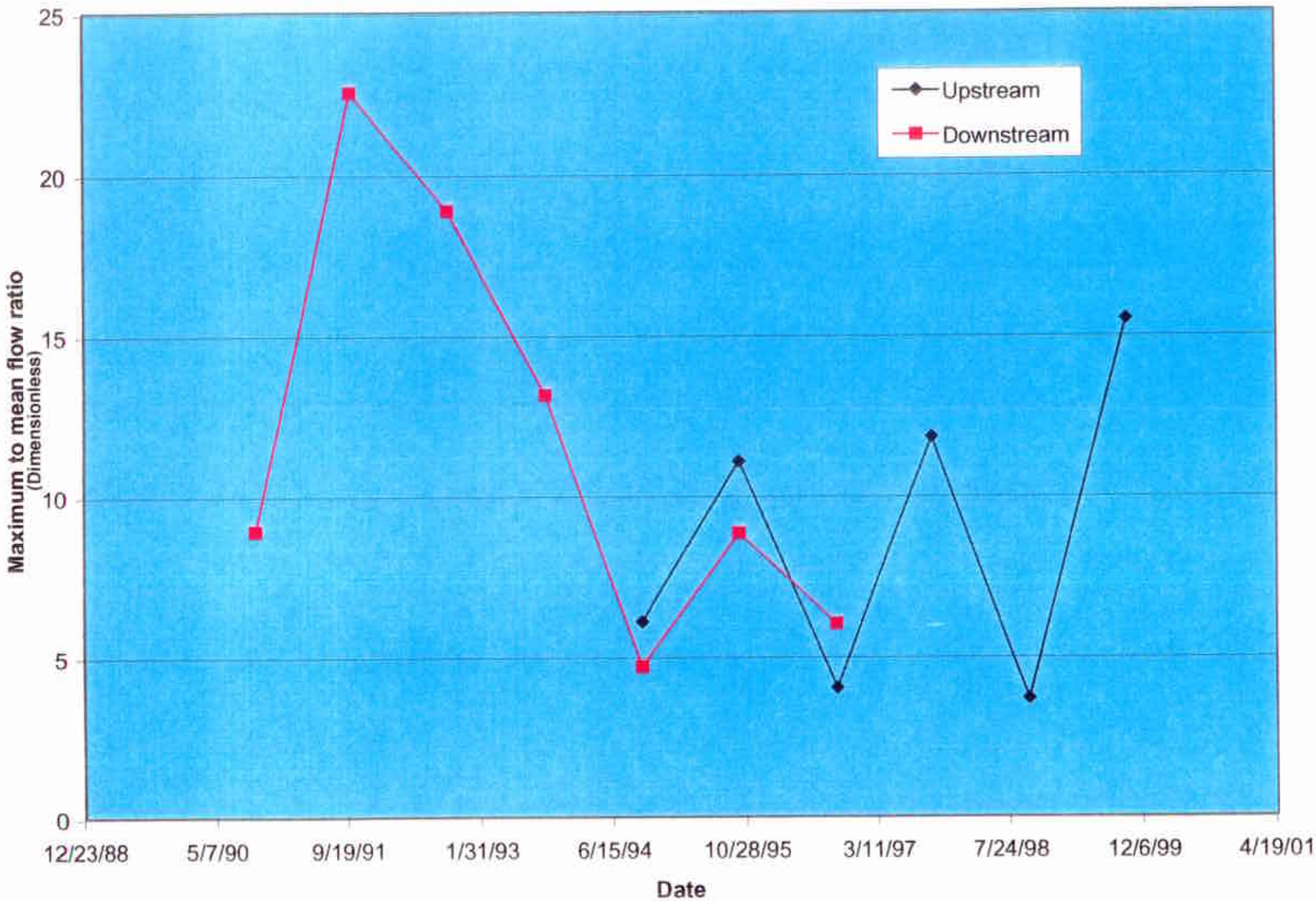
Annual Maximum to Annual Mean Flow
South Platte River near Denver, CO
Figure 3-27



Annual Maximum to Annual Mean Flow
Las Vegas Wash near Las Vegas, NV
Figure 3-28



Annual Maximum to Annual Mean Flow
Santa Fe River near Santa Fe, NM
Figure 3-29



Annual Maximum to Annual Mean Flow
Crow Creek near Cheyenne, WY
Figure 3-30

3.5.1.4 Physical Template

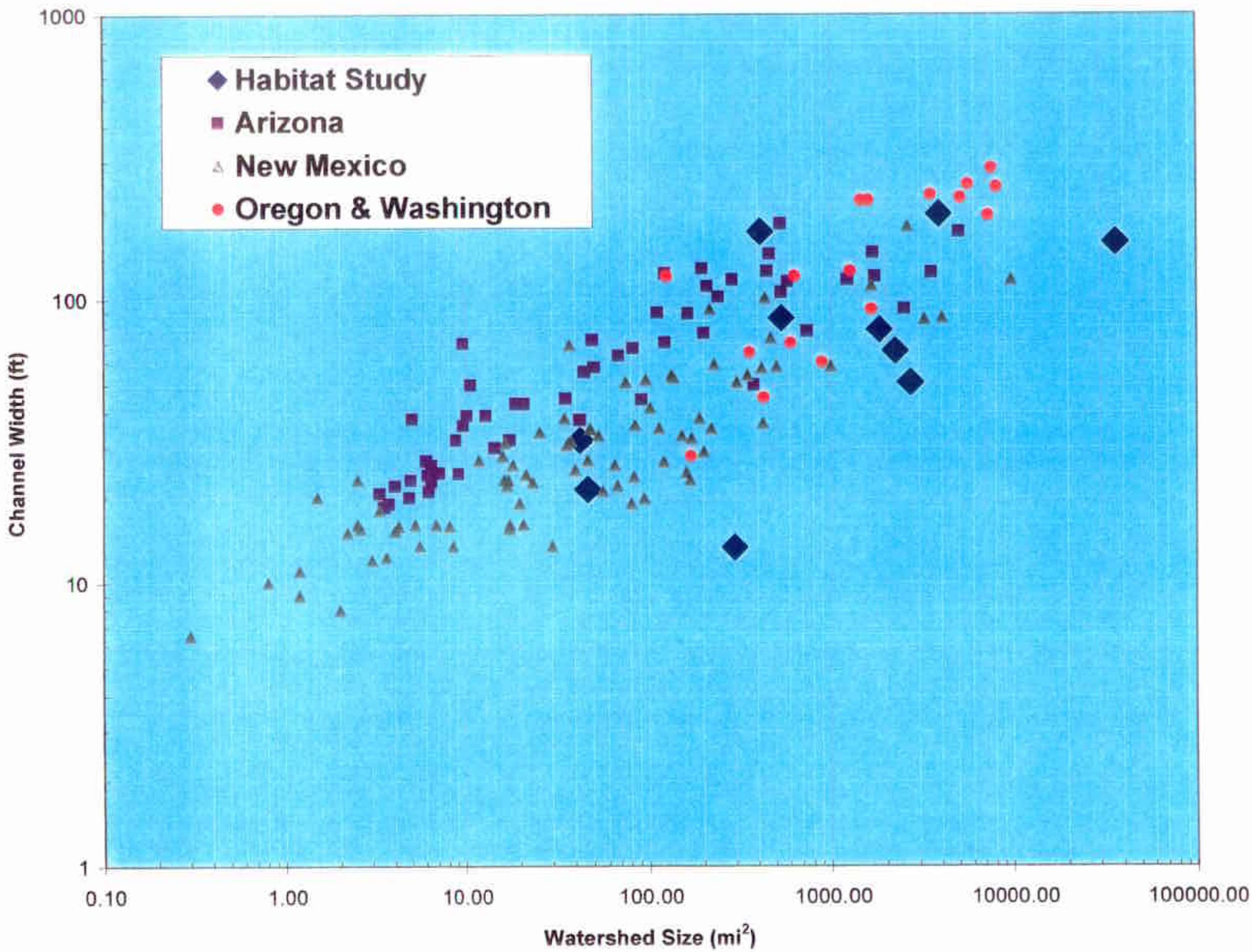
In many of the 10 study areas of the Habitat Characterization Study (e.g., Santa Cruz River and Santa Ana River), the channel used by the effluent discharge has been fixed in place, relative to the upstream reaches. By discharging a steady flow of sediment-free effluent, most of the effluent flows have eroded down into the floodplain and formed entrenched channels. For example, the Santa Fe River directly below the effluent discharge point is clear and immediately begins to cut a new channel into the river bed. Within a mile downstream, the water becomes cloudy as sediment eroded from the channel becomes suspended in the stream. Eventually, the downcutting ends as the new stream adjusts to the existing, larger channel.

Any natural stream channel with a mobile bed represents a balance between the energy used for erosion and movement of the bed and the gravitational settling of the transported material. For a perennial stream, the shape of the channel will represent the most frequent flood event that maintains this balance over time, the so-called "channel-maintaining" or bankfull flow (Andrews 1980; Emmet 1975; Leopold et al. 1963; Schumm 1956). For an ephemeral flow, water in the stream is only an occasional event and it is not clear what defines a channel-maintaining flow in such a system. For an effluent discharge channel that is protected from floods, once again, the channel-maintaining flow may simply be the average effluent discharge released by the WWTP.

