APPENDIX E

DISCUSSION

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APPENDIX E

1.0 DISCUSSION

1.1 STREAM ECOSYSTEM ECOLOGY

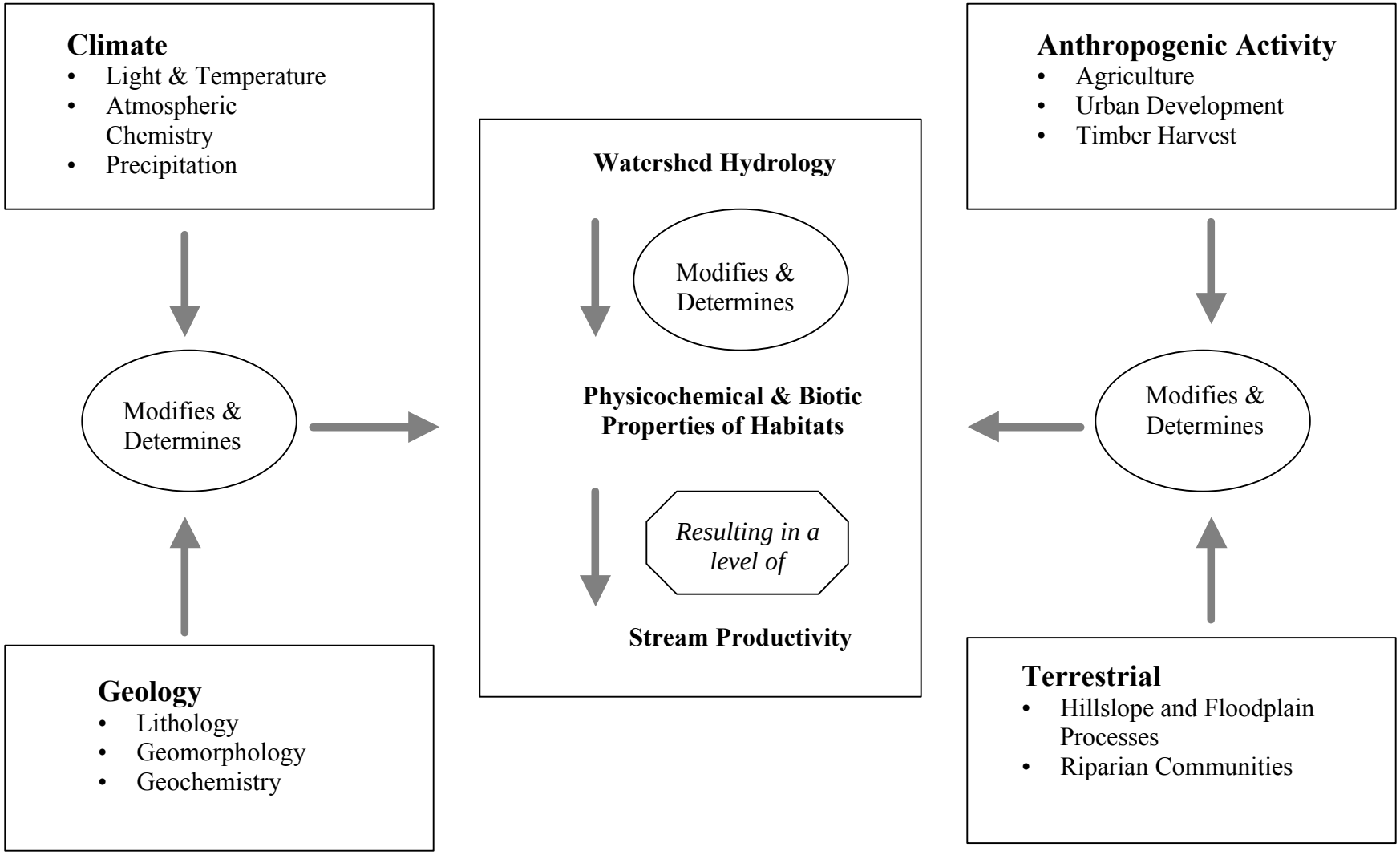
Ecosystems associated with flowing water systems are complex, with many possible pathways for energy transfer and material cycling. The fundamental components of a stream ecosystem consist of the physical, chemical, and biological attributes that collectively define the system. The basic concepts of energy flow through a system and nutrient cycling within a system are covered in any basic ecology textbook (e.g., Resh and Rosenberg 1984; Smith 1990). The movement of water through a system increases the possible movement routes for energy and nutrients, and a river system must be considered as a continuum that is intimately connected to upstream and downstream systems, with spatial and temporal continuity (Vannote et al. 1980). Ward (1989) emphasized the four-dimensional nature of lotic ecosystems. These dimensions include a longitudinal component along the length of the river, a lateral component to encompass the floodplain and associated riparian vegetation, a vertical component to include the substrate and groundwater, and a temporal component.

Conceptual models of stream ecosystems have generally been devised for natural drainage basins or those in which land use stresses do not cause huge changes over very short channel reaches. In contrast conceptual models for effluent-dependent waters, especially those created in drainages that have only intermittent or ephemeral streamflow, have not been developed. Stream ecosystems created from the discharge of effluent may exhibit substantial physical and chemical differences from naturally flowing channels. To evaluate the degree of difference between a natural and created stream ecosystem, the following discussion is provided. This discussion 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.

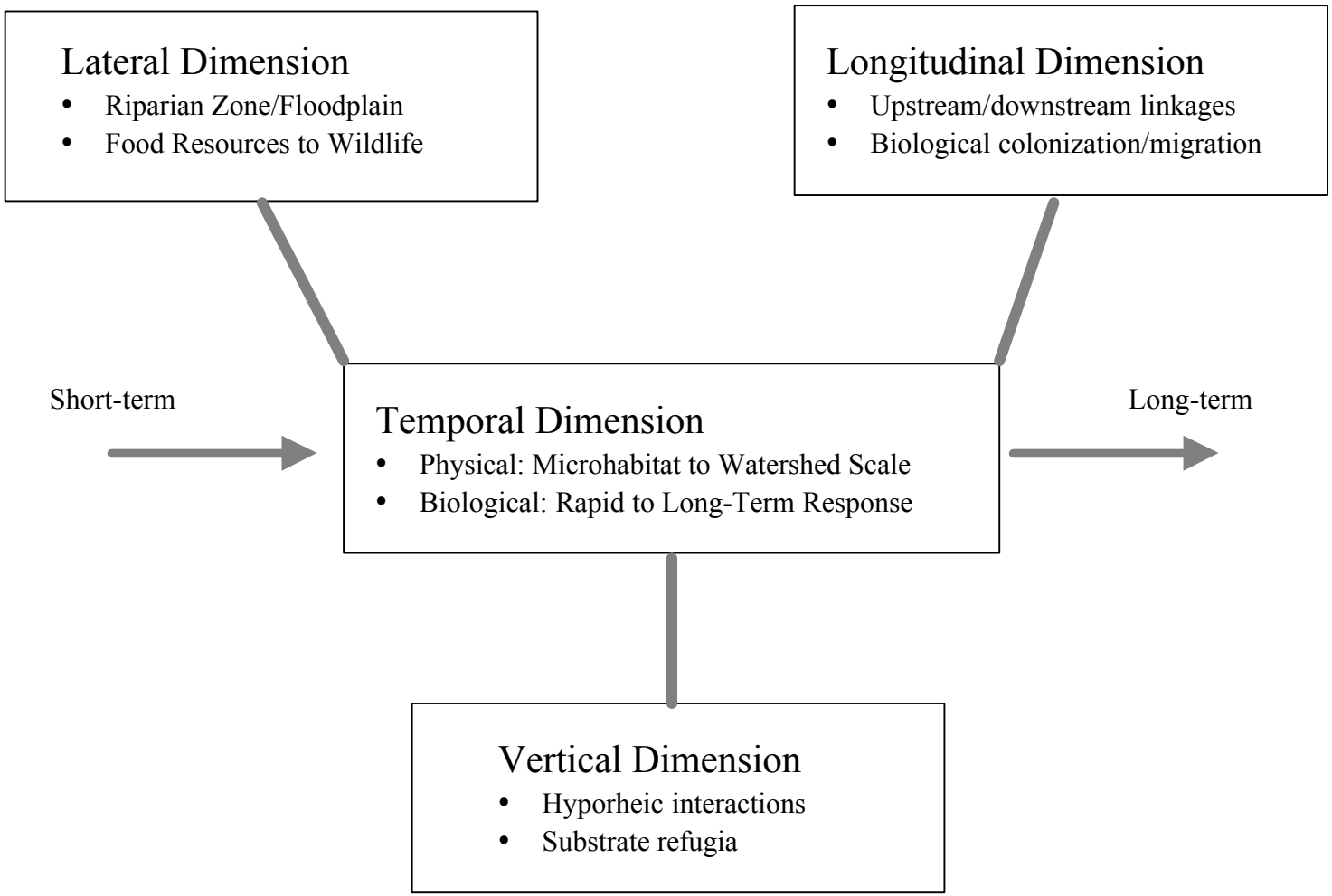
1.1.1 Conceptual Model of an Undisturbed Stream Ecosystem

1.1.1.1 Four-Dimensional Model of Stream Ecosystems

Patterns and processes in riverine systems have been summarized in the literature in varying ways (e.g., **Figure E-1**; Gregory et al. 1993; Resh and Rosenberg 1984). The four-dimensional template published by Ward (1989) provides a particularly useful approach upon which to describe characteristics of natural streams (i.e., naturally ephemeral, intermittent, and perennial waters) (**Figure E-2**). This template becomes particularly useful because it includes both spatial and temporal components, elements with important meaning in an arid environment.



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



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

Longitudinal Dimension

The concept of a longitudinal river continuum was introduced into the literature most explicitly by Vannote et al. (1980). Vannote et al. (1980) observed that most streams gradually changed in hydraulic properties in a downstream direction but that, within individual reaches, these gradual changes could be related to an equivalent change in the ecological communities within the stream. In this way, the river formed a continuum of biological change directly dependent upon downstream physical changes (Osterkamp et al. in press).

In addition to the physical longitudinal gradient described by Vannote et al. (1980), other longitudinal gradients exist as well, albeit often as a result of the physical template. Hynes (1975) emphasized the interconnected nature of watersheds and how changes in the watershed influence downstream biological communities:

> We may conclude then that in every respect the valley rules the stream…It is also clear that changes in the valley wrought by man may have large effects. Some are obvious…but others are very subtle. For example, even to replace mixed woodland with one species, or to fell one type of tree selectively could affect the litter regime and the supply of ions.

Because bottomlands are usually the most recent part of the physical landscape, a geological framework should be presumed. This framework will include the lithology of the valley, structural features (folds and faults), and a regional groundwater system. All of these characteristics can change with distance downstream. In most cases, a stream channel uncontrolled by external structures or 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: (a) longitudinal patterns in nutrient dynamics as nutrients are taken up, processed, excreted, or leached (Newbold et al. 1981, 1982a, 1982b); this pattern actually occurs in a spiraling fashion because of the downstream displacement that occurs as nutrients move downstream; (b) fish migrate longitudinally along a river continuum (e.g., see Lowe-McConnell 1987); and (c) an upstream-downstream linkage exists between the downstream drift of immature aquatic insects, often as larvae, and the dispersal of adult insects upstream following emergence (e.g., see Müller 1954).

Lateral Dimension

The lateral dimension consists of the relationship between the stream channel and the landscape 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.

Channel form and distribution within the floodplain is frequently complex and can grade from a single, incised, or stable channel to a braided system of several active channels separated by numerous vegetated islands or gravel bars. In the arid West, both deeply incised and highly braided channels can be found within the same stream. This apparent contradiction is seen because the streams can be locally modified by society on a shorter time scale than the regional geomorphic response. Many of the ephemeral, stormwater-dependent washes are highly braided (for example, ephemeral reaches of the Gila River), while streams that receive sediment-free wastewater are often incised.

At the surface, in a natural, unregulated river system, the floodplain, which is intermittently inundated as flood crests pass through the reach, is the region adjacent to the main channel. The predictability of flood events in the floodplain is somewhat related to the weather patterns of the region in which the river is found. However, because of dams and diversions, the natural flood cycle may be lost. Where this flood event still occurs with relative predictability, 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). Movement of biological organisms passively or actively onto the floodplain serves several purposes including (a) refuge from the flood event (i.e., rates of flow are greatly reduced on the floodplain where physical complexity works to "slow" the movement of water), and (b) access to additional food resources that become available during inundation. In addition, the lateral migration of water onto the floodplain serves to transport important nutrients to the river channel (e.g., the products of terrestrial decomposition processes occurring on the floodplain, especially following leaf fall, are picked up by flood events and transported to the river channel). Thus, the floodplain can serve as a "storage reservoir" for nutrients that enter the system in pulses following flood events (Cummins et al. 1983; Merritt and Lawson 1979).

