PROCESS-ORIENTED ECOLOGICAL RESTORATION OF FRESHWATER SYSTEMS

by

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(Under the Direction of R. Alfred Vick)

ABSTRACT

The ecological restoration of aquatic systems is becoming a common goal in ecological management. Although ecological restoration shows great promise, a high failure rate still persists. Many projects do not assess or address the altered environmental processes that cause ecosystem degradation, due in part to a lack of collaboration between practitioners and scientists. This thesis examines the need to focus on processes in ecological restoration design, as well as communication and collaboration between multiple fields. The environmental processes that shape ecosystems are discussed, and their role in a case study is explored. Finally, the role of the landscape architect in ecological restoration planning is discussed and a process-based restoration protocol is proposed.

INDEX WORDS: ecological restoration, environmental process, landscape architecture, ecology, environmental management, aquatic ecosystems, Truckee River

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CHAPTER 1

INTRODUCTION: THE NEED FOR PROCESS-ORIENTED DESIGN IN FRESHWATER ECOLOGICAL RESTORATION

Ecological restoration of freshwater ecosystems has gained popularity in recent years. There is increasing demand for stream channel naturalization, especially in urban areas (Wade et al. 2002), and wetland restoration has become a commonplace activity as a component of land development, infrastructure improvements, and other civil projects (Marble 1992, National Research Council 1992). This demand is driven by increased awareness of recreational, aesthetic, and public involvement benefits (National Research Council 1992, Purcell et al. 2002) as well as a greater understanding of ecology and the recognition of the high economic value of ecosystem services (Mitsch and Jørgensen 2004). These services include the filtering and cleansing of ground and surface water, providing flood storage areas, regulating discharge rates, and maintaining fisheries and other biologic communities (Mitsch and Gosselink 1993), as well as soil stabilization (Palmer et al. 2004), and even maintaining global climate (Mitsch and Gosselink 1993). Additionally, the intrinsic value of natural systems, as well as mankind's place in them, has been more widely recognized (Bookchin 1985, Sessions 1985). Ecological restoration of aquatic systems occurring as mitigation to lessen current damage has increased as a result of government regulation, such as the National Environmental Policy Act, the Clean Water

Act, and state regulations (Doppelt et al. 1993). As a result, ecological restoration has become a sizeable business; an average of over $1 billion has been spent annually since 1990 on stream and river restoration alone (Bernhardt et al. 2005). Restoration projects are often headed by governmental agencies or private entities such as developers (Bash and Ryan 2002, Moerke and Lamberti 2004).

Ecological damage to aquatic habitats can occur either directly, through direct alteration of wetlands, streams, or rivers, or indirectly, as a consequence of activities in the watershed. Direct alteration of aquatic systems was a common occurrence from the Industrial Revolution until recent times. Waterways were channelized and/or dammed for navigation, irrigation, power generation, and flood prevention (Williams 2001). Stream stabilization has traditionally been accomplished via hard armoring for the protection of nearby infrastructure (Li and Eddleman 2002). Wetlands were commonly seen as wastelands and were ditched, drained, or filled, often with the encouragement of governmental agencies. As a result, more than half of the wetlands in the lower 48 states have been destroyed (Mitsch and Gosselink 1993). Indirect damage to aquatic systems can occur as a result of changes in the watershed due to development and/or agriculture. Flows are altered by land use changes which affect infiltration and runoff rates, water quality is affected by nutrient supply and pollutants, and landscape fragmentation impedes the flow of biotic materials. These types of indirect effects are a major source of environmental degradation today (Doppelt et al. 1993).

The increased interest in restoration has resulted in an explosion of research and discussion in the scientific literature, and ecological restoration has been called "one of the fastest-growing fields in applied ecology" (Choi 2004). Whole journals have been established on the topic, such as Ecological Restoration and Restoration Ecology, while restoration topics are also commonly seen in scientific journals such as Ecological Applications, Environmental Management, Ecological Engineering, Wetlands, Freshwater Biology, the Journal of Applied Ecology, and Wetlands Ecology and Management, as well as non-scientific journals such as Landscape Architecture, Contemporary Sociology, Landscape and Urban Planning, Cultural Geographies, and the Journal of Agricultural and Environmental Ethics.

Although the field of ecological restoration has much to gain from collaboration between practitioners and scientists, there is little discourse between the two groups. Collaboration between the related disciplines of engineering, hydrology, geomorphology, ecology, and others can be difficult due to disparities in terminology, levels of precision and accuracy, spatial and temporal scale, and discipline focus (goals and values) (Benda et al. 2002). Project budgetary constraints can preclude detailed site analyses, while stakeholders can demand specific, visible outcomes (Tompkins and Kondolf 2003). A lack of appreciation for ecological complexities, combined with the fact that waterways projects were historically the exclusive domain of civil engineers (Williams 2001), has contributed to a perception that restoration projects can be installed as discrete objects in the landscape. However, for ecological restoration projects to be truly successful, collaboration between scientists and restoration practitioners is essential in restoration design so that all facets of the system and its design can be well understood (McDonald 2002).

The most commonly accepted definition of ecological restoration is that presented in The Society for Ecological Restoration (SER) International Primer on Ecological Restoration, as follows: "Ecological restoration is the process of assisting the recovery of any ecosystem that has been degraded, damaged, or destroyed" (SER International Science and Policy Working Group 2004). Ecosystem is further defined as consisting of the biota, the sustaining environment, and their interactions. The goals of successful restoration include: containing a characteristic assemblage of species and all appropriate functional groups of species, capability of sustaining reproducing populations, absence of signs of dysfunction, elimination and/or reduction of potential threats to ecosystem health and integrity, resiliency to normal periodic stresses, and the potential to persist indefinitely (SER International Science and Policy Working Group 2004).

It is widely understood by ecologists that ecosystems are the result of the actions and interactions of many environmental processes (National Research Council 1992, Whisenant 1999). Environmental processes are those forces that affect landform, resource availability, and disturbance and stressors. Such processes are both abiotic (such as water discharge, runoff, infiltration, sediment erosion, transport, and aggradation, and aeolian and solar influences) and biotic in nature (such as succession, nutrient cycling, decomposition, primary productivity, herbivory, and predation). The ecosystem is a result of the forces of the actions and interactions of these forming processes, which vary in time and space by natural disturbances. The ecosystems formed are thus in dynamic equilibrium with these forces, constantly shifting and adjusting in response to changing processes (Hobbs and Norton 2004). Environmental disturbances can result in the alteration of ecological systems by disrupting one or more of these processes, shifting the equilibrium state (Whisenant 1999, Li and Eddleman 2002). Such disturbances can be short term (pulse disturbances) or long term (press disturbances) (Glasby and Underwood 1996). The intrinsic resiliency of ecosystems can accommodate most natural short-term disturbances (Chapman 1998, Palmer et al. 2005), and many aquatic ecosystems rely on such disturbances to maintain stability (Middleton 1999). Long-term environmental change can occur when press or catastrophic pulse disturbances alter the processes beyond what the ecosystem can absorb (Whisenant 1999). The restoration of the natural regime of processes that form the desired habitat (National Research Council 1992, Whisenant 1999, Palmer et al. 2005), and the identification of the type of disturbance which caused the environmental degradation, if possible (Chapman 1998), is therefore necessary for successful ecological restoration.