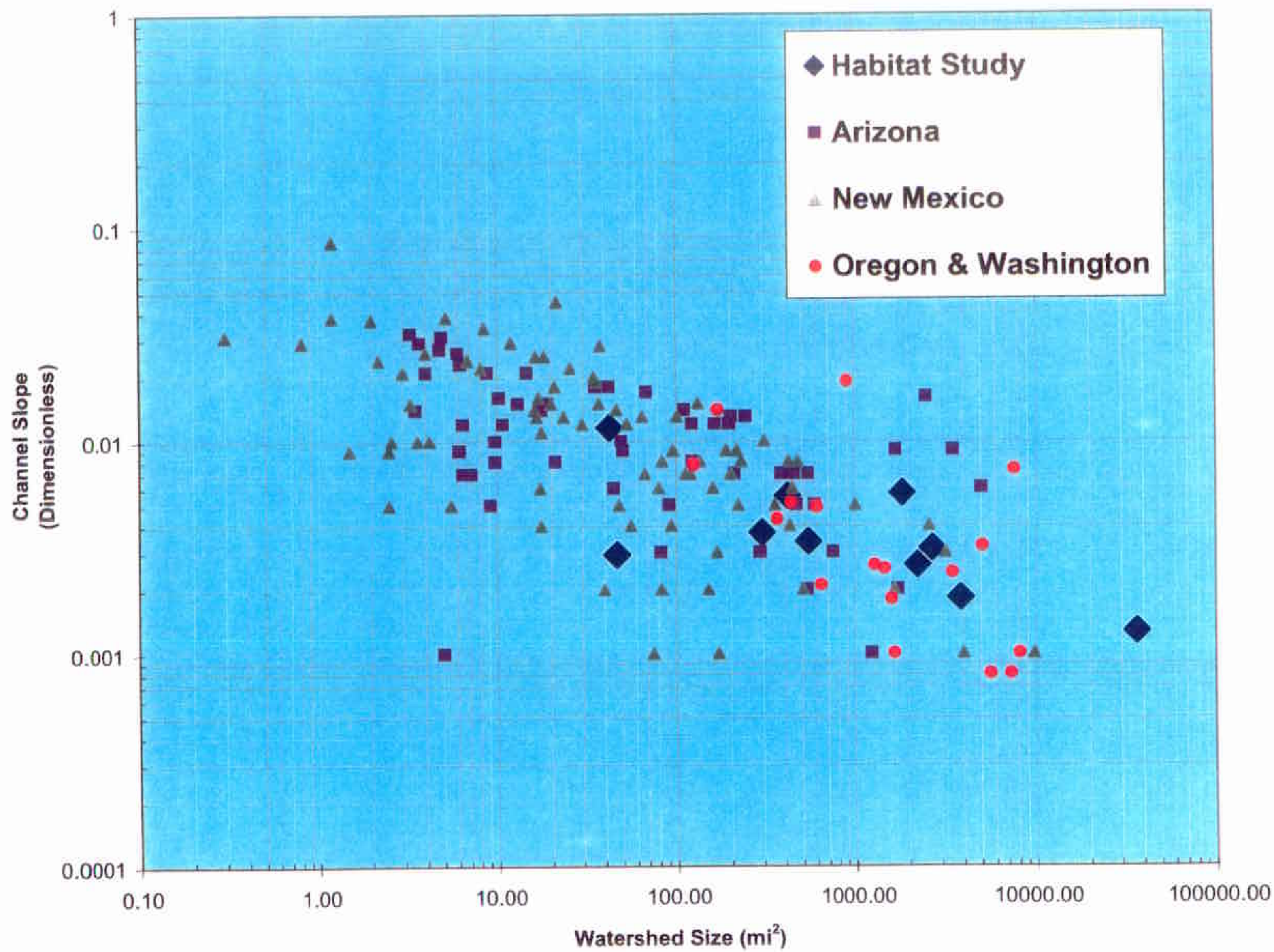
For perennial streams, the comparison of stream shape and watershed size to the long-term record of floods has determined that there is a relationship between these geomorphological variables and the size of the channel-forming flood. Regression analysis of these records for many perennial rivers has suggested that, with minor local variation, it is the 1.5-year flood that is best represented by the shape and size of the active channel. In other words, the discharge that occurs about every 1.5 years is usually about equal to bankfull flow.

If this concept extends to ephemeral rivers, it is not entirely clear how it works. However, Moody (2000) has found that many intermittent to ephemeral rivers in the arid West do follow the "1.5-year recurrence interval" rule. Despite this, the Habitat Characterization Study found clear differences between the bankfull flow of perennial rivers and the 1.5-year recurrence interval of the 10 effluent-dependent streams (**Figures 3-31 and 3-32**). Using the data of Moody (2000) and the arid watersheds included in Castro and Jackson (2001), the 10 watersheds are similar in gradient to the other streams but are slightly smaller in width than a perennial stream of equal watershed area. Several regression lines were fit to the Habitat Characterization Study data (**Figure 3-33**) and various subsets of the Moody and Castro and Jackson datasets. This suggests that the habitat data are most similar to the New Mexico streams, perhaps reflecting the wide variety of streams, from the Chihuahuan Desert to the southern Rocky Mountain alpine ecoregions of the state.

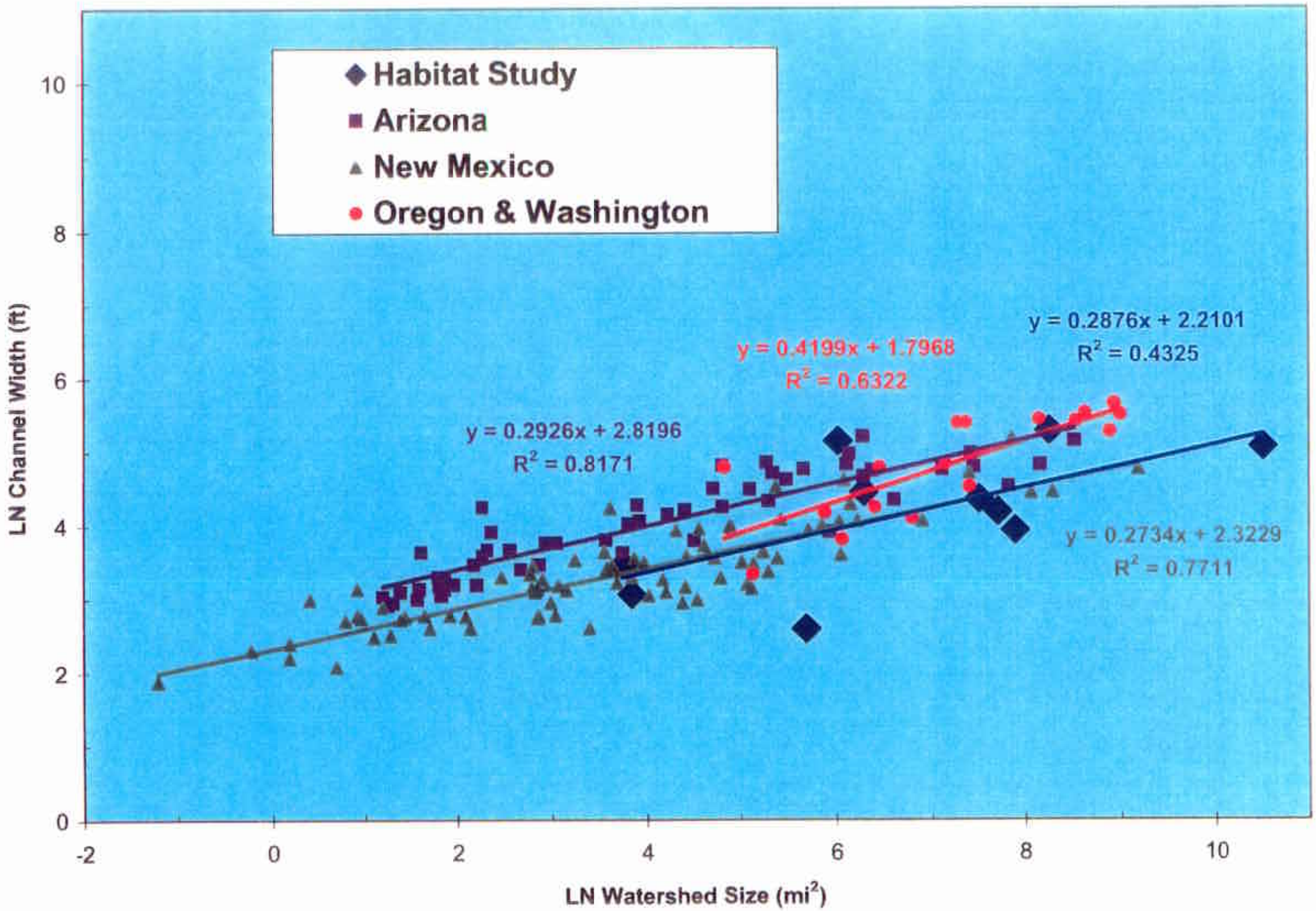
The 1.5-year interval flood flows were calculated for each of the 10 study areas. These discharges were compared to the various WWTP discharges, summed in cases where there is more than one plant (**Table 3-2**). In most cases, the effluent discharge is much lower than the 1.5-year flood. This could suggest that the effluent stream is too small to change the form of the river channel and that the habitat will hydrologically remain similar to an ephemeral stream.



Measured Geomorphic Parameters for Habitat Characterization Areas vs. Other Western Streams
Figure 3-31



Measured Geomorphic Parameters for Habitat Characterization Areas vs. Other Western Streams
Figure 3-32



Linear Regression Curves for Several Western Stream Databases
Figure 3-33

Table 3-2
Comparision of 1.5-Return-Interval Discharge to Introduced Effluent Flow

Stream	1.5-year Flood (cfs)	Total Effluent Flow (cfs)
Salt River	9,373	92.60
Santa Cruz River – Nogales	1,781.9	10.923
Santa Cruz River – Tucson	1,677.3	46.99
Santa Ana River	380.6	62.70
Fountain Creek	2,846.4	38.50
South Platte River	3,069.3	149.00
Las Vegas Wash	105.91	134.05
Santa Fe River	681.1	5.494
Crow Creek	40.00	9.966

In the case where the effluent discharge is much larger than the 1.5-year flood, there may be more geomorphological resemblance to a perennial stream. In the case where the two discharges are about equal, it may be difficult to predict what sort of channel will evolve, being stable one year and migrating across the floodplain the next. It is not clear how this variability might affect the habitat that is established. Such a system might be under constant disturbance.