In other areas of the West, rainfall is so scarce in a watershed that the return period of any flood-producing storm is more than a year. The course of the stream is marked by a braided pattern made up of numerous cut and abandoned channels. When flows enter such a wash, the activity of any particular channel can best be described as probabilistic and similar to models for alluvial fan flow (see numerous publications used by the Federal Emergency Management Agency [FEMA]). Obligate upland plants may dominate such systems, often to higher density than adjacent desert, with only occasional representatives of riparian species.

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 to irrigate plant life. The effects of channel distribution on species selection have been discussed by Bacon et al. (1993) who found that interconnected shallow (i.e., braided) channels covering 10 to 20 percent of the forest floor were as efficient as sheet flooding in establishing riparian stands of river red gum. More recently Bagstad and Stromberg (2001) have looked at 18 sites along the San Pedro River in Arizona and report that beta diversity can be directly correlated with geomorphic complexity during times of flooding. A number of studies have demonstrated systematic but complex changes in soil moisture that vary in a lateral direction across floodplains (Busch and Smith 1995; Cuomo 1992; Silvertown et al. 1999) and this may be the linkage between geomorphic complexity and selection in these cases. In other communities, depth and timing of inundation maybe the determining factor (Levine and Stromberg 2000; Stromberg et al. 1993).

Johnson and Lowe (1985) developed a model for a "transriparian continuum," a gradient in soil moisture from the stream to the upland. Although this concept ignores the smaller-scale, lateral complexities of soil texture, porosity, and saturation, the idea is basically sound. Aquifer or bank storage discharge serves several important functions including support of baseflow and nutrient inputs. During baseflow, areas of the river shoreline that receive aquifer discharge can be identified because of the mats of vegetation that develop (e.g., algae or macrophytes such as watercress) (Ward 1989).

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 two kilometers 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. The significance of this transfer of energy from the stream to the adjacent terrestrial ecosystem has not been well studied. However, a key study conducted in a perennial desert stream in Arizona, Sycamore Creek between Phoenix and Payson, demonstrated the potential importance of this event to terrestrial fauna (Jackson and Fisher 1986). In this study

it was shown that less than 4 percent of the biomass of emerging insects returned to the stream (in the form of dead insects or newly laid eggs). The remaining 96 percent of the biomass resulted in a net export of 22.4 grams per square meter per year of biomass or food to terrestrial insectivores (e.g., birds and bats).

Vertical Dimension

The vertical dimension primarily describes the interaction between the 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 recharging 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. For example, bedrock and deposition of fine inorganic and organic materials can limit interstitial porosity while deposition of large substrate materials (e.g., gravel and cobble) increase porosity and thus potential for interactions between surface and subsurface flows (Stanford and Ward 1993). Although we discuss the hyporheic zone in terms of the vertical dimension, it is apparent that this zone underlying rivers has its own lateral and longitudinal components, the extent and characteristics of each are dependent on substrate and flow patterns (White 1993). As such, Gibert et al. (1990) referred to the size of the hyporheic zone as "elastic" over time.

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). The hyporheic zone is also the most important heat sink for the system, as cooler, subsurface water exchanges with water from the stream surface (Morrice et al. 1997; Poole and Berman 2001; White et al. 1987; Wondzell and Swanson 1996).

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, 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).

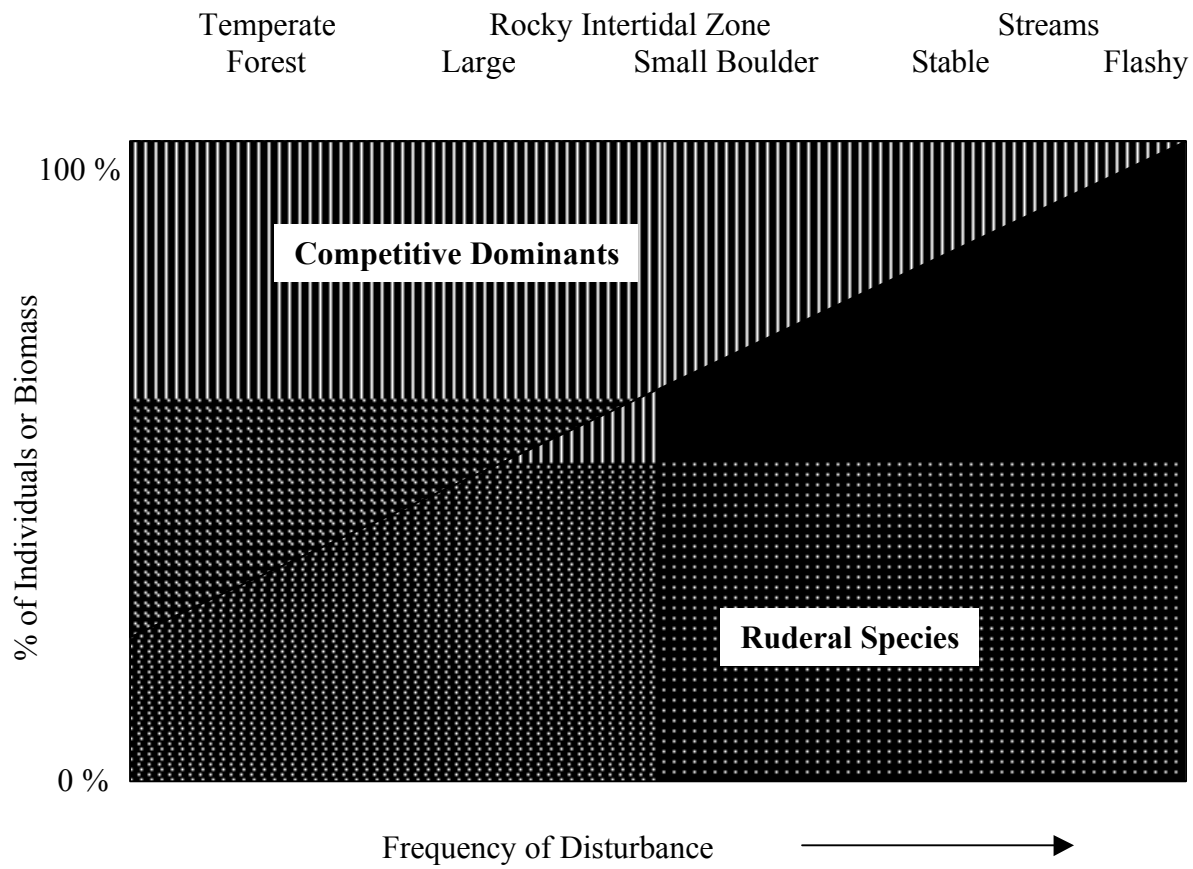
Physically speaking, in the unmanaged environment, surface water channels owe their geometry not to the "100-year flood" or even the "10-year flood" but to the more common flood events with sufficient tractive energy to move and redeposit the dominant (e.g., D50) particle size of their channel. These "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 concept of bankfull flow may be more elusive. Work by Moody et al. (in press) at Northern Arizona University (NAU) has calculated bankfull flow for 55 channels in Arizona and New Mexico. The NAU research team produced curves to relate drainage area to bankfull flow and calculated recurrence intervals of between 1.1 and 1.8 years for channel-maintaining events. These estimates are similar to the range reported by Leopold (1994).

However, there are difficulties in extending this concept to arid environments. In a system that conveys only storm events, the active channel may change with each event. In this case, measuring bankfull stage may only measure the last event at a particular point. In fact the concept of bankfull discharge has never been adequately defined for ephemeral streams. Despite this, the 1.5-year storm does represent the most common large annual event for a reach and provides a useful point of comparison to non-arid stream systems. In this sense, it may be a good indicator of one scale for flooding in a stream ecosystem.

It has been suggested that the frequency of disturbance can have a significant impact on the types of aquatic species that can be expected at a given site. For example, Townsend (1989) hypothesized a general biological relationship between the frequency of disturbance and the relative abundance or biomass of competitive dominant species and ruderal species (also known as fugitive or weedy species—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 ruderal species is expected to increase (**Figure E-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).



Hypothetical Relationship Between Frequency of Disturbance and Relative Representation of Species. Adapted from Townsend (1989).
Figure E-3

A review of the literature by Wallace (1990) and Canton (2000) 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). Because the habitat changes differentially affected different aquatic insect groups, Molles (1985) observed a variable recovery period of 60 days to 2 years depending on the aquatic insect species.

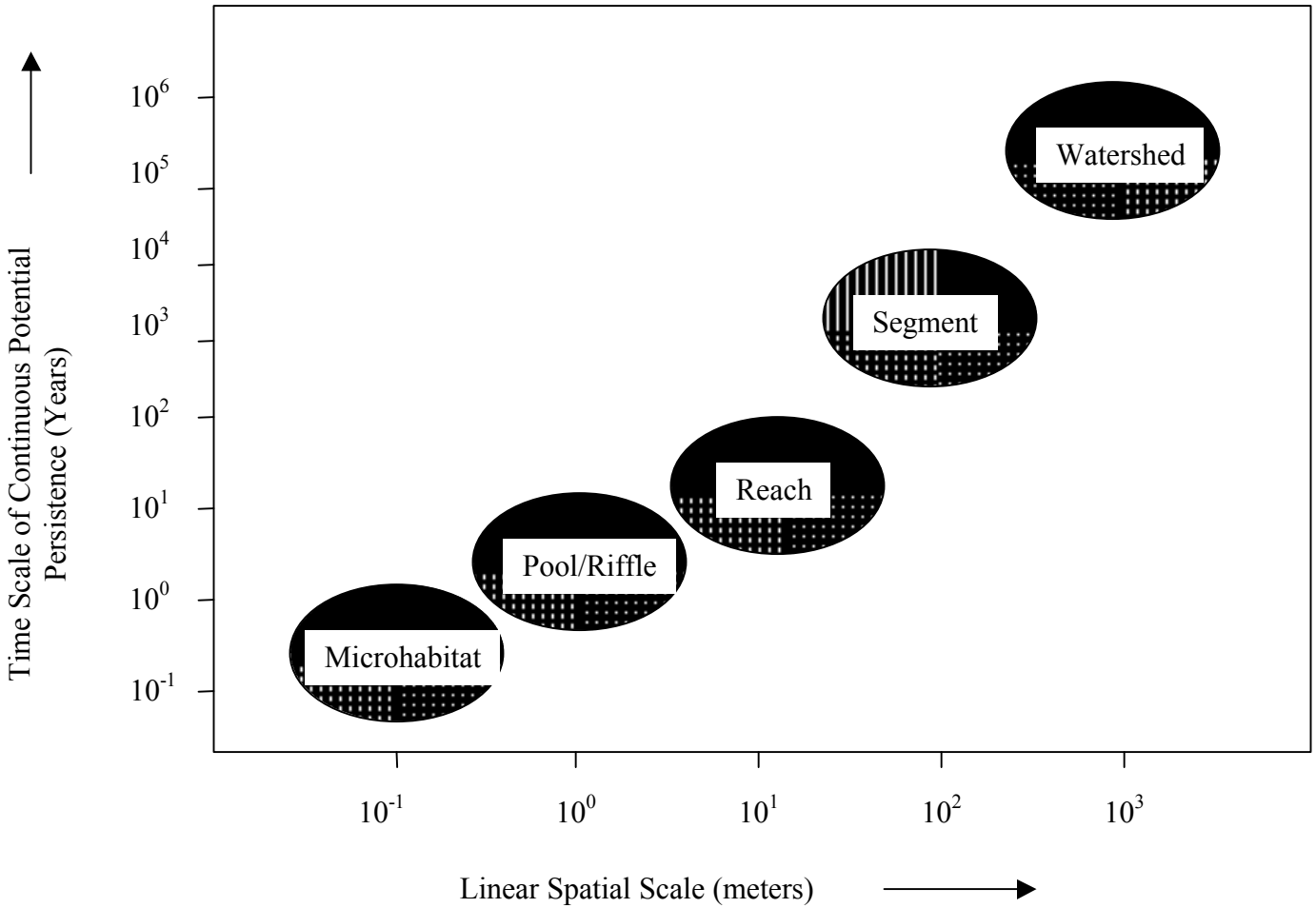
These observations on recovery periods were obtained from studies conducted on streams where the flood disturbance was generally limited to the active channel. Other studies have shown that in disturbances where the channel, as well as the adjacent floodplain, is impacted, response times can be somewhat 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 Mount St. Helens. After 10 years, the aquatic invertebrate community had still 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. Frissell et al. (1986) described this relationship for typical second and third order Cascade Mountain streams by relating the persistence of habitats at different spatial scales (**Figure E-4**). At the smallest spatial scale, the microhabitat (e.g., a linear patch of stream approximately 0.3 foot in length) and pools and riffles typically persist 1 to 10 years. A typical stream reach (approximately 30 feet) persists for 10 to 100 years, and a stream segment (approximately 325 feet) persists for 1,000 to 10,000 years. At the upper end of the scale, the watershed (mainstream length approximately 3,280 feet) persists for >100,000 years. Understanding this relationship 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).