A great deal of the scientific literature discusses means of determining success or failure in ecological restoration (Zedler and Callaway 1999, Nienhuis and Gulati 2002, Hobbs 2003, Palmer et al. 2005). An improved rate of success is desired not only for the achievement of particular ecological goals, but because of the great expense involved in restoration projects. Such expenses can range into the billions of dollars for large-scale, high profile restoration projects, and can include thousands of hours of manpower (Holl et al. 2003). River restoration projects can cost $100,000 per kilometer (Malakoff 2004).

In spite of the growing interest in ecological restoration, a large proportion of projects do not truly restore the processes and functions of the ecosystem (National Research Council 1992, Whisenant 1999, Ward et al. 2001). While failure is often not reported (Nienhuis et al. 2002), large portions of past restoration projects can be considered unsuccessful (Choi 2004). For example, a survey of stream restoration projects in Illinois rated all restorations as “moderately successful” or “extremely successful” by their respective restoration designers. However, habitat quality of post-restoration conditions was not found to be significantly higher than in pre-restoration conditions, as measured using the Qualitative Habitat Evaluation Index (QHEI) (Moerke and Lamberti 2004). In addition, reported successes can sometimes be based on satisfaction of regulatory requirements, achievement of development goals, or aesthetic improvement (Moerke and Lamberti 2004) rather than on the ecological integrity of the project area.

Within the restoration literature, the need for both baseline analyses and routine post-project monitoring have been repeatedly stressed (National Research Council 1992, Box 1996, Hobbs and Norton 1996, FISRWG 1998, Zedler and Callaway 1999, Kentula 2000, Bash and Ryan 2002, Downs and Kondolf 2002). Pre-design assessment is necessary to understand the current state of the degraded ecosystem. The types and degrees of ecological damage must be identified and clear, measurable goals should be established. Such goals must be able to reflect the objective of the project. Since ecosystems are dynamic and constantly adjusting to alterations in the environment (especially in the early stages of recovery), the desired restoration outcome might be best thought of as a trajectory; the ecological direction in which the habitat is progressing. Therefore, goals such as recruitment rates of different of plant species are likely to be more appropriate than goals stating a required percent cover.

Post-project monitoring is necessary to assess the progress of the recovering system. As the project becomes established, adaptive management plans must be implemented to correct problems with installation or to help guide recovery towards the desired trajectory (FISRWG 1998, Kentula 2000, Downs and Kondolf 2002). However, many restoration projects are planned without baseline assessments. Projects with assessment and/or monitoring components may be as low as 10% nationally (Bernhardt et al. 2005). Projects with an engineering focus were found to be much less likely to collect baseline data or perform post-restoration monitoring than other projects (Bash and Ryan 2002). The goals of these projects were found to be more concerned with meeting regulatory requirements than with restoring ecological functions or processes. Surveys in Illinois and Washington state found that baseline monitoring occurs in less than or up to half of all restoration projects (Bash and Ryan 2002, Moerke and Lamberti 2004); while post-project monitoring was performed in about half the projects, it was required in only

9% (Bash and Ryan 2002). Such ad-hoc approaches to restoration design, combined with limited understanding of ecological principles, contribute to the lack of restoration successes (Choi 2004).

While the restoration of aquatic ecosystems is clearly complex, simplistic approaches are often used (Williams 2001, Tompkins and Kondolf 2003, Palmer et al. 2005). Restoration projects are often entirely designed by civil engineers, engineering firms, or others who may have difficulty incorporating all of the processes involved in creating a healthy ecosystem (van Diggelen et al. 2001, Williams 2001, Palmer et al. 2003) or hesitate to use biological and ecological solutions to environmental problems (Mitsch 1998). While some restoration practitioners have limited experience in hydrology or geomorphology (Malakoff 2004), others lack an understanding of the needs of biotic communities (Palmer et al. 2003). Additionally, some projects focus mainly on non-ecological issues and include large social, economic, and aesthetic components. These components often compete with ecological goals and can degrade aspects of aquatic ecology by dictating bank form, flood flows, soils, and adjacent ecological communities (Palmer et al. 2005). Practitioners should be cautious with non-ecological objectives in ecological design, and ensure that ecological recovery persists as the primary goal of an ecological restoration project. When non-ecological objectives become the primary goals, the ecological aspects of a project might be better termed "ecological enhancements" than "ecological restoration".

A common approach to ecological restoration involves the physical reconstruction of the desired habitat through alteration of land form and planting of vegetation (Whisenant 1999). However, this tactic is unlikely to result in successful restorations if the ecological processes of the site and those needed for the desired habitat have not been adequately understood and addressed (Tompkins and Kondolf 2003). In most cases, ecological damage is the result of altered processes (Whisenant 1999). Where the original habitat type is re-installed without addressing altered processes, these same processes can continue to degrade the project area. For the new ecosystem to persist and become self-sustaining, it must be supported by the environmental processes which shape it (Whisenant 1999). Therefore, these environmental processes must be accounted for in the restoration design.

In many cases, restoration projects are designed following a pre-scripted approach with minimal site-specific analysis (Miller and Skidmore 2001). The use of the Rosgen classification system (Rosgen 1996) is commonly used to plan stream and river restoration projects (Miller and Ritter 1996, Doyle et al. 1999, Kondolf et al. 2001). The intent of the classification system was to classify rivers, forecast how their form might change in response to environmental changes, and identify channel shapes that can remain stable for use in restoration design (Rosgen 1996). While this classification system can provide an excellent means of communicating stream morphology (Miller and Ritter 1996), there is risk of misapplication by inexperienced practitioners (Li and Eddleman 2002). In addition, its use in restoration design has been strongly criticized in the scientific literature as lacking a sufficient analytical foundation in geomorphological processes and oversimplifying the relationships between environmental processes and bank form (Gillilan 1996, Miller and Ritter 1996, Doyle et al. 1999, Harmel et al. 1999, Kondolf et al. 2001, Miller and Skidmore 2001). Rosgen (1994) acknowledges that environmental processes are responsible for shaping river morphology and that the “natural stable tendencies” of rivers must be understood to design successful restorations. However, the suggestion that ecological improvement can be achieved simply by the construction of new, more appropriate, channels (Rosgen 1994) is unsound and contraindicated. Since stream morphology is directly influenced by hydrogeomorphic processes (Rosgen 1994), a degraded system is either at or progressing towards dynamic equilibrium with altered processes. Permanent alteration of stream morphology requires addressing and manipulating processes to support the desired form. The construction of new a stream form without addressing the processes needed to support it will result in structural failure (National Research Council 1992, Kondolf 1998). The construction of a new stream form that is at equilibrium with the altered processes may produce a stable stream bed, but not necessarily the restoration of a locally appropriate ecosystem (Palmer et al. 1997, Kondolf et al. 2001). Such stream reconstruction can be an important component in ecological restoration, but when considered alone it fails to address other aspects of the ecosystem (e.g. community development, food web dynamics, and nutrient cycling). Still, the system is widely used as it is easier to perform than many others (e.g. modeling), personnel can be trained in week-long courses, and is often required by state and federal agencies (Doyle et al. 1999, Miller and Skidmore 2001, Malakoff 2004).