Stream systems that experience rapidly recurring natural disturbance might be predisposed to more resistance against anthropogenic disturbance; however, not all disturbances would be expected to be equivalent. The data developed from the Habitat Characterization Study and **Appendix I** do suggest that the three-year recovery period that is used in water quality regulations might be longer than the time frame for recovery seen naturally in arid West streams. Further, the recurrence interval for floods within the existing ephemeral drainage must be compatible with the recovery period applied to the effluent-dependent system. Further hydrological and geomorphological research into this question is needed.

3.5.2 Wastewater Treatment and the Aquatic Community

Site reconnaissance and historical data suggest that improvements in wastewater treatment may yield only limited improvements in the aquatic community, especially with regard to taxonomic richness. 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 provision is available for in-stream dilution). As a consequence, the effluent limitations incorporated into NPDES permits are typically equivalent to the water quality standard. In most instances, the most stringent water quality criteria established for the protection of arid West streams are those established to protect aquatic life. 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 nationally recommended criteria are used), then one should expect that wastewater treatment improvements should result in improvements to the aquatic community (e.g., species richness or composition, resident downstream of an effluent discharge).

As appropriate, given data availability, case study data were evaluated in two ways in relation to the level of wastewater treatment: (1) site reconnaissance data were compared upstream and downstream of the effluent discharge; and (2) historical data were evaluated to document long-term changes in aquatic community characteristics 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 (Table 3-3). At the lowest levels of treatment, with chlorination but no dechlorination, there was a sharp decline in taxonomic richness between the sites above and below the effluent discharge. However, at higher levels of treatment, both increases and decreases in taxonomic richness occurred. In some cases it appeared that changes in richness could be more related to changes in habitat quality than chemical quality (Table 3-3). Taxonomic composition varied somewhat with increased levels of treatment, especially at the highest level of treatment (i.e., chlorination with dechlorination, nitrification, and denitrification, and filtration). Sites with this high level of treatment had increased abundance of "cleanwater taxa," so-called EPT organisms. However, it should be noted that these "cleanwater" taxa were often limited to or dominated by baetid mayflies. Other EPT taxa were generally absent (Table 3-4).

Table 3-3
Comparison of Habitat Quality, Taxonomic Richness, and Percent Cleanwater Taxa Upstream and Downstream of Effluent Discharge at 10 Study Areas

Treatment Level		Habitat Score		Taxonomic Richness		Percent Cleanwater Taxa	
		Above	Below	Above	Below	Above	Below
Higher Quality Effluent	Chlorination only	131	143	16	2	8.99	0
	Chlorination only	160	143	15	5	1.15	0
	Chlorination with dechlorination	26	62	0	2	0	0
	Chlorination with dechlorination; Filters	56	84	0	9	0	0
	Chlorination/dechlorination; Nitrification/denitrification	103	72	9	6	1.4	0
	Chlorination/dechlorination; Nitrification/denitrification	53	156	0	5	0	0.02
	Chlorination/dechlorination; Nitrification/denitrification	44	81	5	11	30.8	11.2
	Chlorination/dechlorination; Nitrification/denitrification; Filters	35	89	0	10	0	16.9
	Chlorination/dechlorination; Nitrification/denitrification; Filters	111	86	9	3	96.4	99.4
	Chlorination/dechlorination; Nitrification/denitrification; Filters	76	93	11	5	9.5	25.3

Table 3-4
Summary of Wastewater Treatment vs. Macroinvertebrate Community of Study Areas Where Flow Occurred Upstream and Downstream of Effluent Discharge

	Treatment Level	Taxonomic Richness - Upstream vs. Downstream of Discharge	Percent Cleanwater Taxa Downstream of Discharge
Higher Quality Effluent	Chlorination with no dechlorination	Substantial decline downstream of discharge	None present
	Chlorination with dechlorination; nitrification with denitrification	Variable; increase or decrease downstream of discharge	From none present to less than 10% of aquatic community
	Chlorination with dechlorination; nitrification with denitrification; filtration	Decrease below discharge	Present, percentages range from 17 to 99%. Cleanwater taxa limited to or heavily dominated by baetid mayflies.

An evaluation of long-term changes in aquatic communities relative to upgrades in wastewater treatment can be evaluated only at sites where aquatic species data are available over a sufficient period of time during which upgrades in wastewater treatment were implemented. The sites available for this evaluation were limited to the South Platte River, Fountain Creek, Santa Ana River, and Santa Fe River (**Appendix D**). A site-specific comparison of long-term changes in aquatic communities and concomitant changes in water quality treatment levels show that improved treatment capabilities resulting in improved water quality are not always manifested in an improved aquatic community; moreover, in one instance, the fish community improved following treatment upgrades while the macroinvertebrate community declined (Table 3-5). These mixed results from these four study areas suggest that factors (e.g., habitat limitations), other than wastewater treatment improvements have influenced aquatic community characteristics.

Table 3-5
Summary of Changes in Aquatic Community Structure Following Wastewater Treatment Upgrades

Study Area	Data Record	Wastewater Treatment Upgrade History	Aquatic Community Characteristics
Santa Ana River, California	Macroinvertebrates and fish sampled in 1991 and 1998.	Two discharges combined into single discharge. Tertiary treatment implemented; nitrogen removal.	Macroinvertebrates: Cleanwater taxa abundance increases both upstream and downstream of effluent discharge; prior to treatment upgrades, highest numbers of cleanwater taxa found downstream of effluent discharges. Fish: Species richness increased both upstream and downstream of effluent discharge.
South Platte River, Colorado	Macroinvertebrates sampled 1988 to 1993; fish sampled 1988 to 1998.	Nitrification and denitrification treatment processes added to North Complex by 1991.	Macroinvertebrates: Taxonomic richness remains essentially the same. Fish: Species richness essentially unchanged downstream of effluent discharge.
Fountain Creek, Colorado	Macroinvertebrates sampled in 1980, 1989, 1998 and 1999. Fish sampled in 1980 and 1989.	Dechlorination added to treatment facility in mid-1980s; nitrification and denitrification added in 1996.	Macroinvertebrates: Taxonomic richness markedly lower in 1989 than in 1980; richness rebounds to 1980 levels by 1998 and 1999. In 1998/1999 richness upstream and downstream of the discharge similar. Cleanwater taxa

Table 3-5
Summary of Changes in Aquatic Community Structure Following Wastewater Treatment Upgrades

Study Area	Data Record	Wastewater Treatment Upgrade History	Aquatic Community Characteristics
			abundance greater upstream of discharge in 1998/1999; but cleanwater taxa richness similar at sites immediately upstream and downstream of discharge. Fish: Species richness increased between 1980 and 1989.
Santa Fe River, New Mexico	Macroinvertebrates and fish sampled in 1994 and 2000	Between 1994 and 2000, City of Santa Fe upgraded wastewater facility to include filtration and replaced chlorination with ultraviolet disinfection.	Macroinvertebrates: Taxonomic richness increased Fish: Abundance increased

3.5.3 Limiting Factors: Physical vs. Chemical

3.5.3.1 Overview

As fully described in the discussion on stream ecosystem ecology, the biological community observed in a given stream or river is to a large degree dependent on the physical and chemical template of the environment in which it lives. Therefore, explaining why the biological community of a given stream has the qualities it does requires an understanding of what factor or factors limit the community.

Understanding limiting factors has critical importance in how dischargers and regulators go about the business of implementing water quality programs. This fact is applicable to all stream types, but especially effluent-dependent waters where the "river" is, for the most part, treated effluent. If the goal is an improved aquatic community and the emphasis of the water quality program is only on improving chemical water quality, but it is determined that physical habitat is the limiting factor, then efforts to improve the aquatic community by only focusing on water quality may produce only limited results (Figure 3-34).

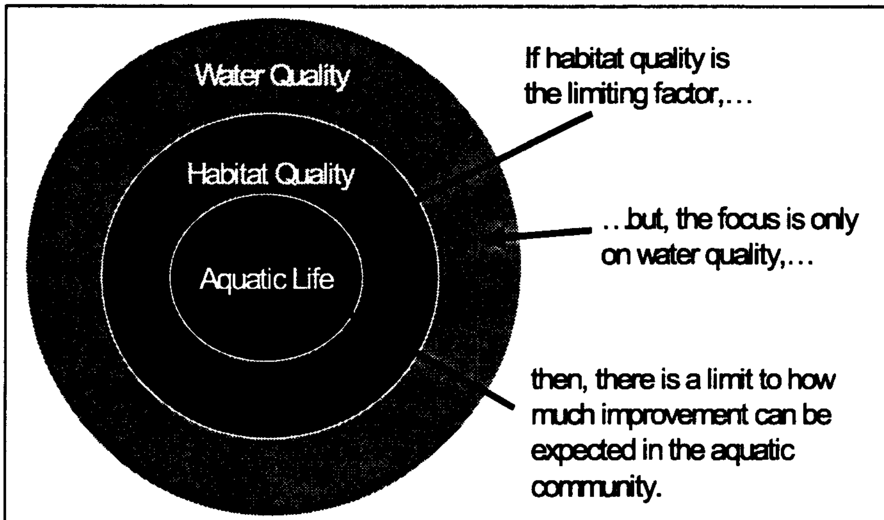


Figure 3-34. Importance of Understanding the Relationship Between Limiting Factors and Aquatic Life.