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.



Relative Temporal and Spatial Scales of Habitat Systems of Second and Third Order Mountain Streams. Adapted from Frissell, et al. (1986).
Figure E-4

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. Where physical gradients change both horizontally and vertically, habitat is varied and species diversity may be large. Alluvial landforms along channels may vary in structure, shape, size, and age and generally do not necessarily reflect prevailing hydrologic conditions, thereby yielding complex down-valley habitat gradients that are responses to position (reach conditions), processes of formation, and flood magnitudes. Changes along the physical gradient can by nature be abrupt rather than gradual.

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.

1.1.1.2 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 water body.

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 not unusual 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, etc.

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 (Harris 1986; Hupp 1983; Osterkamp and Hupp 1984, 1985; Turner 1934; Sigafoos 1961). Harris (1986) provides a good review of the dependence of riparian microhabitats on floodplain structure.

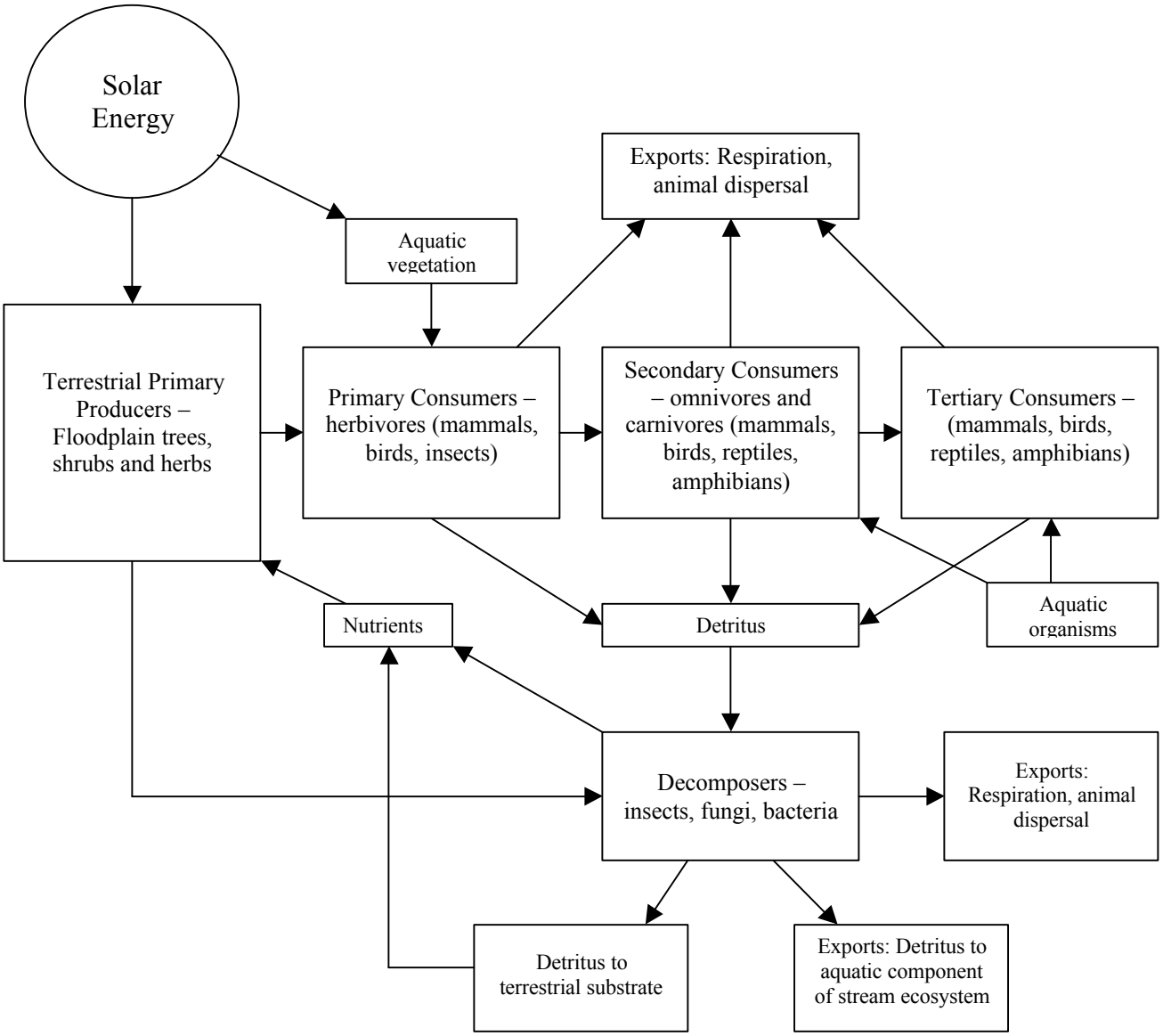
The abruptness of riparian boundaries can also extend to the subsurface, which can be exacerbated by excessive groundwater pumping. A particularly instructive series of observations was made during the early 1990s Arizona Supreme Court deliberations over the definition of subflow (Goodfarb 1994), a legal term broadly equivalent to the hyporheic zone in perennial to intermittent rivers. Several environmental activist groups challenged the impact of groundwater pumping on the riparian communities of the San Pedro River. Prior to issuing its opinion, the Court examined expert testimony, photographic records, and field evidence on the variations of geomorphic structure, vegetation patterns, and changes in subflow along the San Pedro River from the early part of the 1900s to the present. Historic photographs supplied by local ranchers indicated that profound changes in riparian vegetation had occurred during the time that groundwater pumping had increased in the Sierra Vista basin along the San Pedro River. Based upon the scientific evidence, the Court concluded that changes in sediment texture along the river created variations in permeability within the floodplain significant enough to laterally confine subflow. Vegetation patterns could be relied upon as one indicator of shallow groundwater distribution.

1.1.1.3 Biological Community Structure and Function

The structure and function of biological communities depend up on the physical and chemical characteristics of the stream ecosystem. Given the four-dimensional aspect of the ecosystem, there are many potential pathways for energy transfer and material cycling. The basic concepts of community structure and energy flow through the stream ecosystem and nutrient cycling within the system are covered in detail by any basic ecology textbook (e.g., Giller and Malmqvist 1998; Resh and Rosenberg 1984; Smith 1990). These concepts are briefly described in the following paragraphs.

Terrestrial Component of the Stream Ecosystem

Figure E-5 provides a typical energy flow diagram for the terrestrial component of the stream ecosystem. The primary producers in the terrestrial environment are green plants in the tree, shrub, and herbaceous layers. Rooted emergent vegetation, such as cattails, is considered part of the aquatic portion of the ecosystem. Solar energy and to lesser extent, flowing water, provides the primary source of energy for vegetation. Additional energy and material inputs include organic material and chemical nutrients taken up through the root systems from the substrate and topsoil. Energy is transferred out of the primary producers through consumption by the primary consumers. Vegetation respiration accounts for significant energy losses from the terrestrial



Energy Diagram for Terrestrial Component of Stream Ecosystem
Figure E-5

environment. Dead plant material, including leaves from deciduous trees, shrubs, and annual herbaceous plants, provides an energy source for the decomposers. Detritus from the vegetation also provides an energy source for the aquatic portion, particularly where deciduous trees overhang the stream channel. Airborne dispersal of seeds, leaves, or other organic material is another means of energy transfer.

The primary consumers are the terrestrial herbivores that feed directly on the foliage, fruit, seeds, or woody parts of the vegetation. This guild includes a wide variety of insects, grazing or browsing mammals feeding on foliage, rodents, and other small mammals feeding on foliage and seeds, and birds feeding on fruit and seeds. The primary energy source for this guild is the production from the primary producers. Some terrestrial species are also able to consume rooted or floating aquatic vegetation. Animals in the primary consumer group provide the main food and energy resource for the secondary consumers. Feces and dead bodies of animals in the primary consumer group provide an energy source for the decomposers. Energy losses from this set of animals are primarily through respiration. Dispersal of animals in this group could be a gain or loss of energy in the system, depending on the net direction of movement.

The secondary consumers in the terrestrial environment are the insectivores, carnivores, and omnivores. This guild includes predatory insects and other arthropods, insectivorous and carnivorous mammals, birds, reptiles, and amphibians. Omnivorous mammals and birds may be either primary or secondary consumers, depending on their food consumption, which may vary daily, seasonally, or opportunistically. The primary energy source for this guild is direct consumption of primary consumers. Some secondary consumers can also consume primary or secondary consumers from the aquatic environment. Some secondary consumers may be consumed by higher trophic levels (e.g., tertiary consumers). For example, hawks could consume insectivorous birds. Feces and dead bodies of animals in the secondary consumer group provide an energy source for the decomposers. Energy losses from this set of animals are primarily through respiration. Dispersal of secondary consumers could be a gain or loss of energy, depending on the net direction of movement.

The decomposer guild includes some insects and a wide variety of fungi and bacteria. These organisms are generally found in the upper part of the topsoil and in a layer of fresh organic debris, primarily leaf litter on the ground surface. Deciduous leaves, dead plants, animal waste, and dead animals are the main energy sources available to the decomposers. Airborne movement of plant material, primarily deciduous leaves and seeds, will provide some energy to this guild. Energy in the form of processed organic material can be transferred to the substrate at the lower levels of the topsoil, or it can be recycled through the root systems of the primary producers. Detritus from this layer can also be transferred to the aquatic component of the ecosystem by erosion during rainstorms. Primary and secondary terrestrial consumers could also consume some decomposers.

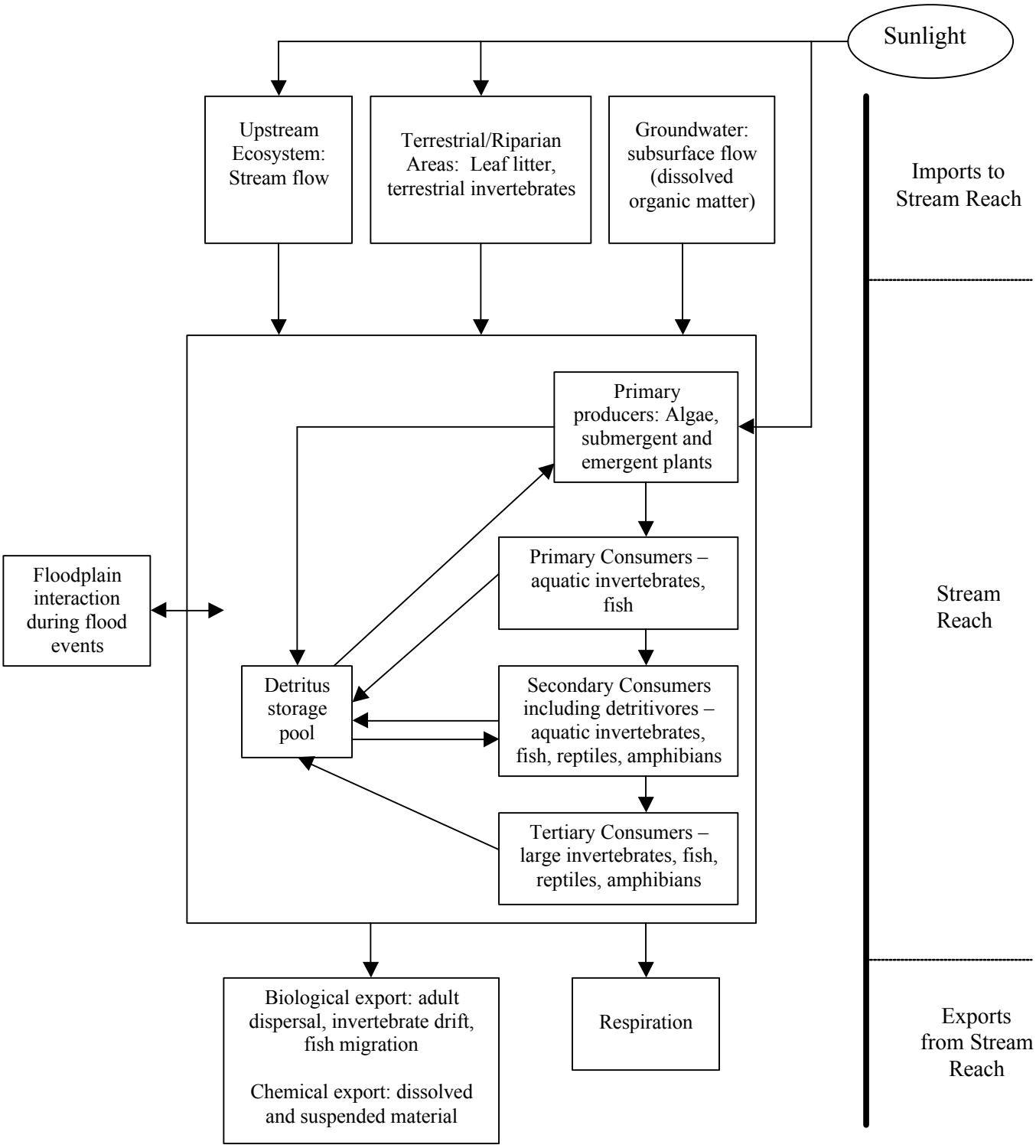
Aquatic Component of the Stream Ecosystem