The failure of many stream and river restoration projects has been attributed to the exclusive use and/or misuse of channel classification systems and/or form-based design, without proper analysis and integration of the geomorphological and other processes that are active at the project sites (National Research Council 1992, Sear 1996, Kondolf et al. 2001, Xiong et al. 2003, Malakoff 2004). However, since few restoration projects receive post-construction monitoring and restoration failures are seldom reported, such restoration designs are perceived to be successful (Tompkins and Kondolf 2003). Form-based project design failure was exemplified in Uvas Creek in Gilroy, California, where a prescribed stream channel with regular meandering bends was installed to mitigate the effects of gravel mining. Within four months, the stream abandoned the constructed channel and assumed a braided form in the newly constructed floodplain. Although plan documents stated that the designed channel type was what had existed before disturbance, historical and geomorphological analyses were not performed. A third-party post-project analysis indicated that there was no historical evidence for the stream type used, and geomorphological analyses indicated that the planned channel design would be metastable at best and prone to braiding. During a calculated 5-6 year storm flow, the new stream abandoned its prescribed banks and buried some of the installed revetments and grade controls. Kondolf et al. (2001) concluded that the processes that had formed the historic braided channel at the site would tend to re-create a similar channel during adequate flows. As shown by this example, a thorough investigation into the history of the ecosystems form and processes is necessary, as well as a comparison to its current form and processes. The restoration design can then aim for a form that is ecologically appropriate and the processes needed to sustain it.

Bioengineering (also called biotechnical engineering or ecological engineering) has become a popular tool for river and stream restoration and stabilization (National Research Council 1992, Li and Eddleman 2002). This practice involves the use of live riparian vegetation, along with biodegradable structures (and occasionally rock structures) to stabilize bank slopes (Li and Eddleman 2002). Many projects also include engineering methods such as rip-rap stabilization and introducing boulders into the stream channel (Moerke and Lamberti 2004). Bioengineering is an attractive alternative to hard armoring methods, as it can provide some improvements by: improving surface water and groundwater interactions (Ward and Trimble 2004), reinforcing soils and root anchoring of slopes (Li and Eddleman 2002), reducing soil compaction and particle detachment (Li and Eddleman 2002), reducing stream velocity and slowing surface runoff (Li and Eddleman 2002), improving wildlife habitat (Morrison 2002), improving water quality and nutrient absorption (Hammer 2000), and increasing shade and lower stream temperatures (Allan 1995).

Although bioengineering is an important tool, it must not be confused with ecological restoration. Unfortunately, many projects termed "ecological restoration" use little more than bioengineering techniques to manipulate stream beds and banks. The goals of bioengineering are to stabilize the bank and prevent erosion and stream migration (Callahan 2001, Tompkins and Kondolf 2003). These goals are inappropriate for ecological restoration, however, and ignore both the natural migration patterns of rivers and streams (Callahan 2001) and the importance of natural bed movement to support healthy, regionally appropriate aquatic ecosystems (Kondolf et al. 2001, Tompkins and Kondolf 2003). The stability of ecosystems must not be confused with geomorphic stasis. Ecological and geomorphic stability rely on a state of dynamic equilibrium, where natural disturbances regulate the populations of some organisms, while permitting the growth of others (Palmer et al. 1997). This temporal variability allows for greater biodiversity (Wisheu and Keddy 1992, Palmer et al. 1997) and greater functional redundancy within the ecosystem (Palmer et al. 2004).

A principal shortcoming of many projects is that they do not assess and address the sources of environmental degradation (Larson et al. 2001). As a result, these projects are not likely to result in the long-term restoration of healthy ecosystems. There are two manners in which these types of projects are likely to fail. The first is that the structure of such projects as installed are not likely to persist long term. If the altered processes that resulted in damage to the original ecosystem have not been addressed, these same processes can fail to support or even undo the installed project (Kondolf et al. 2001, Bond and Lake 2003). This type of failure would include streambank erosion as biotic stabilization materials degrade, inappropriate sedimentation rates, low survival rates of transplanted vegetation, or the failure of vegetation to propagate. By understanding the altered processes that caused the environmental damage, the practitioner can both manage these processes to support the desired ecosystem structure, and design a structure that can be supported.

The second manner in which projects can fail is that they neglect requisite components of a healthy ecosystem and focus mainly on site morphology and general plant establishment, thereby failing to recruit appropriate biota. Stream restoration projects usually focus on channel patterns and cross sections (Smith et al. 1999) and often neglect non-geomorphologic ecosystem processes (Whisenant 1999). This practice is reinforced by the restoration literature, which places heavy emphasis on channel morphology (e.g. Henshaw and Booth 2000, Tompkins and Kondolf 2003). This may be due in part to a historical lack of understanding of the role that areas contiguous to rivers play in sustaining ecological processes (Ward et al. 2001). It is assumed that if stable soils and vegetative cover are provided, other forms of biota will be able to invade, persist, and reproduce (the Field of Dreams hypothesis) (Palmer et al. 1997). Physical stability alone does not provide for a healthy ecosystem (Henshaw and Booth 2000) and it has been shown by several studies that the presence of appropriate plants does not necessarily result in a functioning ecosystem (Palmer et al. 1997). However, little effort is put into determining the specific habitat requirements of all component species nor towards the establishment of functional connections between individual organisms, the biological community, and the physical environment (Ehrenfeld and Toth 1997). There are likely several reasons for a lack of attention to these issues. The first is that a complete habitat assessment and plan taking into account all the expected species, their needs and interactions with other biota and environmental factors as well as the expected characteristics and actions of all environmental processes would require substantial effort and expertise from many disciplines (such as geologists, hydrologists, soil scientists, and plant and wildlife ecologists). In addition, a lack of appreciation of the importance of the above issues by the public can make justification of the effort and expense difficult. More problematic is that many environmental practitioners are also likely unaware of the importance of addressing all the interactions and requirements of the desired ecosystem (as evidenced by the myriad of restoration plans that do not take them into account). Secondly, methodology and/or guidance on planning process-based ecological restorations have not been established. As a result, designing such restoration plans can require that an ad-hoc method be generated for each project. There is, of course, difficulty in providing specific guidance on project design due to the lack of complete (and constantly evolving) scientific understanding of many of the components of the ecosystem. As a result, no absolute guarantee of success can be provided with any methodology or guidance, due to the level of uncertainty involved.