Unfortunately, there are a multitude of variables that can be measured on any stream. If the question is how the physical or chemical characteristics of a stream affect fish species richness or invertebrate abundance, determining which variables to measure to answer this question can be a daunting task. Moreover, even if the multitude of available variables are measured, it is still unclear how to evaluate the data in a manner that provides insight into determining what the limiting factor is affecting the biological variable of interest.

3.5.3.2 Identifying the Limiting Factor

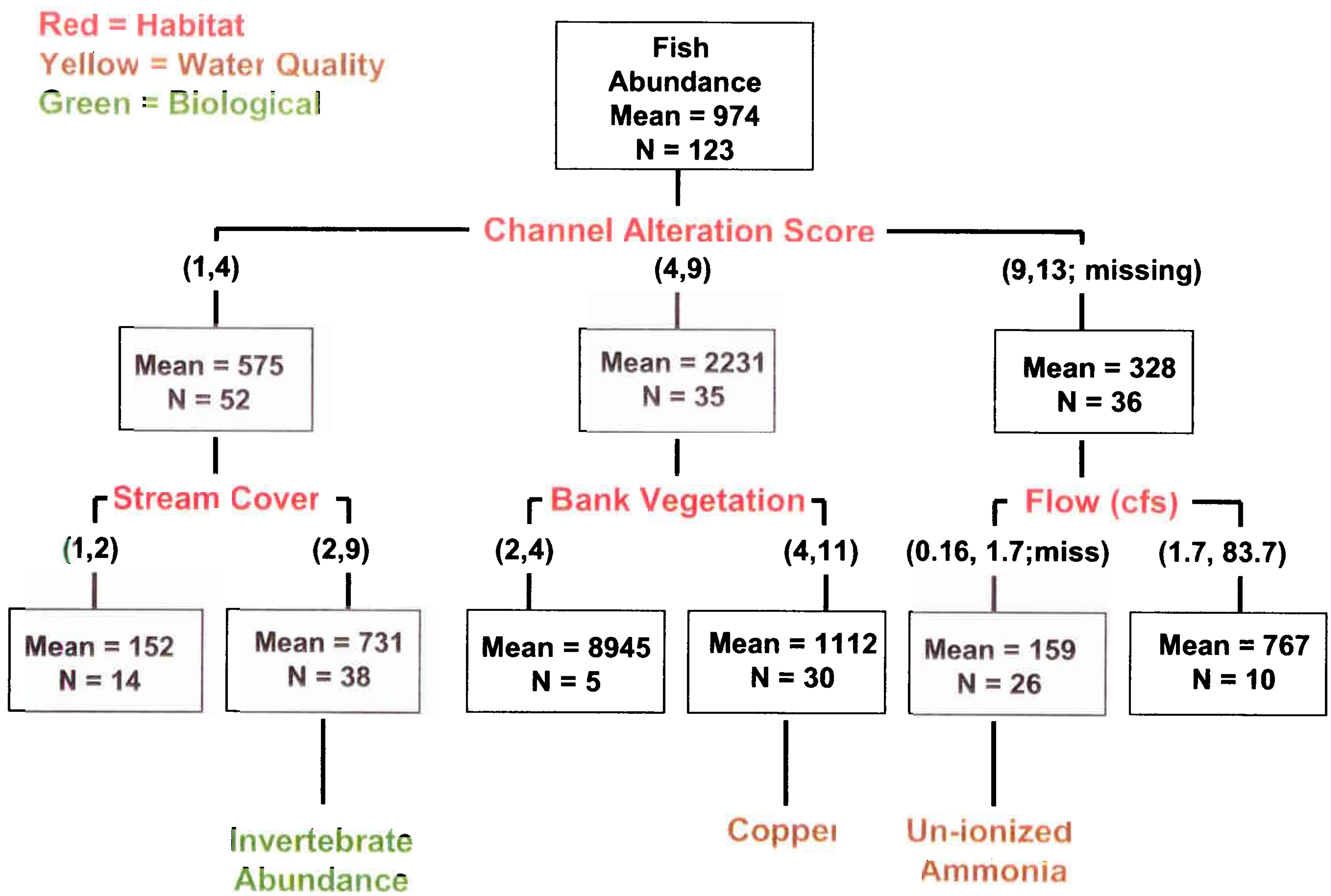
Providing an answer to the problem of identifying the limiting factor has been the subject of recently completed research commissioned by the Water Environment Research Foundation (WERF): *Ability to Discriminate Chemical vs. Habitat Limitations, Project No. 98-WSM-1.* This project developed a multivariate statistical approach using principal components analysis, all regressions analysis and Chi-Square Automatic Interaction Detection (CHAID) to evaluate data and identify the variables that exert the greatest influence on biological response variables (e.g., fish abundance).

CHAID is a non-parametric technique used to identify relationships between one response variable and the independent variables. It additionally looks for interactions among the independent variables and identifies contingent relationships. The output from a CHAID analysis displays a classification scheme in a decision tree format. The first independent variable listed after the response variable is the most important parameter. This independent variable branches into nodes, each of which represents the ranges of the independent variable that act similarly on the response variable. From the nodes representing the first independent variable, additional nodes may branch out indicating that one or more additional independent variables are important within certain ranges of the first independent variable (i.e., a contingent relationship exists between the second and first independent variables).

The importance of this WERF research to the implementation of water quality programs in effluent-dependent waters is significant. At all study areas evaluated by this project, the emphasis of water quality control programs historically has been on improving the chemical quality of effluent produced by the WWTP responsible for creating the effluent-dependent flow. However, two important findings documented by the WERF study are that (1) physical limitations on in-stream habitat appear to be greater than previously understood, and (2) the emphasis on wastewater treatment upgrades has produced only limited improvements in the aquatic communities of effluent-dependent waters.

These findings suggest that the limiting factor in many of the study areas is not effluent quality, but physical habitat. This supposition has been documented by WERF research on two of the study areas: Santa Ana River and Fountain Creek. Results from the multivariate analysis using CHAID demonstrated that physical factors were of greater importance than chemical factors in driving the aquatic community, as exemplified by the two study areas discussed below.

Santa Ana River – The biological response variable, fish abundance, was most affected by channel alteration scores, which were separated into three categories ranging from low scores (i.e., highly altered channels) to high scores (i.e., little channel alteration) (**Figure 3-35**). Fish abundance was highest at intermediate scores of channel alteration. Other independent variables



Simplified Chi-Square Automatic Integration Detection (CHAID) Analysis, Preliminary Diagram – Santa Ana River, CA
Source: WERF Study: Ability to Discriminate Chemical Versus Habitat Limitations, Project No. 98-WSM-1.
Figure 3-35

(e.g., bank vegetation, metals, and ammonia) were found to be important within various ranges of channel alteration scores.

Fountain Creek – The biological response variable, macroinvertebrate taxa richness, was most affected by average embeddedness in stream substrates. Preliminary results from the WERF study found that this variable separated into seven nodes or ranges. At the sixth highest range (87 to 97 percent embeddedness), BOD_5 was found to be a secondarily important variable and at 100 percent embeddedness, the presence of silt, clay, marl, muck, and organic detritus was shown to be important.

The results from the WERF study that included two of the Habitat Characterization Study areas illustrate how the aquatic community can be structured by a complex set of varying physical and chemical variables. Determining which of these variables is the most important in influencing the aquatic community can benefit water quality program efforts to improve aquatic communities. Understanding the most important or limiting factor can help focus and deliver resources to the correct problem.

3.5.4 Chemical Characteristics of Effluent-Dependent Waters

3.5.4.1 Overview

The chemical composition and ionic strength of an aqueous solution can have a significant effect on the organisms inhabiting that solution. In general, ionic strength effects can act to structure or limit an aquatic community (e.g., influencing species composition or species abundance). The simplest example of this effect is demonstrated by the fact that 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 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 (**Appendix J**).

Ionic strength effects are generally evaluated using a variety of water quality measurements, most commonly conductivity, total dissolved solids, 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 (**Appendix J**).

As a measure of how chemical composition and ionic strength may affect arid West streams, the water chemistries from the 10 study areas were compared to following: (1) standard Whole Effluent Toxicity (WET) testing water chemistry; (2) the toxicity database water chemistries used in deriving aquatic life criteria for ammonia, cadmium, copper, and zinc; and (3) water chemistries of the Kansas and North Carolina non-arid sites.

3.5.4.2 WET Test Water Chemistry

Freshwater WET tests are usually conducted with a zooplankton (*Ceriodaphnia dubia)* and the fathead minnow (*Pimephales promelas)* using moderately hard synthetic (MHS) freshwater as a control and diluent. MHS water is obtained by adding magnesium sulfate (1.20 grams $MgSO_4$), sodium bicarbonate (1.92 grams $NaHCO_3$), potassium chloride (0.08 grams KCl) and calcium sulfate (1.20 grams $CaSO_4 \cdot 2H_2O$) to 20 liters of deionized water.

A Piper diagram was developed to provide a visual comparison of MHS water chemistry with the water chemistry at study areas with sufficient data (refer to **Figure 3-13; Appendix B**). The relative proportions of many of the major cations and anions differ between the chemistry of the MHS water and the typical water chemistry of the study areas (Table 3-6). In addition, the WET water composition is also lower than the arid streams with regard to total dissolved solids concentration, as shown by the size of the total dissolved solids circles on the Piper diagram, where an increased radius indicates increased total dissolved solids concentrations (refer to **Figure 3-13**).