This component has subsystems similar to those of the terrestrial component (**Figure E-6**). The primary producers are algae and green plants, including rooted and floating aquatics, and rooted emergent species. Solar energy provides the primary source of energy for these species. Additional energy and material inputs include dissolved and suspended materials in the water column. Nutrients and energy are also available from shallow sediments in the streambed by uptake through root systems. Energy is transferred out of the primary aquatic producers through consumption by the primary consumers. Downstream flow with dissolved and suspended material, including some living plant material, results in energy transfer from a particular point in the stream. Respiration by vegetation accounts for other energy losses. Some terrestrial species, such as ducks and coots, consume rooted or floating aquatic vegetation. Dead plant material, including leaves from deciduous trees, shrubs, and annual herbaceous plants, provide an energy source for the decomposers.

The primary consumers in the aquatic environment are the aquatic herbivores that feed directly on the vegetation, primarily algae. This guild includes zooplankton, a wide variety of insects, and fish and larval stage amphibians. The primary energy source for this guild is the production from the primary producers. These animals in turn provide the main food and energy resource for the secondary consumers. Feces and dead bodies of animals in the primary consumer group provide an energy source for the decomposers. Energy losses from this set of animals are primarily through respiration. Downstream flow in the stream channel can also remove some of these organisms.

The secondary consumers include predatory insects and other arthropods, fish, aquatic reptiles, and amphibians. The main energy source for this guild is direct consumption of primary consumers. As in the terrestrial environment, some secondary consumers may be consumed by higher trophic levels (e.g., tertiary level). Feces and dead bodies of animals in the secondary consumer group accumulate in the streambed and provide an energy source for the decomposers. Energy losses from this set of animals are primarily through respiration. Downstream flow could also remove some of these organisms.

The decomposer guild includes insects and bacteria in the shallow sediments in the streambed. Deciduous leaves, dead plants, animal waste, and dead animals are the main energy sources available to the decomposers. Energy in the form of processed organic material is utilized by detritivores, transferred downstream as fine particulate or dissolved organic matter or recycled through the root systems of the primary producers.



Energy Flow Diagram for Aquatic Component of Stream Ecosystem. Adapted from Giller and Malmqvist (1998), originally published by Fisher and Likens (1973).
Figure E-6

1.1.2 Conceptual Model of Effluent-Dependent Stream Ecosystems

1.1.2.1 Creation of Effluent-Dependent Waters

Effluent-dependent stream ecosystems are created by the discharge of treated wastewater into either naturally ephemeral or intermittent stream channels or streams channels that are ephemeral or intermittent as a result of hydrologic modifications in the watershed. The use of water for agriculture, power, and mining coupled with the rapid urbanization of the arid West has resulted in the need to dam existing rivers to provide a reliable surface water supply, caused the depletion of groundwater, and led to the importation of even greater quantities of water over large distances and elevations.

For this study, the majority of effluent-dependent waters result from the creation of stream ecosystems in hydrologically modified watersheds. Exceptions include Las Vegas Wash, and the Santa Cruz River at Tucson, where historically instream flows were intermittent or ephemeral. In cases where hydrologic modification has occurred, some of these modifications have been significant. For example, historically the Salt River was a major perennial tributary to the Gila River. However, with the construction of upstream reservoirs and the Granite Reef Diversion Dam, by the 1930s the Salt River had been converted to a dry riverbed (see Tellman et al. 1997 for additional examples).

As urban centers have grown the need to dispose of treated wastewater has grown as well. Various options exist to dispose of wastewater, including discharge to river channels, and in this regard the West is no different than any other part of the United States. However, because so many surface waters in the West are ephemeral or intermittent, the discharge of treated wastewater to such water bodies creates a new aquatic ecosystem that either replaces predevelopment baseflow or creates a new perennial water where only ephemeral flows may have previously existed. In addition, the discharge of effluent has created opportunity for either the creation of a new or the reestablishment of a prior existing riparian community along the stream channel. It is in this regard that the arid West is fundamentally different from non-arid regions.

Superimposed on the created effluent-dependent stream ecosystem are the climatic characteristics of the arid West. Precipitation, being lower in annual volume and often delivered by intense, monsoonal summer storms, limits western surface and groundwater resources. Stream flow, in particular ephemeral-channel flow, can be non-existent for years; when the rain comes, stream flow is often catastrophically dynamic (Baker 1977).

It is now thought that climate is only part of the story. The runoff patterns in the arid West may be more uniquely dependent upon soil and vegetative factors (Osterkamp and Friedman in press), rainfall/runoff curve relations (Hawkins and Khojeini 2000), or other surficial factors. Whatever the cause, the fact remains that stream flow hydrographs of arid West storms generally have steeper limbs (more “flashy”) than storms in non-arid areas, signifying the potential for more dynamic flooding.

Impressed upon these conditions is the persistent high pressure, low humidity, continental air masses, which produce higher evaporation rates than the eastern United States. The combination of high potential evaporation, steep rainfall-runoff curves, and low annual rainfall conspire to create a soil moisture deficit (Dunne and Leopold 1978), a condition in which the moisture level of the soil declines faster than the ability of precipitation to replace it. Soil moisture deficit does occur in the eastern United States, but usually indicates extreme conditions (drought). Most cropped plants require imported water to survive in the West because they are non-native; native plants have adapted well to arid conditions.

1.1.2.2 Physical, Chemical, and Biological Attributes of Effluent-Dependent Waters

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 Habitat Characterization Study, which documented changes that 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.

Physical Observation

To some extent, each of the 10 stream systems 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.

For the purposes of this study and following the model of Stanford et al. (1996), the natural physical equilibrium of a stream ecosystem would be a condition in which the distribution and abundance of plants and animals in the watershed were only determined by geological or climatic parameters. The routine disturbance of flooding within the basin would then be included within the natural equilibrium as a consequence of natural variations in climate. Also included in this stasis would be the very long period disturbance caused by tectonic uplift, seismic and

volcanic activity, and sea-level variations. Such forces can cause the erosion, sedimentation, and channel avulsion of streams that are relatively unaffected by human activities.

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 to 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: in 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. With flow loss, the channel tends to widen.

Many of the streams examined in the Habitat Assessment Study reconnaissance were incised into their floodplain. At the WWTP outfall, the Santa Fe River was incised several centimeters right at the point of discharge. The clear water of the treated effluent became turbid less than one kilometer downstream.

As discussed elsewhere, the discharge value corresponding to the 1.5-return interval storm can be estimated from a flood frequency analysis of the river gage data. Using these discharge estimates, the scale of the introduced flow can be compared to a value corresponding to annual floods for perennial streams. Ultimately, it may be necessary to develop a concept of "bankfull flow" for ephemeral streams (i.e., a measure of hydrologic and geomorphic stability that can be compared to the introduced effluent stream).

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 instream habitat and establishment of riparian vegetation. At most sites anthropogenic activities other than effluent discharge also 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 stormwaters 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 will also 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.

Water Quality Observations

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 instream quality is the same as effluent quality depends on how much instream 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 wastewater treatment facility. If upstream flow is present, the quality of the effluent will typically 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 instream water quality.

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

instream physical, chemical, and biological processes modify the chemistry. This is especially true for water quality parameters such as temperature, DO, pH, nitrogen, and phosphorus. For example, some degree of oxygen depletion can occur for some distance below the discharge point because of the high BOD of the effluent. The degree to which this oxygen depletion occurs depends on the level of treatment. Water temperature will likely be relatively stable near the point of discharge, but with increased distance downstream, water temperature will tend toward a more typical diurnal cycle.

Aquatic Biology Observations

The input of a continuous stream of effluent at a discrete point along an intermittent or ephemeral stream significantly changes the natural 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 would naturally 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 effluent (e.g., high BOD) is not particularly supportive of a diverse community of aquatic organisms, 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 new 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 towards 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 species richness may be low, biomass production could still be quite high in effluent-dependent waters, especially among aquatic groups such as oligochaete worms and chironomid midges. 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 utilized by terrestrial vertebrates as a food source.

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, and the Salt River in Phoenix, Arizona, Las Vegas Wash in Nevada, the 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 stand in contrast to 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; and Knopf 1985). Lizard densities have also 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 may also 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/downstream differences in streamside vegetation were less pronounced at the Colorado Springs (Fountain Creek) and Denver (South Platte River) sites. 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 was 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 are generally 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 multilayered 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 may also 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. They 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.

1.1.2.3 Arid West Effluent-Dependent Stream Ecosystem

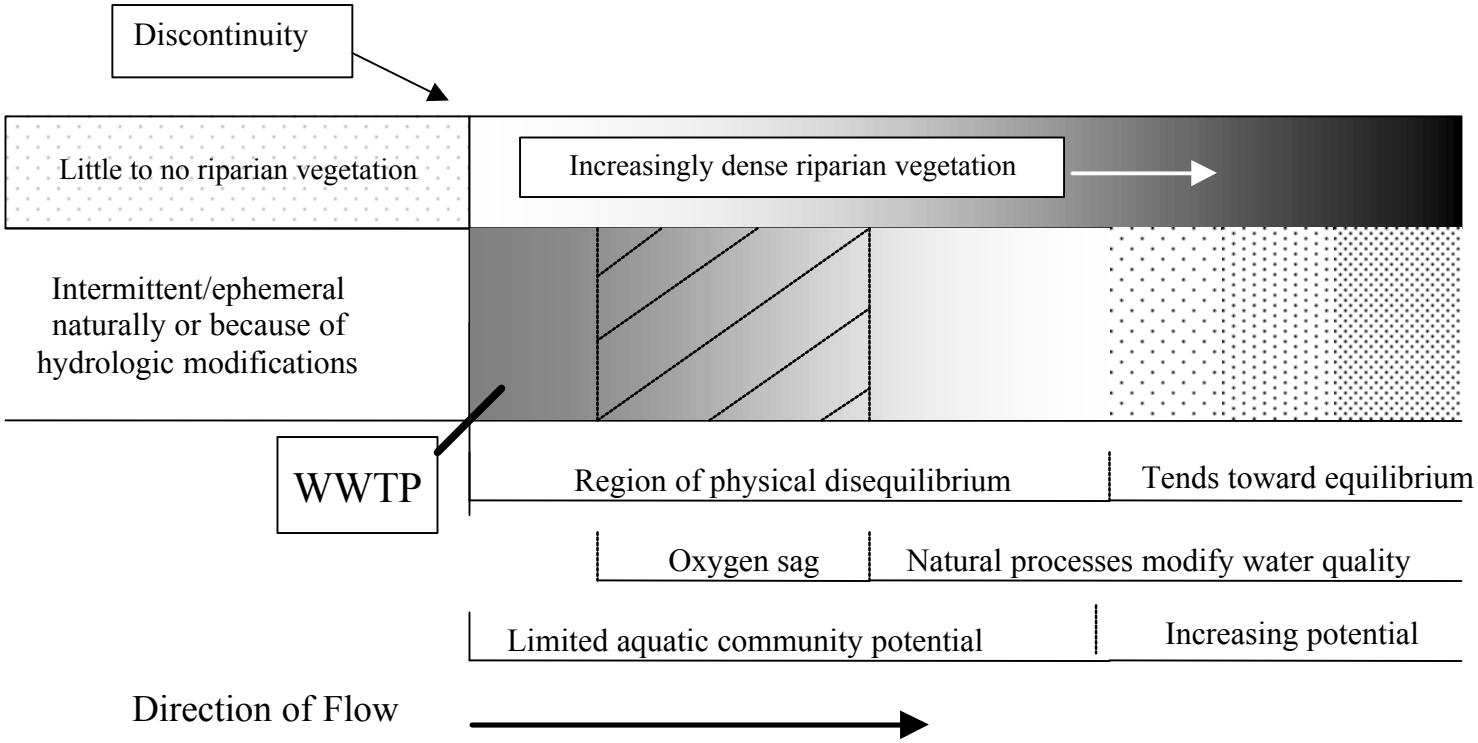
Stream ecosystems naturally achieve an equilibrium that is closely linked to the physical characteristics in which they exist. The discharge of effluent to naturally intermittent or ephemeral streams represents a discontinuity resulting in a disruption to the natural equilibrium as it exists at the time the discharge begins (**Figure E-7**). 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 E-7 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 E-7** 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).