The current lack of appreciation of all environmental components can be manifested in numerous ways. Often overlooked is the variability within ecosystems essential for providing multiple niche requirements for the component biota (Palmer et al. 1997, Middleton 1999). Of equal importance (yet unaddressed in many plans) may be the arrangement, shape, and size of microhabitats for the establishment and persistence of biota (Palmer et al. 1997) and for influencing stream morphology and flow regimes (Sear 1996). Most restoration practitioners now recognize the importance of stream riffles, pools, and runs (National Research Council 1992, Rosgen 1996, Knighton 1998, Morrison 2002), but many other types of microsites are not addressed. For example, Moerke and Lamberti (2004) found that only 10-30% of stream restorations studied in Illinois intentionally created microsites such as undercut banks, backwater areas, and vegetated bars. Projects relying mainly on bioengineering as a technique do not necessarily address all the issues generally required to be considered ecological restoration, such as the restoration of an appropriate assemblage of species (flora and fauna) and functional groups, the environment that sustains this biota, and the interactions between biota and the environment. While bioengineering for bank stabilization can improve aquatic invertebrate habitat, it can not mitigate other types of ecological damage (Sudduth 2004). As a result, most bio-engineered restoration projects are not comparable to natural systems, do not restore natural river dynamics, and can restrict other processes necessary for natural ecosystem functioning (Callahan 2001). It must be remembered that bioengineering is an important tool for achieving restoration goals, but use of this tool alone is unlikely to result in long-term success. Of paramount importance in ecological restoration is sufficient attention to the restoration of biotic interactions of ecosystems, as well as their ecological functions.

It is widely recognized that upland, floodplain, and wetland areas play an important role in aquatic ecosystem health (National Research Council 1992, Horne and Goldman 1994, Allan 1995, Jungwirth et al. 2002). However, areas contiguous to flowing waters, such as floodplains, wetlands, and watersheds are routinely not considered as a part of restoration theory or during restoration planning (Smith et al. 1999, Ward et al. 2001), and the scale at which landscape processes operate are rarely considered (Briggs 2001). A large number of restoration projects are limited to small segments of rivers and streams (Moerke and Lamberti 2004), and common restoration methods rely on physical manipulations that are restricted to the bed and banks (Kondolf 1996, Sear 1996, Tompkins and Kondolf 2003). However, relying solely on in-stream and bank manipulations does not address the altered processes that are the cause of environmental degradation (Callahan 2001), and the exclusion of contiguous areas and their influences on ecological processes can limit the success of restoration projects (Ward et al.

2001). To maximize the potential for successful restoration, the entire riparian landscape must be considered (Allan 1995)

Where habitat is a specific concern in restoration, it often focuses on a single or small group of species of concern (National Research Council 1992). Endangered species are a particularly important target population in ecological restoration (Morse 1996). These types of projects have the potential to neglect the interactions of the species of interest with the other (biotic and abiotic) components of the environment. The result of this type of narrow focus is often that other species are not protected and supported in the project area (National Research Council 1992). The entire community of organisms should be considered, as they are an integral part of maintaining ecosystem functions.

A common tool in stream restoration is to use large woody debris and other structures as methods of improving habitat, especially with respect to fish populations (FISRWG 1998, Larson et al. 2001). The use of in-stream structures has proven to increase geomorphic stability and habitat variability and improve community structure in some circumstances (Gore et al. 1998, Purcell et al. 2002, Harrison et al. 2004). Habitat management of small areas can be beneficial, but does not address the needs of species with multiple habitat requirements during different seasons or lifecycle stages (Briggs 2001). In addition, it has been shown that the use of in-stream structures alone may not positively affect habitat or sedimentation rates, especially in urban streams (Larson et al. 2001). Moerke and Lamberti (2004) stress that in-stream manipulations alone are unlikely to result in successful restorations, and that the contiguous area must be included. For example, the placement of large woody debris in streams is a temporary measure, as it does not ensure future wood recruitment (Palmer et al. 2005). Where woody debris is lacking in a stream system, it is necessary to evaluate the process of wood recruitment and retention. Restoration projects should ensure that the surrounding ecosystem can accommodate future woody debris inputs, as well as other appropriate biotic materials.

Successful ecological restoration projects include the restoration of an ecosystems biota, sustaining environment, and their interactions, and should also be as self-sustaining as possible (National Research Council 1992, SER International Science and Policy Working Group 2004). A process-oriented restoration approach is most likely to ensure long-term success, as it yields the support structure and feedback needed to create the system of functions and interactions in the environment (Bradshaw 1996, Tompkins and Kondolf 2003). To lessen further degradation of the restored system, the source of environmental damage must be determined and its effects reduced. The remainder of the processes that shape the environment must then be restored and balanced to guide the ecosystem towards its desired trajectory (Kondolf et al. 2001).

The return to a pre-disturbance state, or the re-creation of a reference system is rarely possible due to the variability of ecosystems in space and time (Gillson and Willis 2004), as well as effects from disturbance size and permanence. Rather, project goals should focus on functions for the present and future environment (Choi 2004). In many urban settings, the processes that formed the original ecosystem are damaged beyond repair, or sufficient restoration of these processes is prevented by infrastructure needs. A process-oriented design can help develop realistic restoration goals (Tompkins and Kondolf 2003) by assessing the processes that are available to shape the ecosystem, those that can be restored, and those that can not be restored and must be managed within the human environment.

The following chapters of this thesis discuss the principles of ecological processes and their potential applications. Chapter 2 includes a survey of environmental processes that shape ecosystems, typical process alterations that can cause ecological degradation, and possible manipulation strategies. Chapter 3 evaluates restoration efforts in the lower Truckee River, Nevada, to assess where process-oriented approaches are used and where processes are potentially neglected. Chapter 4 discusses process-oriented restoration in the context of landscape architecture, land use planning and design, and societal structure.

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CHAPTER 2

ENVIRONMENTAL PROCESSES AND THEIR RELATIONSHIPS TO ECOSYSTEM FORMATION

Introduction

Community composition and ecological functions are affected by a variety of environmental processes. Process-oriented ecological restoration requires a thorough understanding of all the potential ecosystem-shaping processes, their actions, and interactions, on the land form, biota, and chemistry. These include hydrologic, geomorphologic, chemical, and biotic processes. These processes should be encompassed by the restoration design such that they form the web of interactions that create and support the desired system. Many restoration projects can be designed such that these processes are manipulated to shape the desired system (Whisenant 1999). However, as environmental processes often affect the environment slowly, there can be a great deal of short-term uncertainty as processes begin to form habitats (Tompkins and Kondolf 2003). In addition, the manipulation of environmental processes can not be performed in isolation, as the interrelations of processes affect the entire ecosystem (Xiong et al. 2003).

This chapter is organized by process type (water balance, fluvial, chemical, and biotic processes), rather than by environmental characteristic. The goal of this organization is to present the complete role of each process in the environment. Many freshwater restoration books are organized by environmental characteristic (e.g. geomorphology, plant communities, or fish habitat), with the related processes discussed for each type. Such organization may encourage the manipulation of environmental characteristics in isolation, while neglecting the effects of altered environmental processes on other ecological features. It can also encourage the formation of inappropriate goals: the aim is not to repair geomorphology or establish certain plant communities, but to repair the processes that establish and maintain these features. Either of these tactics can lead to the neglect of other components of the ecosystem or even the manipulation of processes to benefit one aspect of the environment while unknowingly degrading others.

When manipulating environmental processes for restoration, the following must be considered: 1) why a process should be changed, 2) the desired direction of change, 3) how the process might be changed, and 4) other processes or environmental components that might be affected. For a restoration project to be considered process-oriented, all the processes affecting the ecosystem should be examined to increase the likelihood of environmental stability.