Table 3-6
Comparison of Chemistry of MHS Water and Study Area Waters

Source	Concentration (Percent millequivalents/Liter)					
	Mg	Ca	Na+K	Cl	SO4	CO_3+HCO_3
MHS Water	35	25	40	2	60	40
Study Areas (range)	7 – 25	25 – 42	43 - 63	15 - 70	15 - 57	10 - 72

3.5.4.3 Toxicity Test Water Quality Characteristics

Water quality characteristics of test waters used to develop national water quality criteria for cadmium, copper, zinc, and ammonia were obtained by reviewing the original references documented in the EPA water quality criteria documents for these constituents (**Appendix K**). The ranges of hardness, alkalinity, and conductivity concentrations for the study areas were often greater than the ranges of concentrations used in toxicity studies for the constituents reviewed (Table 3-7). For one study area, Las Vegas Wash, the difference between the hardness and conductivity concentrations of in-stream waters and toxicity test waters was substantial.

Table 3-7
Comparison Between the Water Chemistry Associated with Water Quality Criteria Toxicity Studies and Study Area Waters

<table>
<tr><th rowspan="2">Source</th><th colspan="4">Concentration</th></tr>
<tr><th>Hardness (milligrams/Liter)</th><th>Alkalinity (milligrams/Liter)</th><th>Conductivity (umhos/centimeter)</th><th>pH (Standard Units)</th></tr>
<tr><td>Toxicity Studies</td><td>50 – 200</td><td>25 – 175</td><td>0 – 500</td><td>6.0 – 9.0</td></tr>
<tr><td>Case Study Sites*</td><td>100 - 500</td><td rowspan="2">50 – 300</td><td>500 – 1200</td><td rowspan="2">6.0 – 9.0</td></tr>
<tr><td>Las Vegas Wash*</td><td>600 - 900</td><td>2000 - 3000</td></tr>
<tr><td>North Carolina Sites</td><td>≈ < 25</td><td>≈ < 25</td><td>0 – 400</td><td>6.0 – 9.0</td></tr>
<tr><td>Kansas River</td><td>100 - 400</td><td>100 – 250</td><td>300 – 1600</td><td>6.0 – 9.0</td></tr>
</table>

* - For specific parameters, Las Vegas Wash is separated from other study areas.

3.5.4.4 Non-Arid Stream Water Quality Characteristics

Chemical data from seven sites on three non-arid streams were compared with chemical data from nine of the study area streams. The seven non-arid stations included three stations on the French Broad River, two stations on the Tar River, and one station on the Ararat River, all in North Carolina; and one location on the Kansas River in Kansas.

The conductivity of North Carolina streams was similar to the toxicity test conditions; in contrast, the conductivity in the Kansas River was similar to the majority of the study areas (refer to Table 3-7). Hardness and alkalinity in the North Carolina streams were low, at the lower end of concentrations used in toxicity tests. As with conductivity, hardness and alkalinity concentrations in the Kansas River were similar to most of the study areas.

Major cation/anion data were available from only one North Carolina site (French Broad River) and the Kansas River. The Kansas River had a similar average major ion composition as the 10 study areas (refer to **Figure 3-13**). The French Broad River, however, had a higher Na+K composition (70 percent millequivalents/liter [meq/L]) and lower Ca composition (20 meq/L) than the arid streams (43 to 63 and 25 to 42 percent meq/L, respectively). In addition, total dissolved solids in the French Broad River appeared to be lower than for the 10 study areas (compare relative sizes of circles on the Piper diagram). French Broad River anion composition more closely resembles the WET test composition than the case study streams. However, cation composition, especially Na + K, was different from the composition for both the MHS water and 10 study areas.

Ammonia concentrations in the non-arid streams were typically much lower (<1 milligrams/liter [mg/L] as N) than concentrations in the arid streams (range of <1 mg/L to as high as 40 mg/L with a median often between 2 and 10 mg/L). As would be expected, the highest ammonia levels for the study areas were observed downstream of the effluent discharges. Typically, ammonia levels were greatest immediately below the discharge, but decreased with increased distance downstream. This trend was not apparent in the non-arid sites, probably due to the significant influence of dilution at these locations.

3.5.4.5 Chemistry Summary

Based on the review of chemical data, the following conclusions may be made:

- Important ionic composition differences exist between the study areas and the water used to conduct WET tests. This difference was exemplified best in the Las Vegas Wash.
- Important differences exist between the ionic composition of waters used to develop water quality criteria for cadmium, copper, zinc, and ammonia and the ionic composition of waters from the study areas.
- The 10 study areas have greater ionic strength than the North Carolina streams as measured by total dissolved solids, conductivity, hardness, and alkalinity. Additional chemical data from non-arid streams over a broader geographical area would need to be reviewed to determine the geographical extent of observed differences.
- The effect of dilution on in-stream ammonia concentrations was evident when the study areas were compared to the Kansas and North Carolina streams.

3.5.5 Ecological Benefits of Effluent Discharge

3.5.5.1 Overview

At the Habitat Characterization Study areas the discharge of effluent either augments flow from upstream sources (i.e., the stream is effluent-dominated), or constitutes the only flow in what was an otherwise dry riverbed (i.e., the stream is effluent-dependent). Whether a stream is "dominated" or "dependent" on effluent on a given day can vary from season to season or climatologically (i.e., wet and dry cycles lasting for periods of years). During this study's site reconnaissance, flow augmentation was observed in the Santa Ana River, Fountain Creek, South Platte River, Las Vegas Wash, and Crow Creek. Sites where 100 percent of the flow was effluent included the Santa Cruz River at Nogales and Tucson, Salt River, and Santa Fe River.

The addition of effluent, which augments or creates in-stream flows, has the potential to influence aquatic and terrestrial species richness and diversity in stream ecosystems in the arid West. This influence may be positive or negative depending on many factors including both habitat and water quality. Regardless of whether the influence is positive or negative for aquatic species, the potential for a positive response from or benefit to the terrestrial ecosystem associated with the stream ecosystem is great. These 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.

3.5.5.2 Taxonomic Richness of Aquatic Communities in the Arid West

Aquatic communities in relatively 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 decreased persistence and stability in the face of unpredictable disturbances (Peckarsky 1983; Poff and Ward 1989).

Arid West streams, especially those in the driest regions, are considered to be relatively harsh environments (**Appendix I**). As such, these streams would be expected to have relatively low species richness.

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 2002). 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 0.7 square meter (m^2) (7.5 square feet).

The compilation of samples, which do not distinguish between impacted and non-impacted sites, shows that in the arid West, especially in lowland areas (i.e., drier environments), the number of macroinvertebrate families was most commonly found to be between 6 and 15. Sites with 5 or fewer families were rare and sites with samples containing more than 20 families were increasingly uncommon, except possibly at higher elevations, which are typically less arid areas.

ADEQ recently summarized the results of aquatic invertebrates samples collected from warmwater non-effluent-dependent 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 1 m^2 (10.8 square feet) during the spring season. At the "family" level of identification (insects identified to family; non-insects identified 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).

At the 10 study areas for this habitat characterization, the median number of taxa ("family" level identification) observed during the reconnaissance level survey ranged from 3.5 to 12.5 with most sites having a median of 6 to 9 taxa (Table 3-8). A review of available historical data, where the sampling effort was greater than that conducted during the site reconnaissance, found that the median number of taxa ("family" level identification) ranged from 6 to 13.5 (Table 3-8). The slightly higher median value for these historical studies likely represents improved data quality (i.e., the site reconnaissance data were based on field identification only, while the historical data were based on a laboratory analysis of field samples which in some cases included seasonal data).

Table 3-8
Number of Taxa (Family/Order/Class Level) at Study Areas (Dry Sites Excluded)

Data Source	Waterbody	Year	Number of Families/Site Range	Median
Site Reconnaissance	Santa Fe River	2000	10 – 17	12.5
	Santa Cruz River, Tucson	2000	2 – 7	3.5
	Santa Cruz River, Nogales	2000	6 – 15	10
	Salt River	2000	5 – 11	7
	Las Vegas Wash	2000	5 – 11	6
	Santa Ana River	2000	3 – 10	9
	Carrizo Creek	2000	5 – 15	9
	South Platte River	2000	5 – 9	7
	Fountain Creek	2000	5 – 11	6
	Crow Creek	2000	2 – 16	6
Historical Studies	Santa Ana River (quarterly sample means)	1991	5 – 12	9
	Santa Ana River	1998	6 – 20	10
	Crow Creek	1993	4 – 17	6
	South Platte River	1988	2 – 8	7
	South Platte River	1989	4 – 14	7
	South Platte River	1990	7 – 11	8
	South Platte River	1991	6 – 10	9
	South Platte River	1993	6 – 9	9
	Santa Fe River	1994	9 – 13	10.5
	Santa Cruz River, Nogales (monthly sample means)	1993	9 – 17	13.5
	Fountain Creek	1980	9 – 18	12
	Fountain Creek (Fall)	1999	11 – 17	13

With few exceptions, the number of "families" observed using reconnaissance or historical data falls within the range of 6 to 15 families found by NAMC for sites throughout arid regions of the West. Median numbers are lower than the 16 taxa found as the median number of "families" observed 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 all anthropogenic activities, both point and nonpoint sources.

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 sites. As a consequence, while the number of "families" was greater upstream of the wastewater effluent in five of six cases, it is unknown if the upstream richness was indicative of typical expectations for arid West streams as a whole.

The results from the review of historical and reconnaissance data strongly suggest that for effluent-dependent waters in the arid West the number of "families" is lower than might be expected for non-effluent-dependent waters. However, results from the NAMC data compilation effort suggest that as a whole, taxonomic richness is lower in arid West streams when compared to non-arid western streams.