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 was generally limited to what was 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.



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 towards physical equilibrium.

Conceptual Model of an Effluent Dependent Stream Ecosystem. Tendency Towards State of Physical Equilibrium and Increased Biotic Potential Assumes No Additional Anthropogenic Stressors Influence the Created Ecosystem.
Figure E-7

1.1.3 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 achieve 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 considerations.

Of course, limitations imposed on the biological community by the chemical characteristics of effluent cannot be discounted. Arguably, increased levels of treatment, resulting in improved effluent quality, will result in an improved 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.

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 a new equilibrium.

Although the aquatic system may be limited by habitat and water quality considerations, the terrestrial community is only limited by factors associated with habitat (often temporary) and non-native species. As indicated above, the contrast between above and below the effluent discharge can be significant, and supporting greater vegetative diversity will provide increased benefits for other terrestrial species.

1.2 INTRODUCTION TO INTEGRATED DATA ANALYSIS METHOD

Streams are often characterized by their biological, physical, and chemical characteristics. There are a multitude of variables that can be measured on any stream. Often, we wish to know what variables are important in determining the behavior of a variable of interest (e.g., How do the physical and chemical characteristics of a stream affect fish species richness or invertebrate abundance?). Determining which variables to measure can be a daunting task. In order to begin identifying regional patterns that allow for simplified biological assessment, analyses must be performed on comprehensive data sets to see if biological patterns follow predictable changes as the result of physical or chemical processes.

With the increasing popularity of multimetric approaches for biological assessment of streams and rivers, a suite of metrics is often measured that are hypothesized to vary predictably with disturbance. The applicability of this approach has been questioned in the arid western United

States where the major assumptions associated with many metrics have not been rigorously tested. However, it is still necessary to identify what physical and chemical factors are the most influential in structuring biological components in the West.

The goal of this short example is to show that new tools are available to look at large data sets to help identify which variables are important in structuring different biological response variables in stream systems. These approaches may have utility in the arid regions of the western United States where many of the underlying theories of stream ecology were not developed or tested rigorously.

1.2.1 Study Sites and Methods

We present three examples with data sets from the western United States. The first two data sets are from the Santa Ana River, California, and Fountain Creek, Colorado. Theses data sets are from a Water Environment Research Foundation (WERF) study that is using these two sites as part of a larger study to evaluate methods to discriminate what factors may limit the aquatic community of a particular waterbody (WERF study: *Ability to Discriminate Chemical vs. Habitat Limitations, Project No. 98-WSM-1*). The third data set contains measurements taken from the 10 arid West study streams and the four non-arid comparison sites including the following:

Arid West Study Areas

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

Non-Arid Comparison Sites

- Kansas River near Kansas City, Kansas

- Tar River near Rocky Mount, North Carolina
- French Broad River near Asheville, North Carolina
- Ararat River near Mount Airy, North Carolina

Data sets were comprised of various biological, physical, and chemical variables. Basic statistics were run on all variables in each data set and included count, mean, median, standard deviation, variance, minimum, maximum, range, interquartile range, skewness, kurtosis, and percentiles. These basic statistics were used to: (1) remove variables that were mostly non-detect, (2) detect and examine outliers, (3) detect and remove variables with low numbers of data points, (4) detect and remove completely redundant variables, and (5) check for normality. If skewness indicated that data were not normal, the data were log base-10 transformed for analysis with parametric statistics. If this transformation did not improve the skewness, untransformed data were used (i.e., at this time, no attempt was made to try other transformation techniques).

After any necessary transformations were completed, Principal Component Analysis (PCA) was used to analyze all sets of independent and dependent variables separately. Variable sets included habitat, chemical, and biological sets. For each set, components were examined for variables having values of 0.6 or higher in a rotated matrix—considered significant variables in a component. These variables were considered to be "moving" together and variables that were weaker in the component were removed. Any variable that did not have a score greater than 0.6 was removed from further consideration. A second PCA was then run using only those variables kept for further analysis. This was done for each set until a list of approximately six variables remained for each data type.

All combinations of dependent and independent variables were then considered together (i.e., each macroinvertebrate measure with chemistry, each macroinvertebrate measure with habitat, habitat and chemistry, etc.). This provides a means to better see how all sets of data interact. The strongest independent variables for each response variable were then identified. This procedure helped identify which biological response variables were the most responsive to the independent variables and vice versa.

Once the list of biological response variables and independent chemical and physical variables were narrowed down using PCA, a second technique was used—All Possible Regressions. This was run with all the remaining habitat and chemical variables together for each remaining response variable. The best model was chosen by examining the root mean square error plot from all possible regressions, which indicates the lowest variable count that explained the most variance. The variables with the highest R2 from the list of models with the appropriate number of variables were then chosen for the final variable list. From the list of models, the dependent variable was chosen based on responsiveness to the physical and chemical variables in the PCA and All Possible Regressions. As with PCA, this technique helps find variables that may be important in structuring the biological response variable previously chosen.

The biological response variable's relationships to the various chemical and habitat parameters were then analyzed using Chi-Square Automatic Interaction Detection (CHAID). 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. Each response variable was analyzed with the chosen habitat and chemistry variables from each individual data set chosen during the PCA and All Possible Regressions Analysis, and then with all variables from all the sets combined. Since CHAID is a non-parametric technique, no transformed data were used in the analyses, only the original raw data.

The output for CHAID displays a classification scheme in a decision tree format. The first variable listed after the dependent variable is the most important parameter. It branches out into nodes representing ranges of the independent variable that act similarly on the dependent variable. From the first independent variable, additional nodes may branch out from specific ranges indicating that an additional independent variable is important for a certain range of the first independent variable (i.e., contingent relationship). The PCA, All Possible Regressions, and CHAID analyses can be used to help identify the variables that are important parameters to measure for future monitoring or research efforts.

1.2.2 Preliminary Results

1.2.2.1 Santa Ana River

For the Santa Ana River, California, data set, 4 biological, 12 habitat, and 10 chemical variables were available for consideration. All variables were checked for normality, transformed if necessary, and carried forward into the PCA and All Possible Regressions analysis. PCA and All Possible Regressions demonstrated that fish abundance was the most responsive biological variable. All independent variables were carried forward for the CHAID analysis.

CHAID output is interpreted by following the nodes through the various tiers of output. Channel alteration scores were shown to be the most important for structuring fish abundance and broke out into three categories ranging from low channel alteration scores (i.e., highly altered channels) to high scores (i.e., little channel alteration). Fish abundance was highest at intermediate scores of channel alteration. At low scores (highly altered), a second level-interaction was found with flow scores important. In this case, lower flows had greater fish abundance than higher flows. At low flows, a third level interaction is present as substrate scores became important with stable substrates having higher abundance. For high flows, the third tier is represented by un-ionized ammonia, with lower levels of ammonia having greater fish abundance.

Continuing with the CHAID "tree," for intermediate habitat alteration scores, bank vegetation scores are the second tier variable. And in this case, when vegetative cover was high, copper concentrations were shown to be important (third tier), with abundances of fish being lower when copper concentrations were low.

Lastly, when habitat alteration scores were high (little alteration), flow was the next important variable with abundances being higher at higher flows. At lower flows, un-ionized ammonia was again shown to be important with lower levels corresponding with higher abundances.

As can be seen, the relationships of habitat and chemistry with fish abundance in the Santa Ana River are contingent on a number of variables. Ammonia appears to be important in structuring

the fish community, but only at sites with either highly altered channels with higher flow scores or unaltered channels with lower flows. Metals, represented by copper in this case, are important in moderately altered channels with good bank vegetation. This gives an idea of the complexity of even fairly simple systems like the Santa Ana River and highlights the usefulness of the new data analysis techniques like CHAID.

1.2.2.2 Fountain Creek

For the Fountain Creek, Colorado, data set, 178 variables were initially considered. After variables were checked for redundancy and normality, 44 were eliminated from further consideration. The remaining 134 variables were then analyzed using PCA. The PCA output resulted in 21 variables being carried forward for further evaluation. Of these, 10 were identified using all possible regressions and used in the CHAID analysis with the most responsive dependent variable (invertebrate taxa richness).

The variable that was shown to be the most important for structuring taxa richness was average embeddedness, which broke out into seven nodes/ranges. At the sixth highest range (87 to 97 percent), BOD_5 test was the second tier variable and was shown to be of importance at two ranges (2 to 7 and 7 to 15 mg/L). For the highest range of average embeddedness (100 percent), the presence of silt, clay, marl, muck, and organic detritus was shown to be important.

The mean taxa richness generally increased with decreasing average embeddedness but was not consistent in the upper ranges. Mean taxa richness decreased from 15.5 to 3.8, as mean embeddedness increased from the 63 to 68 percent category to the 100 percent category. For 100 percent embeddedness, the presence of any silt, clay, marl, muck, and organic detritus resulted in additional depression of taxa richness.

1.2.2.3 Arid West Stream Database

Initially data on 143 variables were compiled for the 10 arid West sites and 4 non-arid sites. Many variables, mostly chemical, had to be dropped because of lack of data for all streams. The remaining variables were then grouped into flow, chemistry, vegetation, habitat, and macroinvertebrate groups. Each group was analyzed individually with PCA to find important variables, and then each group was analyzed with other groups to look for patterns. A total of 107 variables were carried forward to use in CHAID analysis with the most responsive dependent variable (percent of abundance that are Ephemeroptera, Plecoptera, Tricoptera (EPT) taxa).

The variable that was shown to be most important was the 2-year flood flow variable or the calculated 2-year flood for each of the streams. Five separate ranges were identified. At the lowest range (0 to 200 cfs), the CDM habitat diversity score for pools was shown to be important at 2 ranges (scores of 0 to 3, representing low to moderate diversity, and 4 to 5, representing high diversity). At the second range of the 2-year flood, a second tier was represented by another habitat variable. The habitat variable was an EPA qualitative score related to bank stability. For the third range of the 2-year flood, the second tier was split by a terrestrial riparian habitat score developed by URS. The fourth range shows that a difference in treatment levels is important (secondary treatment or upstream of the WWTP).

Percent of abundance that are EPT taxa did not show a consistent pattern with changes in the 2-year flood variable. Percent of abundance that are EPT taxa increased dramatically from the first flow category (0 to 200 cfs) to the second (200 to 5,000 cfs), dropped for the third category, increased for the fourth category and then decreased for the highest flow category. For the lowest flow category, the habitat diversity (pools) was important, as percent of abundance that is EPT taxa generally increased with improving habitat diversity (as measured by the CDM habitat score). For the second flow category (200 to 5,000 cfs), higher bank stability (as measured by EPA score) did have a corresponding increase in percent of abundance that is EPT taxa. For the third flow category (5,000 to 8,000 cfs), as terrestrial riparian habitat score increased percent of abundance that is EPT taxa decreased. For the fourth flow category (8,000 to 11,830 cfs), there was a slight increase in percent of abundance that is EPT taxa for WWTPs with secondary treatment versus upstream of the WWTP.

1.2.3 Discussion

The results from our three examples show that the stream biota will often be structured by varying physical and chemical variables. The degree of channel alteration was an important variable for the Santa Ana River in California, for describing fish abundance, while embeddedness of fine particles was the most important variable in describing macroinvertebrate taxa richness for Fountain Creek, Colorado. It is important to note that the Santa Ana River data set was comprised of many qualitative scores and may represent more quantitative measures from other data sets. The Santa Ana River data set also illustrates the complexity of interactions that may be taking place at varying levels of these important parameters. Simply compiling data does not help tease out these interactions.

The results from the more regional, but less complete, data set show that the measurement of 2-year flood was the most important variable. This indicates that flow is an important variable when characterizing arid systems. More data, particularly chemical data, would make the regional analysis more robust. In addition, a validation data set is needed for comparison purposes. Therefore, additional data would need to be collected.