Water Balance Processes

Water balance processes and their effects

The availability of water in the ecosystem is regulated by the water balance of the catchment, representing the volume of water that enters, leaves, and is stored by the system. It is calculated as:

$$P = I + AET + OF + \Delta SM + \Delta GWS + GWR$$

Where P = precipitation, I = interception, AET = actual evapotranspiration, OF = overland flow, ΔSM = change in soil moisture, ΔGWS = change in groundwater storage, and GWR = groundwater runoff (Dunne and Leopold 1978). The processes of interception, infiltration and runoff, and evapotranspiration are often addressed in ecological restoration projects.

Interception

A portion of rainwater is intercepted by vegetation or other surface cover and is evaporated before it reaches the ground. Interception is greatest in areas of dense vegetation, and small rain events can sometimes be completely intercepted and evaporated. While interception usually reduces the amount of rain water reaching the ground, it can increase the water budget in very foggy areas by condensation onto vegetation (fog precipitation) (Dunne and Leopold 1978).

Rain interception protects the structure of surface soils; the impact of unintercepted rainfall can disturb aggregates and contribute to surface crusting and decreased infiltration (Dunne and Leopold 1978).

Infiltration and runoff

The rate at which water infiltrates into the soil is regulated by soil's pore space, drainage rate, and initial moisture content. While large pore spaces allow rapid infiltration, small pore spaces can retain water by capillary action. The loss of pore space due to soil compaction or surface crusting impedes infiltration, as does the construction of impervious surfaces. Infiltrated water is stored in the pore spaces until the soil's field capacity is reached; excess drains into the saturated subsurface zone as groundwater recharge. Groundwater flows both vertically and laterally through the soil as a function of gravity, soil characteristics, and geological barriers; the seepage of this water into freshwater ecosystems provides the baseflow of rivers, streams, lakes and wetlands (Dunne and Leopold 1978). Soluble nutrients and pollutants carried by infiltrating water can degrade groundwater quality (Ward and Trimble 2004), and reduced recharge in coastal areas can allow saltwater intrusion (Dunne and Leopold 1978).

Soil moisture is affected (in part) by infiltration rates (Lyon et al. 1952). Changes in these rates can alter plant communities due to increased or decreased soil moisture (Cronk and

Fennessy 2001). Infiltration and drainage also affects soil fertility (Ponnamperuma 1972). Water soluble nutrients can be leached from the soil; the degree of leaching depends on the rate of water movement and the nutrient supply (Horne and Goldman 1994). Moist soils allow for increased decomposition rates (which usually lower soil pH); however, decomposition decreases in consistently saturated, anoxic soils, raising soil pH. Saturated, anoxic soils also exhibit chemical changes such as denitrification, reduction of metals, and solubilization of phosphorus (Ponnamperuma 1972).

The infiltration and water holding capacities of the soil greatly affect plant community composition. While some species can adapt to a range of conditions, many are limited by their drought or saturation/inundation tolerance levels (Cronk and Fennessy 2001). Few species are adapted to a wide range of moisture conditions; drought-facilitating characteristics generally limit saturation tolerance and wetland characteristics greatly reduce the ability of a plant to survive drought (Hook 1984).

When the infiltration capacity of the soil is exceeded during storms, surface accumulation and runoff occurs as a thin sheet of water over the soil surface (Horton overland flow). The amount and timing of stormwater reaching surface water bodies determines various discharge attributes, such as baseflow, mean discharge, flood frequencies, and stream velocity. The combination of baseflow and stormflow constructs the hydrograph of surface water bodies, which can vary during different seasons due to differences in precipitation and/or snowmelt (Dunne and Leopold 1978, Ward and Trimble 2004). Figure 2.1 shows a typical hydrograph.

Alteration of runoff processes in developed areas commonly results in changes to the hydrograph, as increased overland flow results in rapid drainage of stormwater to streams and wetlands. As such, lag time and base flows are reduced, the duration of storm flow shortens, and peak flow rates increase (Ward and Trimble 2004). Increased overland flow also affects surface water quality, especially in catchments where fertilizers are used, by increasing the transport of soluble nutrients (Horne and Goldman 1994). In areas where soils contain high amounts of clay and silt, soil erosion can result in considerable increases in water turbidity and phosphorus loads (Horne and Goldman 1994, Knighton 1998). In addition, runoff from impervious surfaces often carries pesticides, metals, and organic pollutants from automobiles (Ward and Trimble 2004) and can increase surface water temperatures (Dunne and Leopold 1978).

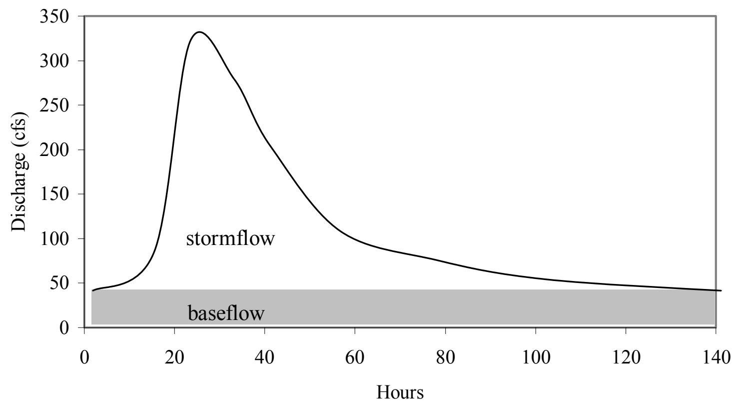

Figure 2.1: A typical storm hydrograph

The runoff of nutrients and pollutants can also affect soil quality in downgradient depressions and wetlands (Horne and Goldman 1994, Knighton 1998), and can dramatically affect plant communities by increasing the competitive ability of some plant species. Large-leaved, rhizomatous species such as *Typha angustifolia* are particularly able to utilize increased nutrient loads and the competitive exclusion of other species often results in monospecific stands (Wisheu and Keddy 1992). Seed-set and germination of some plant species can also be influenced by nutrient availability (Keddy and Ellis 1985, Boyer and Zedler 1999).

Many of the smaller species of wildlife, such as invertebrates, reptiles and amphibians, and small and/or young fish can be directly affected by water availability. Such species rely on damp or inundated areas to avoid desiccation during at least part of the life cycle, or even wetting and drying cycles for reproduction and predator evasion (Gallagher 1996).

Evapotranspiration

Water is lost from the ecosystem into the atmosphere by evapotranspiration, the sum of water evaporated from surfaces and the transpired through vegetation. Affected by temperature, humidity, and the type and density of vegetation, evapotranspiration affects soil moisture content, groundwater recharge, and stream flow, accounting for two-thirds of the water budget in the lower 48 states (Dunne and Leopold 1978). Changes in evapotranspiration rates caused by the alteration of plant densities or community types can even alter the water balance of the ecosystem (SER International 2004).