3.5.5.3 Factors Limiting Species Richness

Determining what promotes or limits 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 to determine which 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) taxonomic richness tends to be greater with larger substrate median particle size; and (2) as the disturbance intensity and/or frequency increase, taxonomic richness typically declines (Appendix I).

- Basin Scale: (1) an increased annual temperature range often results in increased taxonomic richness; and (2) as flow intermittency increases, taxonomic richness typically declines.

- 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.

Several of the above factors that can result in lower taxonomic richness identified by Vinson and Hawkins (1998) can be associated with characteristics of arid West stream ecosystems, especially effluent-dependent waters. For example, the importance of annual temperature range on effluent-dependent waters may be significant. Flow in these waters consists of a relatively constant rate of discharged effluent and the annual temperature range of this effluent at the point of discharge and for some distance downstream likely will be narrow. In addition, species richness in effluent-dependent waters may be negatively influenced by flow intermittency, especially seasonally. Flows in effluent-dependent waters can be subject to seasonal variability as discharge varies because of seasonal reuse.

From the review by Vinson and Hawkins (1998) one could easily conclude that the combined influence of the above factors on species richness in effluent-dependent waters 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 from the available information. Furthermore, to fully evaluate whether lower species richness results from habitat limitations, it would be necessary to eliminate water quality as a limiting factor.

3.5.5.4 Effluent Discharge and the Aquatic Community

Results from the Habitat Characterization Study suggest that (1) improvements in wastewater treatment levels have resulted in only modest improvements to the aquatic community, especially with regards to taxonomic richness (refer for example to Tables 3-3 and 3-4); and (2) the median number of "families" in effluent-dependent waters, while lower than what might be expected as a whole in non-effluent-dependent waters in the arid West, does not appear to be substantially lower.

Results from this study suggest that a viable aquatic community is present even in effluent-dependent streams. A viable aquatic community provides benefits in an ecosystem that, except for the input of effluent, would be dry most of the time. While these benefits could be compartmentalized into ecosystem subcomponents, these benefits generally can be collapsed into a single overall benefit – the aquatic community provides food resources to higher trophic levels, especially fish and terrestrial communities. The link between aquatic and terrestrial communities is not well studied, but as noted above, an important study conducted on a southwest arid non-effluent-dependent water found that 96 percent of the biomass of emerging insects was transferred to the terrestrial ecosystem as food to terrestrial insectivores (e.g., birds and bats) (Jackson and Fisher 1986). The importance of this link cannot be minimized and illustrates well the potential benefit to the terrestrial ecosystem of a viable aquatic community.

Results from the Habitat Characterization Study reconnaissance-level survey show that the creation of effluent-dependent waters yields mixed results with regards to the increase or decrease in taxonomic richness. Clearly, if the waterbody is dry upstream of the effluent discharge, then a net increase in taxonomic richness would be expected downstream of the discharge. Such a scenario was observed in 4 of the 10 study areas. At the remaining study areas, at least some flow was present upstream of the discharge. At Fountain Creek, richness was greater downstream of the effluent discharge than upstream of the discharge. At the remaining five study areas, richness was lower downstream of the discharge.

While the above comparison focuses on taxonomic richness, a better measure of benefit probably would be biomass, since it is the transfer of biomass from the aquatic macroinvertebrate community to higher trophic levels that supports the fish and terrestrial communities. Fewer taxa may not result in lower biomass; thus, to evaluate whether the ecosystem is functioning in a manner that is beneficial to higher trophic levels requires additional data that were not available from this study.

3.5.5.5 Benefits of Effluent Discharge to Terrestrial Communities

Riparian ecosystems develop in the arid western United States in direct response to the presence of 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 (1964) 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." 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 1974; Carothers et al. 1974; Rosenberg et al. 1991; and many others).

The finding that the discharge of treated wastewater influences the presence and structure of riparian systems in otherwise dry streambeds is unequivocal. Of the study areas included in this analysis, 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 results of the terrestrial habitat assessment conducted during the site reconnaissance clearly show that riparian habitat values are lower upstream in those situations where upstream waters are ephemeral (**Table 3-9; Appendix B; Appendix D**). For example, upstream assessment values from the Santa Fe River, Santa Cruz River (Tucson and Nogales), Las Vegas Wash, and Salt River were noticeably lower than assessment scores for areas downstream of the outfall (**Table 3-10**). In contrast, those sites where upstream waters were at least intermittent showed little or no difference in assessment scores upstream and downstream of the effluent outfall (e.g., Crow Creek, South Platte River, Fountain Creek, Santa Ana River, and Carrizo Creek) (**Table 3-11**).

Table 3-9
Terrestrial Habitat Characterization - Standardized Scores for Each Site

Study Area	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 Springs	CS1	74	78	33	100
	CS2	100	47	51	92
	CS3	89	39	26	65
	CS4	74	33	33	65

Table 3-9
Terrestrial Habitat Characterization - Standardized Scores for Each Site

Study Area	Site Name	TBRV	TBHC	TBWO	TBDC

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 3-10
Terrestrial Habitat Assessment Scores, Study Areas with Upstream Site Dry (Site 1 – Upstream; Sites 2 to 5 – Downstream)

Study Area		Location vs. Effluent Discharge	Site Number	Terrestrial Assessment Score
Upstream Site Dry	Santa Fe River	Up	1	58
		Down	2	101
		Down	3	155
		Down	4	200
		Down	5	182
	Santa Cruz River, Tucson	Up	1	57
		Down	2	154
		Down	3	272
		Down	4	220
		Down	5	273
	Santa Cruz River, Nogales	Up	1	93
		Down	2	147
		Down	3	287
		Down	4	235
		Down	5	207
	Salt River, Phoenix	Up	1	54
		Down	2	190
		Down	3	187
		Down	4	144
		Down	5	188

3.5.5.6 Riparian Ecosystems: Arid vs. Non-Arid Regions

It has been assumed that there is a fundamental difference between the terrestrial component of riparian ecosystems in the arid West and non-arid sites. A basis for this assumption is the often obvious distinction between riparian and upland habitats along streams in arid regions and the less apparent distinction between such habitats in non-arid regions. Evaluating the validity of this assumption can be accomplished by comparing lists of terrestrial vertebrate species expected for sites in arid and non-arid settings. Lists of potential mammals, birds, reptiles, and amphibians were compiled for each study area and compared with lists compiled from several non-arid sites in North Carolina and Kansas. 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 available habitat (**Tables 3-12 and 3-13; Appendix M; Appendix** N).

If our assumption regarding differences between arid and non-arid riparian ecosystems is correct, the selected non-arid areas should have a greater proportion of species that are found in both the riparian and the upland habitats, while these same habitats in the arid West should have fewer species in common. **Table 3-14** summarizes the results of a statistical analysis comparing the 10 study areas with the selected non-arid sites. These results show a significant difference between the 10 study areas and non-arid areas for birds, reptiles, and amphibians, but no significant difference for mammals (**Table 3-15**).

The primary reason for the significant difference in bird distributions is probably related to the high proportion of bird species in the 10 study areas that are restricted to riparian habitats. In non-arid conditions, fewer birds are restricted to the riparian zones adjacent to the rivers. The reason for the significant difference for reptiles and amphibians appears to be more complicated than that for birds. In most of the 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, Colorado Springs, and Carrizo Creek) 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 study 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. The lack of a significant difference for mammals may be a result of relatively few mammals being restricted to either a riparian or an upland habitat. Most mammals are capable of using both habitats.

Table 3-11
Terrestrial Habitat Assessment Scores, Study Areas with Intermittent Upstream Site

	Case Study Site	Location vs. Effluent Discharge	Site Number	Terrestrial Assessment Score
Upstream Site Intermittent	Crow Creek, Cheyenne	Up	1	215
		Down	2	232
		Down	3	206
		Down	4	219
		Down	5	240
	Las Vegas Wash	Up	1	150
		Down	2	264
		Down	3	209
		Down	4	256
		Down	5	203
	South Platte River, Denver	Up	1	148
		Down	2	156
		Down	3	189
		Down	4	201
		Down	5	171
	Fountain Creek, Colorado Springs	Up	1	118
		Down	2	198
		Down	3	188
		Down	4	219
		Down	5	255
	Santa Ana River, San Bernardino	Up	1	225
		Down	2	269
		Down	3	302
		Down	4	253
	Carrizo Creek, Carrizo Springs	Up	1	285
		Down	2	290
		Down	3	219
		Down	4	205

Table 3-12
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 Area/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	

Table 3-12
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 Area/Habitat	Mammals		Birds		Reptiles and Amphibians	
	Number of Species	Percent	Number of Species	Percent	Number of Species	Percent
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	
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

Table 3-12
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 Area/Habitat	Mammals		Birds		Reptiles and Amphibians	
	Number of Species	Percent	Number of Species	Percent	Number of Species	Percent
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 3-13
Numbers of Potential Species at Each of the Non-Arid Sites, 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
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

Table 3-14
Mean Numbers of Potential Species at Arid West Study Areas and Non-Arid Sites Sorted by Numbers of Species Restricted to Riparian Habitats, Upland Habitats, or Able to Use Both Habitats

Habitat Type	Mammals		Birds		Reptiles and Amphibians	
	Mean Number of Species (%)	Standard Deviation	Mean Number of Species (%)	Standard Deviation	Mean Number of Species (%)	Standard Deviation
Arid West Study Areas						
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
Non-Arid Sites						
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

Table 3-15
Results of Statistical Analysis Comparing Mean Percentage of Species Using Both Upland and Riparian Habitats in Arid West Study Areas and Non-Arid Sites

Vertebrate Group	Location	N	Total Degrees of Freedom	Calculated t-statistic ($t_{0.05, 12}$)	Significant at 95% Confidence Level
Mammals	Study Areas	10	12	1.486 (2.179)	No
	Non-arid sites	4			
Birds	Study Areas	10	12	7.344 (2.179)	Yes
	Non-arid sites	4			
Reptiles and Amphibians	Study Areas	10	12	4.688 (2.179)	Yes
	Non-arid sites	4			

Overall, with the exception of mammals, the results from the terrestrial species analysis confirm expectations that there is a fundamental difference between the terrestrial component of riparian ecosystems in the arid West and non-arid areas. This finding reinforces the importance of supporting riparian habitats in the arid West, including those created as a result of the discharge of effluent. In addition, the aquatic community supported by the effluent flow can serve as an important food resource for animals using this riparian habitat, especially birds.