Often the biological response variable (e.g., abundance, richness, etc.) appears to be behaving in ways that are not intuitively obvious between categories of the physical and chemical variables. This behavior requires that we think more deeply about the variables that were measured and how they were measured. These techniques provide a way to begin discovering what processes are important in structuring the patterns observed in streams.

This preliminary analysis simply shows that new and innovative ways exist to look at data sets to help identify patterns and processes in streams. These techniques offer new ways to look at relationships between chemical, physical, and biological variables in streams in the western United States. Many of the relationships assumed to be true in multimetric, single-score biological assessment approaches have not been adequately tested in the western United States. This analysis does not attempt to advance ecological theory in streams per se, but introduce more innovative thinking to allow for such advances.

1.3 HABITAT CHARACTERIZATION SCHEME

1.3.1 Introduction

The primary purpose of this study was to characterize the physical, chemical, biological, aquatic, and terrestrial attributes of effluent-dependent streams in the arid West. In addition, data were to be evaluated in an integrated manner to determine whether these arid West systems were sufficiently different to warrant an alternative approach to the development and implementation of water quality standards in these systems.

The previous section provided an overview of the characteristics of a natural stream ecosystem and summarized how characteristics of 10 effluent-dependent waters studied for this project differ from natural streams. With that discussion providing a foundation, in the following sections we will provide a synthesis of the findings from this study and discuss how these findings suggest the need for an alternative approach to regulating water quality in effluent-dependent waters.

1.3.2 Key Findings Regarding Effluent-Dependent Water Habitats

Through the course of integrating historical and site reconnaissance data from the 10 study areas investigated for this study, the following key findings are evident:

- The addition of flow via effluent to alluvial stream systems can result in channel incision and poor instream habitat for aquatic organisms.

- The physical habitat of an effluent-dependent water is determined not just by the physical dynamics resulting from the effluent discharge, 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 in the river system likely influences biological expectations—aquatic and terrestrial.

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

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

- Riparian terrestrial characteristics are a reflection of the physical template resulting from instream flow characteristics (natural and effluent-driven).

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

It is notable that, for the most part, these findings are related to flow, which influences habitat characteristics. These findings are also consistent with results of the CHAID analysis for the specific sites (i.e., Santa Ana River and Fountain Creek) where habitat limitations were found to be important (see Section 5.2).

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. Two of the best examples of this disequilibrium were observed at the Santa Fe River and Las Vegas Wash sites. At both of these sites, the effluent discharge volume exceeds the natural bankfull flow of the channel and the net result is significant channel incision.

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. The imposition of non-effluent related stressors on a stream trying to establish a new equilibrium in response to the initial discharge results in an unstable physical environment, often for some distance downstream of the original discharge.

In addition to the physical changes associated with effluent discharge, the influence effluent water quality will have on instream water quality can vary depending on several factors including (1) the frequency, duration, and magnitude of upstream flows; (2) other flow sources, (.g., from tributaries or agricultural return flows); and (3) instream 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 instream water quality varies depending on site-specific factors.

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 study sites 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 expected. The terms "habitat" and "water quality" are broad terms, which encompass a lot of specific factors, 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.

The results of this study suggest that there are limitations to what is attainable in situations where the effluent comprises the majority of flow in a watercourse. A created stream ecosystem is not the same as a natural stream ecosystem, and any expectation that the biological community of created stream should have the same potential (i.e., reach the same level of attainment) as a natural stream appears misplaced. This finding appears to apply especially in situations where 100 percent of the baseflow is derived from effluent.

1.3.3 Habitat Classification Scheme

An objective of this study was to establish a habitat classification scheme for arid West ecosystems. This objective was based on the assumption that it would be possible to use the results of the site characterizations to classify stream types according to site-specific characteristics. However, the results from this study suggest that a more appropriate approach for habitat classification would be to focus on the establishment of a separate designated use for effluent-dependent waters rather than attempt to classify specific habitat types on the basis of the 10 study sites. This shift in thinking is based on the finding of common physical, chemical and biological characteristics associated with the stream ecosystem created by the discharge of effluent.

While there are certain differences among sites with regard to the dominant riparian species or physical characteristics such as geology, these differences do not lend themselves well to the existing regulatory framework. Moreover, the current study evaluated only 10 sites and the degree to which a classification based on these 10 sites would be applicable to other effluent-dependent sites in the arid West is unknown. Ultimately, site differences, which may be important with regard to the establishment of water quality criteria, could be addressed by the development of site-specific criteria. While a site-specific approach is always available, this approach would ignore the common characteristics of effluent-dependent waters identified by this study.

1.3.4 Modified Stream Subcategory

1.3.4.1 Measures of Attainment

The establishment of beneficial uses forms the cornerstone for the implementation of the Clean Water Act. Evaluating whether a surface water is in compliance with Clean Water Act provisions is based entirely on whether the beneficial uses established for that water body have been attained. Therefore, establishing appropriate beneficial uses and determining the means by which one measures attainment of those uses is a critical step in the process of establishing water quality standards.

Beneficial use attainment can be measured in many ways including through the use of surrogate measures. For example, states typically establish fecal coliform or *Escherichia coli* criteria as indicators for all pathogens that may be present in a surface water. Chemical-specific criteria, while directly measured, are really indirect measures of attainment because there is typically no direct cause and effect observed between the measurement and the use it was established to protect. Biological criteria are direct measures of aquatic life, but its use depends on correctly establishing what is attainable for a given water body.

States establish criteria as a means to measure attainment and ensure that the water body under question functions correctly to meet its beneficial uses. This can be straightforward and direct for uses such as domestic water supply. Water that contains toxic substances above some maximum concentration cannot function as a water supply source for human populations.

The coupling of criteria to the beneficial use becomes more complicated when uses other than domestic water supply are considered. For example, agricultural uses of water include two categories (livestock water and irrigation) in most state water quality regulations, because the different function of water associated with these uses requires protection from different types of pollutants. An even more relevant but important distinction occurs between warmwater and coldwater aquatic life. For this category of use, the regulations are established to preserve a set of conditions that have been determined necessary for the proper function of the habitat, broadly defined to include both physical and chemical factors.

In ecology, the term “function” has a very specific definition. Functional properties are those that transfer energy, nutrients, or biomass through or out of the ecosystem. For example, the hydrologic function of the stream ecosystem has functional importance in each of these transfer processes, supplying water for terrestrial animals, temperature regulation for fish, biomass removal within the plant community, and so forth (see additional discussion on stream ecosystem function in Section 1.1). Each of these functions can be described with a set of parameters (defining the system) and variables (defining the condition of the system) that must remain within a certain range if biota can successfully use the ecosystem. If one was to understand each of these variables and how they function for the benefit of the stream biota, these variables could be established as standards and criteria to protect the aquatic life use that they define.

The results of the Habitat Characterization Study would suggest that according to criteria typically used to evaluate aquatic communities for the purpose of evaluating attainment, effluent-dependent systems would likely fail simply because of a lack of species richness or sufficient

diversity in certain groups used as indicators of clean water. Increased treatment levels appear to have resulted in shifts toward increased numbers of aquatic organisms that are considered more "acceptable" (e.g., mayflies and caddisflies), but even within these groups, richness remains low.

Lost in the discussion of whether or not the appropriate species are present is whether or not the ecosystem created by the effluent discharge is properly functioning. It may be more important to consider this aspect of evaluating use attainment than focusing on measures that are strictly associated with the well being of specific species. For example, biological endpoints that focus only on the aquatic resource rather than other elements of the stream ecosystem (e.g., riparian habitat, food resources for wildlife, and wildlife migration corridors) may be overly limiting and shortsighted. These other biological endpoints may be functioning well regardless of the species richness or composition of the aquatic community.

1.3.4.2 Creation of a Modified Stream Category

Federal regulations recognize at least three basic stream types: Perennial, intermittent, and ephemeral (see 40 CFR 131.10(g)(2) and discussion in **Appendix J**). This recognition is also discussed in federal water quality standards guidance (e.g., EPA 1994) and often acknowledged in state water quality standards. However, although these stream types are routinely acknowledged by definition in federal guidance or state regulations, rarely is guidance provided or criteria established that are specific to these different stream types (see summary of arid semi-arid West state water quality standards in **Appendix K**).

The results of this study suggest that there is a scientific basis for consideration of an additional stream type that is as distinct as a perennial, intermittent, or ephemeral stream. This additional stream type can be referred to as a "modified stream," (i.e., a stream, which is naturally intermittent or ephemeral, where effluent is the primary or only source of non-stormwater flow). These streams are "modified" in the following key ways:

- The discharge of effluent may result in erosion and channel incision, creating an aquatic habitat with decreasing potential nearer the point of discharge.
- The flow is primarily or completely effluent during much of the year and the instream water quality should be expected to be similar to the quality of the effluent.
- Urban expectations for flood and erosion control may nullify the expected gains from improved treatment levels; expectations for returning the watercourse to a natural or predisturbance condition are low or non-existent;
- Other sources of discharge to the stream (e.g., stormwater outfalls and agricultural return flows) influence the stream in their own ways.
- Water resource projects (e.g., diversions, upstream storage reservoirs) have significantly affected the flow regime of arid West streams, especially in urban areas.

While any one of the above ways in which a stream is "modified" is sufficient to impact the aquatic community, the reality is that these modifications often occur in various combinations.

The fact that a combination of factors simultaneously influences these watercourses raises important questions with regard to how water quality standards are established and implemented in modified systems. It is telling that upgrades in treatment appear to have resulted in little change in taxanomic richness; only changes in the types of organisms present have occurred. Therefore, the ratio of increased numbers of taxa versus cost of treatment upgrade is low, which offers the question: Was appropriate biological community response gained for the cost expended?

If taxanomic richness remains low regardless of treatment, what would be an appropriate measure for defining attainment in a modified stream? Simply upgrading a wastewater treatment facility does not appear to be sufficient to produce a water body that meets expectations for a non-impacted aquatic community (based on traditional measures of what constitutes an unimpacted community). Even with improved water quality (e.g., reduced nitrogen) resulting from more costly forms of treatment, the aquatic community does not necessarily become more like a "natural" community. Regardless of this fact, water quality programs continue to focus on treatment facilities as the primary answer or means to achieve aquatic community expectations. In so doing, water quality control efforts have been focused on only specific components of the stream ecosystem rather than looking at the big picture (i.e., the need for a watershed approach that addresses other limiting factors such as habitat).

1.3.5 Alternative Regulatory Approach for Modified Streams

The recognition that effluent-dependent waters, as modified habitats, may require a different approach to managing water quality has been recognized for almost a decade. In 1992, EPA Region IX published its Interim Final *Guidance for Modifying Water Quality Standards and Protecting Effluent-Dependent Ecosystems*. This was the first attempt to address these ecosystems on a specific basis by recognizing their uniqueness. This document also established the term "net ecological benefit," a concept that recognized the potential value of having wastewater discharged to an otherwise dry riverbed. (Note: For the purposes of this discussion it has been assumed that the addition of water to an otherwise dry riverbed is a "benefit." The authors acknowledge that preservation of naturally ephemeral channels can likewise be beneficial. However, in an urban environment such preservation may not always be practical.)

In 1993, EPA prepared, *Supplementary Guidance on Conducting Use Attainability Analyses on Effluent-Dependent Ecosystems*. The purpose of this document was to provide guidance to states on what constitutes net ecological benefit:

> An important consideration in developing water quality standards for effluent-dependent water bodies is that discharges may promote restoration of habitat suitable for formally indigenous species. Effluent flows may enable maintenance of certain forms of aquatic life, wildlife, and riparian or wetland habitat, even if water quality criteria necessary to protect fishable/swimmable uses are not readily attainable. Where water quality criteria are not met, State/dischargers may consider removal of the effluent instead of additional treatment. If removal of the effluent would cause more environmental damage than allowing it to continue, the discharger may be able to demonstrate that the effluent is providing a Net Ecological Benefit. Examples of such benefits include the following:

- Provision or enhancement of habitat or food sources for native or threatened/ endangered aquatic species, or migratory waterfowl

- Provision or enhancement of habitat or food for terrestrial native or threatened/ endangered species

- Enhancement or restoration of riparian values (e.g., increased species diversity, growth of vegetation and improved wildlife/bird habitat)

- Preservation of existing riparian or aquatic habitat that could not be supported without effluent flow

- Restoration of aquatic and riparian values lost due to human activities (prior to new effluent discharge)

- Enhancement of water quality resulting in conditions conducive to ecosystem restoration and/or preservation

- Improvement or creation of habitat capable of supporting fish or allowing migration of anadramous species

- Restoration of species diversity in aquatic ecosystems (page 1-3, EPA 1993)

It is notable that these examples of net ecological benefit are primarily focused on the ecosystem, rather than a specific component of the ecosystem. Moreover, several of the examples consider ecosystem functions (e.g., provision of food resources for terrestrial species).