The manipulation of water balance processes

Water balance processes can be manipulated to affect infiltration and runoff rates, thereby affecting water levels, channel hydrographs, soil moisture, and soil chemical processes. The objective of working with water balance processes in restoration is often to increase infiltration in order to improve groundwater recharge, decrease the frequency and magnitude of flooding, and increase stream base flows (Ferguson 1998).

In recent years, a variety of stormwater management practices have been developed with the aim of slowing runoff and increasing infiltration. Design strategies such as cluster developments and improved street and parking design standards, as well as the use of porous pavements and green roofs can reduce stormwater runoff by reducing impervious surfaces (Ferguson 1998, BASMAA 1999, BASMAA 2003, VanWoert et al. 2005). Runoff that does occur can be mitigated and infiltrated using best management practices (BMPs) such as infiltration basins, vegetated swales and buffers, extended detention (dry) ponds, wet ponds, and biofilters (BASMAA 1999, BASMAA 2003). The water balance can also be managed by increasing vegetative cover to intercept rainwater, thereby preventing soil crusting and improving infiltration capacity. When densely established, vegetation can also be used to increase evapotranspiration rates, thus potentially removing a substantial portion of water from a catchment (Dunne and Leopold 1978). The creation of freshwater wetlands as water management devices incorporates the benefits of infiltration basins with increased evapotranspiration rates, while removing and/or absorbing nutrients and water pollutants (Hammer 2000).

To most closely restore water balance processes to their original state, the entire watershed must be considered, with rates of interception, infiltration, runoff, and evapotranspiration addressed in each sub-basin (Brookes and Sear 1996, Kondolf and Downs 1996). This approach is not easily undertaken, and can require extensive research on current and historical conditions, as well as potentially intensive management over wide-spread areas. This prevents the exact duplication of water balance processes in many areas, especially when cost and land-use issues such as regulation and ownership are considered. Improvements in the entire catchment can be made, however, using BMPs to mitigate land-use effects (Ferguson 1998, Crabtree 2001, Villarreal and Bengtsson 2004). The most effective, and least costly, method of stormwater management is the use of many small control techniques (such as rain gardens and pervious pavements) repeated throughout the project area (BASMAA 1999), which are more likely to simulate the natural water balance processes and the hydrograph.

Where current infrastructure prevents the use of numerous small BMPs, specialized mitigation sites can be used to alleviate increased runoff volumes. Targeted positioning of constructed wetlands, infiltration, vegetated swales, and buffers between impacted areas and the headwaters of streams can help mitigate some of the impacts from upland areas (Ferguson 1998, Ward and Trimble 2004). Such systems collect stormwater runoff from multiple sites (e.g. a neighborhood) and control its entry into ground and surface waters. While this strategy seems attractive in that it offers a centralized location, it is not as effective as multiple smaller BMPs and has higher costs and maintenance requirements (BASMAA 1999).

Fluvial Processes

Fluvial processes and their effects

Fluvial processes are those which are driven by or are strongly tied to the forces of flowing water, such as those affecting the hydrograph, flow characteristics (such as velocity and stream power), the erosion and deposition of sediments, and the geomorphology of surface waters. Strong interactions and feedback systems are present among fluvial processes (Knighton 1998), such as the action of water forces on stream morphology and vice-versa. As a result, fluvial processes should not be examined or manipulated in isolation (Kondolf and Downs 1996, Ward and Trimble 2004).

The degree to which we understand fluvial processes is constantly evolving. Their many aspects form an intricate web of cause and effect interactions (Knighton 1998), and there is much controversy over whether these processes or their results can be accurately predicted (Darby and van de Weil 2003, Ward and Trimble 2004). As a full discussion of all the aspects of fluvial processes would give rise to many volumes, the goal of this section is to provide a basic outline of the fluvial processes that should be considered when addressing channel form and related issues.

Flow characteristics

The energy of water flow produces the primary earth-shaping forces of freshwater systems. The forces of water within a stream reach are affected by the slope of the system, as well as the shape of the bank and bed (Knighton 1998). Stream discharge equals the cross-sectional area times the stream velocity (Ward and Trimble 2004); however, velocity varies with distance from the stream perimeter and with differences in turbulence (Allan 1995, Knighton 1998). Generally, velocity is fastest near the surface and approaches zero at the streambed (Allan 1995). It is affected by flow resistance from the grain and roughness of the bed material, the planform of the channel, and turbulence (Knighton 1998). Manning's equation is commonly used by planners to describe stream velocity:

$$u = \frac{1.49R^{2/3}S^{1/2}}{n}$$

Where R = the hydraulic radius (ratio of the cross-sectional area of the channel to the wetted perimeter), S = the energy gradient (approximated by the slope of the water surface), and n = the Manning resistance coefficient (which varies with the roughness of the channel boundary) (Dunne and Leopold 1978). Discharge is positively correlated with increases of depth, width, and/or velocity (Allan 1995).

While various expressions of stream flow and flood stage are used by hydrologists, perhaps the most important flow value for fluvial processes is stream power, representing the energy available to perform physical work within the system. It is defined as:

$$\Omega = \gamma Qs$$

Where γ = the specific weight of water, Q = discharge and s = slope (Knighton 1998). Closely related is sheer stress, the force exerted on particles by water as it flows downstream.

Velocity commonly increases as a result of stream channelization, due to decreased channel roughness and/or increased channel slope. Such streams can become flashy, with rapidly drained discharge, higher peak flows, and lower base flows (Dunne and Leopold 1978, Knighton 1998).

Transport of materials: erosion, scour, and deposition

Streambed and bank erosion is a constant feature of running waters as stream power and shear stress move particles downstream. The transportation of a particular particle depends on these forces as well as the particle's size and shape; larger particles require greater force if they are to be moved (Sear 1996). Streambed materials can be transported as suspended load, where smaller particles are suspended in the water column, and as bed load, where larger particles are transported by rolling, dragging, or skipping along the stream bottom. Under constant discharge conditions, the rate of particle transport is variable due to minute adjustments of channel geomorphology (Ward and Trimble 2004). Streambed and bank scour naturally occur during storm events, when increased stream power (due to increased discharge) increases both the suspended and bed loads. Materials are deposited when stream power decreases, such as after storm events or where the channel becomes widened. Some very fine particles such as clay often never settle out of the water column; the force from even low flows can be sufficient to keep them suspended (Sear 1996, Knighton 1998). The riffle-pool sequence is also an important factor in sediment transport, as fine sediments are stored in the low-flow environment of pools, becoming mobilized during floods (Brookes and Sear 1996).

A large catchment can generally be divided into three zones along its length: the headwater sediment production zone, the sediment transfer zone, and the lowland sediment storage zone (Sear 1996). In the headwater production zone, the breakdown of parent material and the frequent contact between the stream or river with valley slopes allows sediments to enter the stream. As such, land use changes in stream headwaters can alter the amount and types of sediment and biotic material entering the channel (Kondolf et al. 2002).