3.5.6 Applicability of EPA Aquatic Habitat Assessment Protocols to Effluent-Dependent Waters

3.5.6.1 Overview

In-stream 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. EPA's Rapid Bioassessment Protocol (RBP) is an example of this type of procedure.

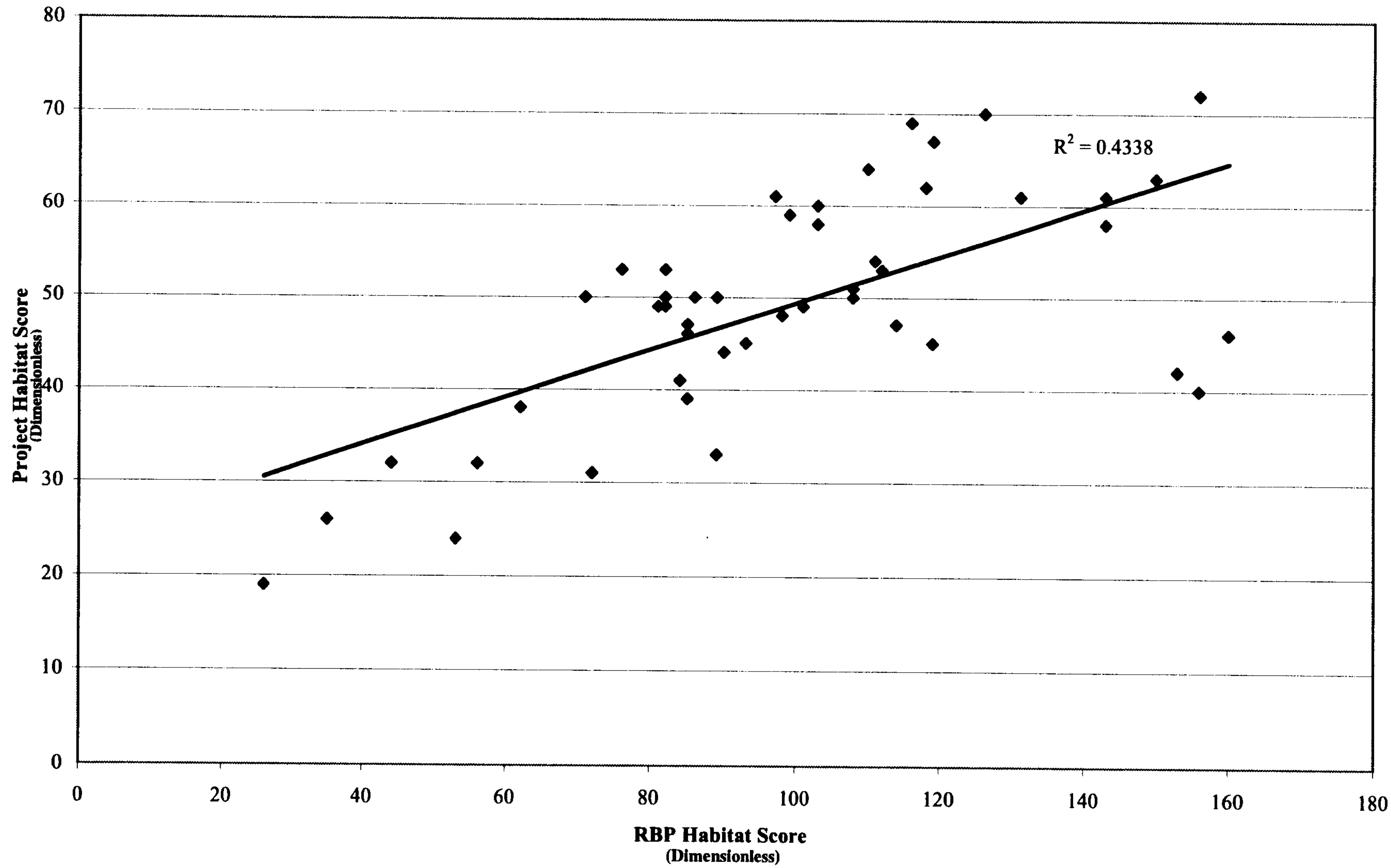
However, assessment of habitat in streams of the arid West may not be accurately portrayed by direct application of the RBP procedures. The original RBP habitat procedures were actually based on habitat assessment techniques developed in Wisconsin and Idaho, and especially trout streams in these states. Processes that create optimum habitat in those regions may not be as important, or even relevant, in the arid West. 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 procedures. However, the ephemeral or intermittent nature of many western 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 that rates habitat characteristics that are more applicable to arid West streams.

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 arid West 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 arid West streams, would lead to a more realistic picture of what the optimum condition is for the substrate/cover parameter.

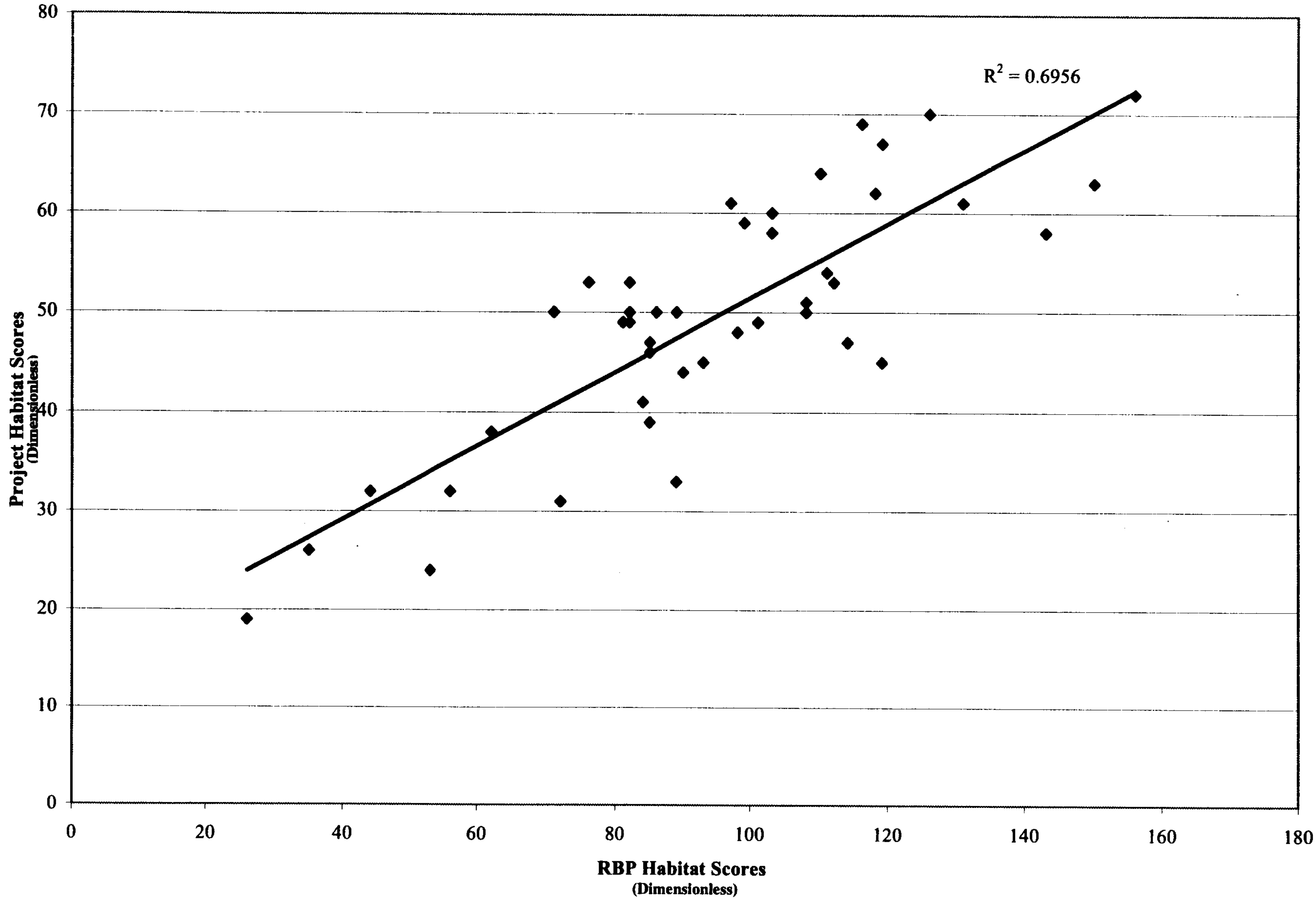
3.5.6.2 RBPs versus Alternative Field Method

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 be 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 3-36**). However, it must be noted that while the relationship using all sites was positively significant, one site showed no relationship (Crow Creek) and one site had a strong but insignificant negative relationship (Carrizo Creek). Removing Carrizo Creek data from the data analysis results in a much greater significant relationship between the two scoring methods (**Figure 3-37**; $r^2 = 0.6956$). Although Carrizo Creek and Crow Creek did not follow the pattern observed at the other eight sites, 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.