The Habitat Characterization Study was initiated for the purpose of documenting the physical, chemical, and biological characteristics associated with effluent-dependent waters in the arid West. As a result of this data gathering effort, many of the elements of what could constitute a net ecological benefit have been observed at the project study sites (e.g., substantially increased terrestrial community supported by effluent, wildlife supported to some degree by food resources generated by the adjacent aquatic community, and enhancement and restoration of riparian values lost as a result of human activities).

The 1993 EPA guidance provided a list of demonstrations that must be made to receive approval of a use attainability analysis based on net ecological benefit:

- The water body is in a primarily arid area where aquatic resources are limited and ecologically valuable. The water body supports an ecologically desirable aquatic, wetland, or riparian ecosystem and supports native plant and wildlife species. For a new discharge, the water body must have the potential to support such an ecosystem.

- Effluent discharges may not produce or contribute to concentrations of pollutants in tissues of aquatic organisms or wildlife that are likely to be harmful to humans or wildlife through food chain concentration.

- The discharger documents that a feasible plan to remove the discharge is under consideration.

- The analysis demonstrates that a continued discharge to the water body has not created and is not likely to cause or contribute to violations of downstream water quality standards or groundwater basins.

- All practicable pollution prevention programs, such as pretreatment and source reduction, are in operation. The discharger verifies that it has responded appropriately to previous and ongoing compliance actions.

- In order to preserve the Net Ecological Benefits associated with the discharge, it is recommended that the discharger commit to providing effluent to the stream that is sufficient to protect and maintain the ecological benefit as determined by EPA, and state and federal wildlife agencies.

The regulatory framework within which the net ecological benefit concept was originally established envisioned the preparation of a use attainability analysis to justify the lowering of one or more water quality standards because one or more of the use attainability provisions at 40 CFR 131.10(g) had been met (typically 40 CFR 131.10[g][3]) (see **Appendix J** for additional discussion on water quality standards program requirements). This approach assumed that the designated uses of the effluent-dependent water were appropriately set in the beginning. Based on the findings of this study, it may be more appropriate to establish an aquatic life and wildlife beneficial use specifically designed for modified streams rather than use the traditional use attainability approach to evaluate attainment. This approach would be in accordance with EPA's water quality standards guidance, which states that:

> "…flexibility inherent in the State process for designating uses allows the development of subcategories of uses within the Act's general categories to refine and clarify specific use classes. Clarification of the use class is particularly helpful when a variety of surface waters with distinct characteristics fit within the same use class, or do not fit well into any category. Determination of non-attainment in waters with broad use categories may be difficult and open to alternative interpretations. If a determination of non-attainment is in dispute, regulatory actions will be difficult to accomplish" (EPA Water Quality Standards Handbook, Section 2.3, page 2-5).

Under this scenario, modified streams would be considered a separate class of streams with unique characteristics that "do not fit well into any category." Criteria would be developed that would allow states or tribes to categorize streams as "modified." The principles of net ecological benefit could be incorporated into this categorization to recognize the benefits of having water flowing in what would otherwise be a dry or mostly dry riverbed. These benefits would be identified and used as the basis for implementing a water quality management program that was focused on optimizing the overall benefits of the resource rather than the more traditional approach—focusing efforts on only one or a few components of the resource.

1.3.5.1 Establishing Criteria in a Modified Stream

The chemical-specific and whole effluent toxicity criteria established to protect the aquatic life and wildlife use in streams evaluated for this study were often no different than the criteria applied to nearby naturally perennial waters. Some locations had adopted site-specific or category-specific aquatic life criteria to address specific constituents (e.g., the South Platte River has site-specific DO criteria, and Arizona has established DO criteria) that are specific to waters designated as effluent dependent. No sites had adopted biological criteria. However, because of the prevailing view that similar chemical-specific and whole effluent toxicity criteria should be applied to both naturally perennial and effluent-dependent waters, one must assume that states expect the same biological result in the effluent-dependent water as nearby naturally perennial streams.

Regardless of the approach used to adopt aquatic life and wildlife criteria, the primary purpose behind these criteria is to support the establishment of a propagating biological community that meets some ideal. This ideal (e.g., measured as taxanomic richness or composition) has not been established for any effluent-dependent waters. Yet, levels of treatment continue to be improved to meet increasingly strict end-of-the-pipe water quality standards and the expectation remains that improved treatment will push the aquatic community toward the ideal. Unfortunately, as indicated by study results, increasing levels of treatment do not appear to be accomplishing any specific goal.

Criteria are established to protect the use and as such it is critical that uses be appropriately defined to ensure that the established criteria actually accomplish the intent. As discussed above, establishment of an aquatic life use envisions more than just numbers of species or species composition. It also envisions a functioning ecosystem that supports wildlife. While this concept is not often the explicit basis for the establishment of aquatic life criteria, it is contemplated in discussions associated with biological criteria and very importantly the net ecological benefit concept.

It is particularly notable that the net ecological benefit guidance as considered by EPA recognized the supportive role that created aquatic habitat could have on riparian and wildlife resources. This same guidance also noted some key limitations that must be considered before the benefits gained from improved riparian and wildlife resources could be acknowledged. These limitations included:

- Effluent discharges may not produce or contribute to concentrations of pollutants in tissues of aquatic organisms or wildlife that are likely to be harmful to humans or wildlife through food chain concentration.

- The analysis demonstrates that a continued discharge to the water body has not created and is not likely to cause or contribute to violations of downstream water quality standards or groundwater basins.

- All practicable pollution prevention programs, such as pretreatment and source reduction, are in operation. The discharger verifies that it has responded appropriately to previous and ongoing compliance actions.

These limitations are designed to ensure that (1) a minimum level of wastewater management and treatment is implemented, and (2) pollutants are not exported from the ecosystem through the food chain. The emphasis is on toxic chemicals, especially bioaccumulative chemicals of concern. Accordingly, from the standpoint of adopting appropriate criteria for modified stream systems, the focus should be on the management of these types of pollutants through pollution prevention and treatment.

1.3.5.2 Performance-Based Regulation

Performance-based regulation is the use of "success criteria" to define observable or measurable management or mitigation objectives for a given water body or watershed (Streever 1999). EPA Region IX's net ecological benefit guidance is an example of performance-based regulation. It accurately accounts for the dynamic forces that arise in response to more stringent permit limits. The policy abandons previous assumptions that "all things are equal" and "everything else will remain the same." Instead, the new policy causes all stakeholders to consider the most likely direct and indirect consequences of new chemical specific limits. More importantly, it emphasizes the need to measure success at the true endpoint: overall integrity of the aquatic ecosystem. However, the concepts identified in the EPA guidance have not been widely applied.

This study provides evidence that although the aquatic communities of effluent-dependent waters do not appear to achieve high species richness even with additional effluent treatment, other benefits of the effluent are obvious. Benefits directly measured include increased extent and density of riparian vegetation and increased numbers of wildlife species. Assumed indirect benefits include the creation of wildlife corridors for the migration of wildlife populations and the transfer of energy from the aquatic community to the wildlife community, especially birds and bats, through the emergence and dispersal of aquatic organisms. None of these benefits is currently considered when implementing water quality programs through the issuance of NPDES permits. Instead the focus is on chemical specific criteria with the assumption that more stringent criteria, and thus more stringent effluent limitations, will lead to a directly measured benefit to the aquatic community.

To resolve this apparent disconnect between the implementation of water quality standards and the apparent benefits associated with the presence of the discharge, it is appropriate that the implementation of water quality standards be approached differently for these created ecosystems. Human-caused conditions from all watershed activities limit ecosystem potential and if the focus remains solely on the level of treatment, considerable effort may be expended with little gain. Instead, if a net ecological benefit approach is used as a means to guide water quality management activities, with the goal being a more holistic approach (i.e., ecosystem management), additional benefits may be achieved.

Under a net ecological benefit approach, permittees and regulatory agencies could work together to define appropriate biological and chemical endpoints that (1) maintain the existing use, (2) ensure antibacksliding regulations are appropriately implemented, and (3) ensure that bioaccumulative chemicals of concern are not exported from the created ecosystem. This approach moves away from the tradition of implementing permits through effluent limitations, and instead provides the opportunity for dischargers and communities to work together to define ecosystem or watershed goals for the created habitat. These goals may still require implementing

flood control activities to protect the community, but implement such activities in a more ecosystem-friendly manner that considers other urban goals such as habitat restoration, recreational opportunities, meeting water rights obligations, or providing habitat for threatened and endangered species.

The use of alternative biological or chemical endpoints for evaluating attainment represents a performance-based approach for water quality management. Such an approach is not new to managing water quality under the Clean Water Act. Stormwater management and non-point source pollution control programs are examples of existing performance-based programs. Controls are based on the implementation of watershed-based water quality management strategies rather than specific effluent limitations. In addition, USACE is using performance-based outcomes to evaluate compliance under the Clean Water Act Section 404 program (Streever 1999).

1.3.6 Restoration of Modified Streams

Implementation of a performance-based approach to water quality protection in effluent-dependent waters would also support EPA efforts to guide Clean Water Act programs toward the concept of ecological restoration. EPA defines ecological restoration as "the restoration of chemical, physical and/or biological components of a degraded system to a predisturbance condition." A driving purpose behind an emphasis on ecological restoration is an increasingly important need to implement water quality programs on a watershed basis, especially where total maximum daily load concerns must be addressed. Water quality concerns that are difficult to implement numerically with effluent limitations (e.g., sedimentation, habitat integrity, and flow modifications) can be addressed through ecological restoration strategies that include addressing local habitat limitations.

Achieving a "pre-disturbance condition" will be difficult in an urban environment, especially given the number of arid West watersheds, which are hydrologically modified for the purpose of providing regional drinking water supplies and flood control. Moreover, as apparent from this study and the previously completed Discharger's Survey (WQRP 2000), many of the effluent-dependent waters are located in urbanized environments where concerns regarding flood control have frequently superseded concerns for maintaining natural habitat. As a result, habitat has already been fundamentally modified by factors unrelated to limitations associated with effluent quality.

Regardless of the restoration limitations imposed on many effluent-dependent waters, the fact remains that the presence of water in what would typically be a dry riverbed provides an alternative source of water in the already hydrologically modified environment. In many instances, this alternative water source provides the only opportunity for restoring aquatic and terrestrial habitats along rivers in urban environments. Given the goals of the net ecological benefit concept, it would seem that the goals driving the need to focus on ecological restoration can be linked to the benefits obtained from the discharge of effluent.

Although not explicitly stated as the purpose for such activities, examples of efforts to link the concepts of ecological restoration and net ecological benefit are already ongoing. For example, USACE has implemented projects in Phoenix and Tucson, Arizona that hope to result in the

partial restoration of aquatic habitats formerly associated with the Salt and Santa Cruz Rivers, respectively. In Tucson, efforts to restore habitats in the Santa Cruz River watershed are being led by the development of the Sonoran Desert Conservation Plan, a Habitat Conservation Plan that will address a host of issues, biological and cultural, in an integrated manner (Pima County 2000). A significant component of this effort is the use of effluent as a primary source of water to restore instream habitats.

1.3.7 Considering Cost Effectiveness

1.3.7.1 Introduction

Inherent in ecological restoration and net ecological benefit is the discussion of cost-effectiveness. As EPA indicates in its ecological restoration guidance,

> selecting the most cost-effective techniques is critical to the success of any restoration project…The two primary economic reasons why restoration may be more cost effective than point source controls alone are that (1) restoration often has lower marginal costs (i.e., the incremental costs of removing an additional unit of pollutant) and (2) restoration provides a wider range of ecological benefits (EPA 1995).

Moreover, the issue of cost versus benefit must be addressed locally since the expected effectiveness of restoration efforts may vary from one location to another:

> Some water quality problems (e.g., loss of habitat) are not amenable to a point source treatment approach at any cost; and some water quality problems cannot be reduced through any reasonable degree of restoration (EPA 1995).

In the arid West, municipalities frequently discharge treated effluent to ephemeral streams. Many of these streams would be completely dry during some part of the year but for the perennial flow from the wastewater treatment plants. Organisms living in or near such streams rely on continued discharges to maintain the aquatic and riparian habitat. The streams they live in are not merely effluent-dominated; they are effluent dependent.