In the sediment transfer zone, slopes are less steep, and banks are protected from the stream by the floodplain. In this area, the inflow of sediments from upstream matches that exported downstream. Natural erosion and deposition of pools and bars in this zone cause channel migration. Particle transport depends on the shear stress against the bed, the degree to which the particles project from the bed, and the cohesiveness of the material (Dunne and Leopold 1978). Very fine particles require less shear, but can sometimes be cohesive and resist erosion as their small profile makes them difficult to entrain. However, once in the suspended load, turbulence can keep these particles suspended. Large particles require higher shear to be moved, and are the first particles to be deposited when velocity decreases. As a result, the patterns of different velocities within a channel determine the patterns of sediment erosion and deposition, and the particle size of the sediment will vary from location to location (Sear 1996).

In the downstream sediment storage zone, stream power declines, and the transported sediments are deposited, particularly during the decline of high flows (Sear 1996, Knighton 1998). A mix of sediment sizes is found in floodplains, as particles deposited outside the channel. As floodwaters recede velocity decreases, coarser particles are deposited, followed by fine particles which occur as a layer on the surface (Sear 1996). Allochthonous biotic material such as leaves and woody debris is transported and deposited in a similar fashion, especially during flooding. Their integration into the layers of deposited sediment contributes to the soil structure and fertility (Dunne and Leopold 1978, Collins and Kuehl 2001). This sorting of particle sizes and organic materials also affects the water holding capacity of soil (Lyon et al. 1952) and plant community composition (Keddy and Constabel 1986, Bedford et al. 1999).

Geomorphology

Land managers are often most interested in channel geometry, consisting of the longitudinal profile (slope), planimetric geometry, and cross-section (Knighton 1998). This geometry is formed by the erosion and deposition of materials from and within the bed and banks, the patterns of which are affected by substrate and debris qualities, and stream power (Dunne and Leopold 1978, Brookes and Sear 1996). Since stream power is a function of the morphological characteristics of the stream, the formative processes of erosion and deposition are in part functions of the stream's own form (Knighton 1998). In this way, stream morphology constantly changes as erosion and deposition re-align the bed and banks, thus altering stream power and its erosional and depositional effects. This dynamic interaction is further affected by temporal variations in the hydrograph (Sear 1996).

The quantification of flow characteristics causing the majority of channel adjustment is of some debate. While bankfull discharge transports the highest sediment load, flows from bankfull, to rarer high flows, to more frequent lower flows have been correlated to channel morphology, with no clear resolution as to which is most influential (Knighton 1998). Lower flows can often transport more sediment over time due to their frequent occurrence, while infrequent, very high flows can exert the force needed for the formation of cutoffs and oxbows. As such, channel form results from a range of discharges and temporal flow variations. The magnitude of the effective discharge likely depends on flood frequencies and the resistance of channel boundary materials (Sear 1996). As a result, predicting channel parameters as a function of particular discharges may be inappropriate (Knighton 1998).

Channel sinuosity is a function of stream power, bank erodability, and sediment transportability. Naturally straight channels generally have low stream power, producing little bank erosion. A lack of deposition contributes to straightness if local sediments resist erosion yet are easily transported downstream once suspended. Meandering streams with higher stream power experience a balance between localized bank erosion and deposition. The initiation of migration thus requires a minimum stream power, above which straightened channels will naturally re-establish meander patterns (Brookes and Sear 1996). Below this threshold, streams do not have the power to erode banks and beds to produce meanders.

Numerous authors have developed competing explanations for the development of pools and riffles, the exact mechanisms of which are not yet clear. While the deflection of water in the riffle-pool sequence has been credited for meander formation, meanders have been noted to also develop in channels lacking riffles and pools (Brookes and Sear 1996, Knighton 1998). However, the flow patterns within established meanders are relatively well understood; these flow patterns are responsible for further adjustments in channel characteristics over time. The velocity of water is highest at the outside banks within bends and within the channel where it crosses between them. The slant of the stream bed towards the outside bend produces a spiraling current where water at the surface is directed towards the outside bend and water at the bed is directed towards the inside bend (Knighton 1998). This spiral causes erosion at the outer bank just downstream of the curve apex and parallel deposition on the opposite bank in the form of point bars. As a result, meander bends tend to migrate downstream as well as increase in amplitude. The increase in meander amplitude increases channel length, resulting in decreased slope, lower stream power, and increased deposition. Cutoffs can form during periods of high flow, increasing channel slope and sediment transport ability (Sear 1996, Knighton 1998).

The characteristics of braided streams and their formation have received little attention, and debate about their process of formation exists (Knighton 1998). Although some river managers may be prejudiced against braided channels, historical evidence indicates that they may be naturally occurring in some instances (Brookes and Sear 1996). Braided channels generally occur where very high stream power produces extensive erosion of bank materials. Although many forces are likely at play, this form may occur naturally if highly erodable banks allow unconstrained channel migration, yet sediments are not as easily transported. Eroded bank sediments could thus be deposited into the stream bed, forming mid-channel bars (Knighton 1998). Other forms of braid initiation are also possible, but the general consensus is that braided channels are formed by erosional origins which cause frequent shifts in channel position.

Floodplains soils are thought to be formed primarily through the deposition of materials on the insides of bends, as well as in the floodplain during overbank flows. These two processes cause a mix of gravels, silts, and clays to be commonly present in the sediments. Coarser materials are deposited during point bar formation, and fine materials are deposited as a layer within the floodplain and over the point bar materials as flood waters subside (Leopold et al. 1964).

Surface water volume and velocity are affected by stream and catchment geomorphology, as well as runoff rates. Steep stream slopes yield higher velocity flow, while broad, shallowly sloping channels cause water to collect and pool (Knighton 1998). The frequency and volume of flooding changes over time due to the effects of natural hydrograph variations and adjustments of channel morphology (Brookes and Sear 1996). The regulation of water flow from dams and weirs removes these hydrograph variations and often prevents flooding. This loss of overbank flooding, often exacerbated by levee construction and wetland filling, has caused the destruction of innumerable riparian ecosystems. Where wetland and floodplain habitats remain, changes in the hydrologic regime can alter both community composition and sediment characteristics

(Keddy and Constabel 1986, Bedford et al. 1999). Flow regulation by dams and weirs also alters downstream erosion and sedimentation processes, and sediment deficits can occur in lower reaches as deposition occurs on the upstream side of these obstacles (Allan 1995).

Stream velocity and sheer stress play roles in the development, maintenance, and distributions of riparian plant and animal communities. Shear stress often prohibits the establishment and/or survival of many aquatic plants in high-velocity streams by scouring seedlings and propagules. Shifting sediments can affect established plants by causing uprooting and/or injury (Keddy 1982, Day et al. 1988). Aquatic wildlife communities are similarly affected by the force of shear stress, as well as the structure of the channel. Many species are adapted to high flow conditions and are able to resist being washed downstream. A variety of insects maintain populations of delicate aquatic juveniles by the upstream migration of flying adults (Allan 1995). While low velocity streams may displace fewer species, many require the dissolved oxygen concentrations of higher flow systems (Connolly et al. 2004). Channel geometry, the sizes of bed material, and vegetation type also shape wildlife communities by providing different types of breeding, foraging, and attachment sites (Allan 1995).