Linear Regression - RBP vs. Project Habitat Assessment Scores
Figure 3-36



Linear Regression - RBP vs. Project Habitat Assessment Scores (Carrizo Creek Removed)
Figure 3-37

While it is recognized that habitat assessment methods developed by EPA and others do not appear to be geared towards arid West type waters, it must be kept in mind that what is critical to a habitat assessment is not the total habitat assessment score. Instead, what is important is how the habitat assessment score for a particular site compares to the habitat score for what is determined to be a representative reference site. For arid West waters, it is likely that some factors in the EPA RBP score sheet may score relatively low in arid West streams – not necessarily because the habitat is "poor" but because the habitat of ephemeral/intermittent streams is naturally limited. If an appropriate reference site is selected, i.e., one appropriate for comparison to an effluent-dependent, intermittent or ephemeral stream, it is expected that the selected reference stream will have the same waterbody-specific habitat limitations as the non-reference stream. As long as this principle is adhered to, the EPA RBP should be applicable to any region, arid or non-arid. Also, it is important to note that EPA recommends gathering additional habitat data to supplement the RBP habitat assessment scoring 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 arid West waters.

3.5.7 Toxicity Database Species

3.5.7.1 Overview

The 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, etc.)
- Aquatic insect family (e.g., Ephermoptera, Plecoptera, Trichoptera, Diptera, etc.)
- 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.

3.5.7.2 Taxonomic Groups Present at Study Sites

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 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 is being evaluated for specific pollutants under the *Extant Criteria Evaluation* study, a separate WQRP-funded project.

Table 3-16 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 sites 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 stream sites are located in biomes that typically have warmwater-type streams.

Cladocera have been recorded at most sites, the exceptions being Fountain Creek, Santa Fe River, and Carrizo Creek. We cannot assume that the lack of a record for Cladocera at these sites 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 and Gila Rivers – *Ceriodaphnia*; Santa Cruz River, Tucson – *Daphnia*). Again, failure to document these taxa from

Table 3-16
Comparison of Taxonomic Groups Known to be Present at Habitat Characterization Study Areas and Taxonomic Categories Required for Calculation of Acute Toxicity Criteria

Categories	Salt/Gila Rivers	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									

the other eight sites does not preclude the possibility of their presence since zooplankton sampling has not been routinely carried out at most study areas.

3.5.8 Ephemeral Waters vs. Effluent-Dependent Waters – Is Wetter Better?

3.5.8.1 Ephemeral Waters: Natural or Created

The addition of effluent to an otherwise dry or frequently dry riverbed results in a change to the existing flow conditions of the receiving waterbody. The existing flow conditions prior to effluent discharge are either ephemeral (i.e., flowing in response to precipitation events), or intermittent, flowing periodically (e.g., seasonally as a result of snowmelt in the upper watershed). Although the terms "ephemeral" or "intermittent" can be used to describe the flow characteristics of the waterbody receiving the effluent discharge, it is important to note that the existing flow characteristics may not represent historical conditions.

Historically, several of the study areas had considerably more flow than what would be present today without the effluent. For example, today the Salt River study area in much of the Phoenix area is classified as ephemeral by the State of Arizona (Arizona Administrative Code, Title 18, Chapter 11, Article 1). However, historically the river was perennial; the river only became ephemeral because of the construction of upstream storage reservoirs and diversion dams prior to 1930 (Tellman et al. 1997). In addition, historically the South Platte River was an intermittently flowing river with significant flows in the spring during snowmelt runoff from the Rocky Mountains, but limited flows in the summer and fall. Today flow in the South Platte River is greatly limited, again the result of upstream storage reservoirs and downstream diversion dams that deliver water for industrial and agricultural uses. For these types of effluent-dependent waters, the addition of effluent returns at least part of the river to a perennial condition, although the addition of effluent cannot be considered as a replacement for natural flows.

In contrast to the Salt River and South Platte River study areas, several areas were either ephemeral historically, or at best seasonally intermittent. For example, Carrizo Creek appears to have been ephemeral prior to the addition of effluent. In addition, both the Santa Fe River and Las Vegas Wash likely had limited flows historically. So, in contrast to areas where effluent acts as a replacement for historically natural flows, some effluent-dependent waters represent a significant change from historical flow conditions.

The addition of effluent to a riverbed is often portrayed as a benefit in the context that any water in a riverbed is better than no water in a riverbed. For sites where the effluent replaces historically natural flows, this thought process makes sense. However, where the effluent discharge creates a flowing river, where none previously existed, the question can be asked, what has been lost or changed by the addition of effluent? This question is relevant because naturally ephemeral streams have important biological attributes that are as distinct as the biological attributes of a naturally perennial river.

3.5.8.2 Biological Communities of Ephemeral Waters

The aquatic biological community of ephemeral waters is limited to opportunistic types of non-fish organisms that can quickly colonize temporary waters. The richness and diversity of this temporary community is greatly dependent on how long water remains in the system. For example, seasonally intermittent flows will support a more substantial aquatic community than the remnants of storm flows that are present only for days or at most a few weeks following the storm event.

In contrast to the aquatic biological community, the terrestrial biological community associated with ephemeral channels can be important, not because of an attribute such as richness or diversity, but because of the functional significance the ephemeral channel can have in the landscape, especially an arid landscape. For example, ephemeral channels serve as important biological corridors for terrestrial organisms, providing connectivity between upland and lowland areas. Vegetation density and diversity is often greater within and along ephemeral channels providing cover and food for terrestrial organisms. In addition, the vegetative cover along ephemeral channels creates a microclimate that helps support and sustain the associated biological community.

Without question, the addition of effluent into what would be a naturally ephemeral channel is a change to the natural state of the system. With the addition of an artificial perennial flow, biologically speaking, the system clearly will be different from how it was historically. Biological attributes such as aquatic community richness and diversity likely will be greater. The increased biological productivity of the aquatic community will provide additional food resources for terrestrial organisms. In addition to these changes in the aquatic community, the terrestrial community will be substantially different, especially in terms of the types of organisms supported.

Are these biological changes good? Is having a wetter channel better biologically? These questions have no simple or single answer. In fact, the answer will depend on public values and local needs. One can easily argue that the number of ephemeral channels, especially in arid regions, far exceeds the number of naturally perennial channels, and thus the creation of a perennial stream in a previously ephemeral stream is a positive benefit. However, in some areas, especially in rapidly developing urban environments, the number of lost ephemeral channels can be significant and the loss of habitats as a result of effluent discharge can be an important issue for the public to consider.

While the public needs to evaluate the benefits of changes that will be invoked by the addition of effluent, important consideration also must be given to where an effluent-dependent channel should be created in the first place. As discussed in **Chapter 4**, one of the important findings from the evaluation of the study areas is the need to consider physical and hydrological principles when selecting a location for an effluent discharge. Therefore, the question of whether wetter is better must be evaluated within many contexts and should be part of the public evaluation process.

3.6 SUMMARY

An effluent-dependent waterway must be viewed as a created system in search of a stable relationship with its surrounding environment. Similarly, an effluent-dependent waterway cannot be viewed as a natural, perennial water in sync with its surroundings. Given enough time and assuming no additional stressors, the created system will establish a new equilibrium, but until that occurs, expectations for a biological community that is similar to a natural stream in the same region cannot be achieved simply based on physical and hydrological considerations.

In addition to physical and hydrological restrictions, limitations imposed on the biological community by the chemical characteristics of effluent also must be considered. Arguably, increased levels of treatment, resulting in improved effluent quality, will result in at least some improvement in the biological community over the long term. However, the degree to which improved treatment will result in this improved biological community is first and foremost limited by the physical template upon which the biological community must colonize. Moreover, the importance in understanding what is limiting the biological community of effluent-dependent waters cannot be emphasized enough.

With these considerations in mind, establishing a goal to achieve an aquatic community in an effluent-dependent water with characteristics similar to an aquatic community in a natural stream may be inappropriate. The physical effects of effluent discharged into a streambed that is dry during most of the year may work against the benefits to the aquatic community that might be achieved from improved water quality. Superimposed on this template are activities that work against achieving a positive physical environment for aquatic organisms, activities such as channel modifications for flood control, hydrologic modifications, water diversions, grade control structures, additional effluent discharges, and bridges. Each of these activities further disrupts the natural tendency for these streams to establish equilibrium.

Although habitat, water quality, or both may limit the aquatic system, the terrestrial community is only limited by factors associated with habitat (often temporary) and non-native species. As indicated above, the contrast between waters above and below the effluent discharge can be significant and the support of greater vegetative diversity provides increased benefits for many terrestrial wildlife species.

Understanding the potential for biological communities in effluent-dependent waters is important not only from a technical or scientific standpoint, but also from a regulatory perspective. After all, determining what is attainable in a waterbody forms the foundation for the establishment of water quality goals under the Clean Water Act. Development of the aquatic and terrestrial community in and along effluent-dependent waters is dependent on a variety of physical and chemical factors. How these factors may limit the development of these communities is critical to an evaluation of what uses are truly attainable in these waters. **Chapter 5** discusses the regulatory implications of use attainability in the context of effluent-dependent waters. However, prior to this regulatory discussion, it is important to consider some economic issues associated with effluent-dependent waters. Specifically, it is important to consider that in the arid West, water is a commodity, and, as a consequence, there is a link between the value of water and its quality. This link is explored in more detail in the next chapter.