Because freshwater is a scarce commodity in the arid West, competition for this resource increases as populations grow throughout the region. Many of the largest cities in Arizona, New Mexico, Nevada, Colorado, Texas, and California find it increasingly difficult to secure the permanent right for adequate freshwater supplies to meet anticipated demand. Naturally, as the problem grows more acute, interest in water recycling also intensifies.

The primary limitations on reclaimed water projects were the general lack of public acceptance and the total cost of distribution infrastructure. Historically, it has always been cheaper and easier to "treat and discharge" than to recycle municipal effluent. However, several factors are now combining to challenge the economic foundations of traditional wastewater disposal strategies:

1. The available supply of freshwater resources is being outstripped by population growth. In addition, environmental concerns frequently preclude municipalities from transferring

water from other jurisdictions. In particular, several significant water rights cases have radically altered the allocation to each state from major rivers flowing through the western United States.

2. The relative scarcity of new freshwater resources has caused the price of that resource to rise dramatically. The historical cost advantage that freshwater has enjoyed over reclaimed water is rapidly eroding.

3. The level of treatment required to continue discharging municipal effluent into natural streambeds is also continuing to climb. Water quality-based permit limits have caused most cities to construct advanced wastewater treatment systems designed to reduce the concentrations of ammonia, chlorine, and trace metals. New regulatory initiatives may require additional treatment to control pesticides, surfactants, pathogens, nutrients, total organic carbon, and total dissolved solids. Added costs encourage permittees to develop more creative strategies to assure long-term compliance with the Clean Water Act. Among the alternatives most frequently considered is increased water reclamation and reuse. By definition, this alternative implies reducing the amount of effluent discharged to local streams and washes.

1.3.7.2 Regulatory Dilemma

By law, water quality criteria must fully protect the designated beneficial uses on each water body. In addition, NPDES permit limits must be implemented to assure that the receiving water quality consistently complies with the applicable criteria. Unless exempted by a detailed site-specific analysis, all surface waters under the jurisdiction of the Clean Water Act are expected to support an aquatic life beneficial use.

Some states refine their use classification system to identify specific categories and subcategories of aquatic life. For example, many states distinguish between a coldwater and a warmwater fishery. Others draw a distinction between streams that support sport fish (bass, trout) and creeks that support primarily non-game fish (minnows, suckers, chubs). Rarely, however, do most states develop separate water quality criteria for each aquatic life category or subcategory. Instead, stream standards and permit limits are almost always based on federal guidance documents.

Unless site-specific water quality criteria are developed, EPA believes that their default standards are necessary to fully protect a stream designated as aquatic life habitat. Moreover, these standards are assumed to be appropriate regardless of whether the stream is comprised primarily of natural flows or is effluent-dependent. As a result, all permitted discharges are expected to meet the same high water quality standards regardless of the stream conditions prior to discharge. In the generic federal guidance, there is no special consideration given for hydrating an otherwise dry wash or creating riparian habitat where none existed before.

However, as noted earlier, increased competition for water resources and increased treatment costs have increased the incentive to divert wastewater flows away from natural stream channels. As a result, initiatives designed to improve water quality may, ultimately, cause the loss of aquatic habitat that the regulations were designed to protect.

A regulatory dilemma arises when one considers what level of water quality is needed to protect the designated beneficial use. EPA guidance distinguishes between an "existing use" and a "designated use" (i.e., a potential use). However, water quality criteria do not reflect that distinction. The same high level of water quality is deemed necessary to protect both. That is probably not true for effluent-dependent streams. In a naturally ephemeral stream, that would occasionally be dry but for flow augmentation derived from perennial effluent discharges, all existing uses arose under ambient water quality conditions. Therefore, one can conclude that existing water quality fully protects existing uses. It may be that better water quality would increase the richness and/or abundance of aquatic species in the effluent-dependent stream. Therefore, it is also possible to conclude that all of the potential beneficial uses are not fully supported by existing ambient discharge quality.

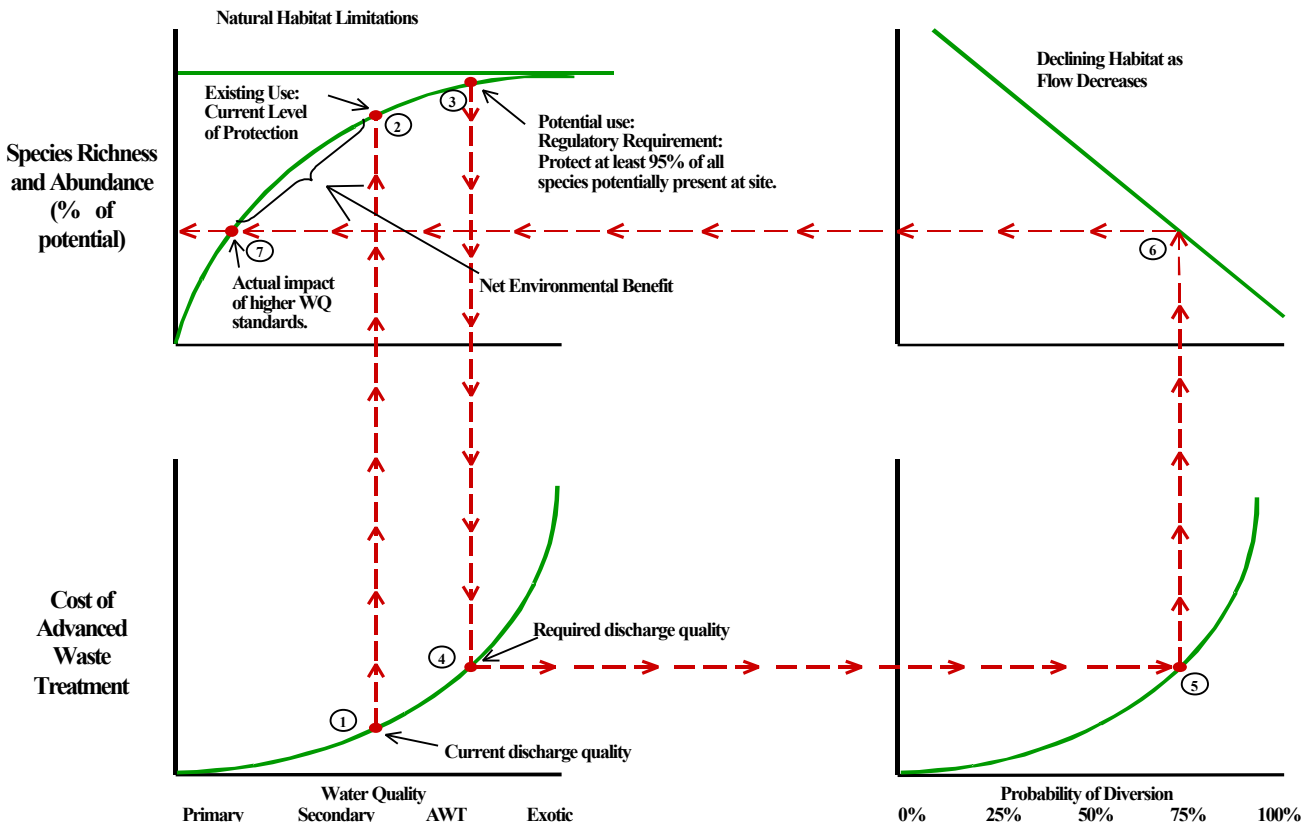
1.3.7.3 Unintended Consequences

In an effort to cause existing uses to achieve their full potential, most permit writers will impose limits designed to assure compliance with state or federal water quality criteria. Doing so, however, may not produce the biological improvement sought by the permit writer. This can best be understood by referring to **Figure E-8** (note the numbers in the following text refer to the numbers in **Figure E-8**).

By statute, all publicly owned treatment works are expected to provide primary and secondary wastewater treatment (∈). That level of treatment assures adequate water quality to cause a population of aquatic organisms to arise in the effluent-dependent stream (©). However, the richness and abundance of species is not as great as it might be if water quality were even better (™). Producing better effluent quality will require advanced waste treatment. And, depending on the facilities required, treatment costs will increase accordingly (∑). Higher treatment costs increase the probability that effluent will be reclaimed and discharges diverted to beneficial reuse (∫).

While it is often argued that water recycling increases to avoid wastewater treatment costs, actual experience indicates something quite different. There is little demand for reclaimed water that only meets secondary treatment standards. This is colloquially referred to as the "Yuk Factor."

The perceived value of effluent rises considerably when advanced waste treatment has been applied. Cleaner water is not only better for aquatic organisms, it is more acceptable to the general public. Nitrification to remove ammonia and filtration to reduce pathogens reduces the so-called Yuk Factor.



Economic Competition for Resources: Risk Adjusted Impact of Advanced Waste Treatment in Arid Regions.
Figure E-8

In the end, effluent is not diverted from discharge to avoid treatment costs but, rather, to recoup the cost of advanced waste treatment. Most public officials would state that they have a fiduciary responsibility to recover their capital investment in new treatment facilities and minimize the ongoing operation and maintenance costs by selling effluent rather than throwing it away. Consequently, wastewater managers are actively developing commercial markets, primarily in irrigation, for reclaimed water. High quality effluent is often used indirectly to recharge local groundwater supplies. It can also be traded to farmers and ranchers to increase the city's potable water resources. And, some cities have recycled effluent to develop non-contact urban renewal projects such as urban river walks. All such activities are considered responsible efforts to support the "sustainable development" promoted by EPA.

Regardless of the specific manner in which recycled water is reused, reclamation increases the probability that the effluent flows will be diverted. And, as a result, the aquatic habitat supported by those artificial flows will be materially reduced (see ⑥). Thus, where water quality was the factor precluding full attainment of the potential beneficial use, inadequate flows and insufficient habitat will severely reduce the maximum potential itself (see ⑦).

All things being equal, better water quality should improve the biological productivity and diversity of a stream. But, all things are not equal. The conclusion that the aquatic population will benefit from improved water quality is premised on the static assumption that everything else will remain the same. Analysis of the resource economics indicates that is extremely unlikely. In addition, results from this study suggest that physical and chemical factors associated with the creation of effluent-dependent waters also limit biological potential.

As the quality of reclaimed water becomes increasingly comparable to freshwater alternatives, and the free-market cost of that water offers significant value to nonpotable users, a significant substitution effect begins to occur. This effect has already been observed throughout the arid West as reclaimed water is used to irrigate golf courses, parks, and street medians. In California, reclaimed water has been used to recharge local groundwater basins for many years. And, there is a growing private market to buy and broker high quality effluent.

Evidence from the experience of early-adopters shows that the "tipping point" for substituting reclaimed water for potable water occurs when effluent undergoes tertiary treatment (nitrification and/or filtration). So, while it may not be possible to quantify the exact relationships depicted in **Figure E-8**, it is safe to say that the shape of the curves is well known.

1.3.8 Summary

The Clean Water Act explicitly provides for the protection of aquatic life and wildlife, but the emphasis in permitting has been almost solely on the protection of aquatic species. Little to no consideration is placed on riparian ecosystem protection, especially from the perspective of supporting and potentially enhancing riparian habitat and associated wildlife species. This may be shortsighted given the significant but unrecognized environmental benefits achieved from effluent supported terrestrial habitats.

NPDES permit effluent limitations, based on water quality standards, traditionally form the basis for protection of aquatic life in all waters, regardless of type. It is assumed that meeting water quality standards should result in increased numbers and kinds of aquatic species, even in created habitats. But if the implementation of wastewater treatment improvements yields little to no enhancement in the aquatic community, we must assume that limitations other than water quality exist. Continued emphasis on treatment without consideration of habitat limitations may lead to unintended consequences: increased demand for water reclamation and loss of effluent supported riparian habitat.

The traditional approach of establishing effluent limits in effluent-dependent waters, solely for the purpose of protecting an aquatic community that may not be attainable, needs reconsideration, especially in regions where water resources are limited. Using a net ecological benefit approach, alternative biological performance measures or outcomes could be considered as a means of measuring compliance with the goals of the Clean Water Act. Moreover, decisions regarding where resources should be expended could be based on a decisionmaking approach that uses concepts associated with ecological restoration, ecological risk, and watershed assessment programs or follows procedures associated with the NEPA process.

Establishing non-traditional alternatives for measuring permit compliance may be a cost effective and ecologically sound approach that recognizes the environmental benefits of having water flowing in an otherwise dry channel. Moreover, alternative outcomes that include terrestrial elements would support efforts to implement permitting on a watershed basis, support ecological restoration efforts, address concerns about the increasing loss of riparian habitat in the arid West, support increased interest in habitat restoration in urban rivers, and potentially benefit threatened and endangered species.