Morphological disturbances common to developed areas include stream and river channelization, floodplain isolation, filling and/or diking, and the piping of smaller streams (Allan 1995, Ward and Trimble 2004). The resultant cross-sectional area of the channel is often larger than the original during low-flow periods and smaller during high flow periods. The consequences of these modifications include changes in velocity, stream power, and sheer stress (Knighton 1998). The alteration of a waterway to a form that is not supported by its newly altered flow processes (and vice-versa) can result in dramatic destabilization of the channel.

The manipulation of fluvial processes

Fluvial processes are often manipulated in an attempt to affect control over flooding and/or channel planform. This is commonly done in an effort to protect infrastructure or to alleviate the negative effects of previous fluvial manipulations (Allan 1995). In some cases, fluvial processes are managed to improve habitat for individual species or species groups, such as endangered or game fish (Morrison 2002). However, the manipulation of fluvial processes is often done in isolation, without consideration of the other processes that influence aquatic ecosystems (Kondolf and Downs 1996, Tompkins and Kondolf 2003). It is critical that this be avoided, as other ecosystem processes can have indirect yet considerable influence over fluvial processes, and the alteration of fluvial processes inherently causes the alteration of other ecosystem processes and characteristics, whether or not intended. The most appropriate strategy is to address fluvial processes in conjunction with the other ecosystem processes to better ensure appropriate and balanced ecological changes.

There are two general approaches for fluvial process management: management of flow and management of geomorphological form. By their very nature, these approaches are interrelated, and one should not be used without considering the effects of the other. The main foci of flow manipulation are the hydrograph, water velocity, stream power, and shear stress. In degraded ecosystems, the qualities of these features are such that they do not support their desired (historical) ecosystem functions, and may be acting to further degradation. Modification of the flow regime is often best accomplished by influencing the water balance processes, thereby restoring the natural regime of runoff and infiltration. The approximation of natural flow regimes using planned releases from dams and weirs requires careful planning (Shiau and Wu

2004). When such modification is needed, it is important to identify and achieve the daily, seasonal, and annual variations in flow that support the desired ecosystem (Richter and Richter 2000).

Modifying the geomorphological form of streams and rivers can be done using varying degrees of intensity: the complete reconstruction of a stream channel, the installation of in-stream structures to direct water and affect the evolution of the desired stream form, and the modification of catchment processes. The most appropriate method is that which is the least intensive yet will achieve the desired outcome. For the purposes of designing balanced, self-sustaining channels, that stream form is a reaction of the bed and bank particles to the forces of water movement and gravity must not be overlooked. Where the straightening of stream channels has occurred, complete channel construction may be necessary. However, prediction of exact channel form and reaction to future water flows may be impossible due to the current lack of scientific understanding in predicting sediment loads, channel change, and morphology (Brookes and Sear 1996). A currently popular method of directing water flow to achieve the desired erosional and depositional processes uses a variety of in-stream structures made of boulders or organic materials (FISRWG 1998). Sufficient planning must be taken, however, to ensure that the processes influenced by these structures will be fully integrated and balanced with the other structure-forming processes of the ecosystem (Kondolf et al. 2001), and that, when biodegradable material is used, the structures forming the long-term evolution of the system are self-replenishing. Lastly, the most benign method of modifying geomorphology is through the adjustment of runoff processes and sediment inputs. In cases where geomorphological damage is not severe, the stream channel may be able to initiate its own recovery when natural runoff and sediment input regimes are restored. This type of self-design may be the most successful in some cases, as erosion and deposition re-establish a dynamic equilibrium with channel sediments (Knighton 1998).

The study of fluvial geomorphology has produced a myriad of mathematical relationships between many morphological features of streams and rivers. For example, riffles and pools commonly occur at an interval of 5 to 7 times channel width, meander wavelength is approximately 10 to 14 times channel width, meander radius of curvature is about 2 to 3 times channel width, and width is roughly proportional to the square root of discharge (Knighton 1998). These relationships are often used in restoration planning (e.g. following Rosgen 1996); however, they represent generalities about which there is some degree of variation. Variations in morphological patterns along an individual reach are produced due to physical factors which vary both between and within catchments (Knighton 1998). As a result, streams may be inherently asymmetric and there is no guarantee that a particular feature (such as meanders) will be regularly spaced (Knighton 1998).

Adequately addressing fluvial processes often requires gathering a considerable amount of information. The past, present, and likely future characteristics of the stream and catchment are all important components (Kondolf and Downs 1996). A historical survey should be performed to understand the correlations, patterns, and consequences of past land use, sediment and deposition patterns, and channel morphology (Sear 1996). The current upstream catchment influences, such as sediment and water sources, must be understood, as well as the potential influence from future land use change (Brookes and Sear 1996). While qualitative assessments (like analysis of catchment history) do not lead to specific channel design, they can provide guidance on appropriate morphology and probably dynamics, and can help determine where detailed modeling is needed to protect infrastructure (Brookes and Sear 1996).

Modeling of fluvial geomorphology can be an essential component of restoration projects (Knighton 1998, Kondolf 1998). A variety of models have been developed, and can be classified as physical, conceptual/theoretical, statistical/empirical, analytical, or numerical simulations (Darby and van de Weil 2003). Each of these models has its strengths and weaknesses, and the best model for a given circumstance depends on factors such as the required output, data needs, and model complexity, as well as the applicability of the model to the system being addressed (Ward and Trimble 2004). However, care must be taken to use models appropriately. Statistical models can be misused when applied to rivers that are not similar to those used in model development, or if used to predict historical discharge based on observed channel dimensions (since the channel dimensions are dependent variables in models of geomorphology) (Darby and van de Weil 2003).

Chemical Processes

Chemical processes and their effects

The importance of chemical processes in ecosystem development is often overlooked by restoration designers. However, these processes, particularly those that regulate the concentrations and availability of nutrients, play a considerable role, influencing both ecosystem type and trophic status. Wetlands, streams, and lakes can vary from oligotrophic to eutrophic based on the availability of nutrients, especially nitrogen and phosphorus (Horne and Goldman 1994). Spatial and seasonal chemical variability affect community structure and species distributions within individual ecosystems, even at the microsite level (Breen et al. 1988). The ability to understand, predict, and manipulate chemical processes is therefore as important as any other aspect of ecological restoration.

Oxygen availability and anaerobiosis

The chemistry of aquatic and wetland ecosystems is markedly different from that of terrestrial systems, primarily caused by the limited availability of oxygen in saturated and inundated soils. The resulting anaerobic conditions affect both organisms and soil water chemistry. Decomposition of organic materials by anaerobic bacteria results in reduction reactions of NO_3^-, MnO_2, $Fe(OH)_3$, SO_4^{2-}, and CO_2 (Ponnamperuma 1972). The rate at which these reactions occur affects the concentration, form, and distribution of these compounds in the ecosystem. Soil pH is also affected, such that oxygenated soils with high decomposition rates can be very acidic while reduction reactions in submerged soils can raise pH (Ponnamperuma 1972, Burbage 2004).

Oxygen availability and decomposition rates both affect and are affected by the other. The consumption of oxygen during aerobic respiration reduces the concentration of oxygen available for further decomposition. As a result, organic matter accumulation is common in reduced sediments, and more aerated sediments, where decomposition and respiration can be highest, contain more mineral content (Lyon et al. 1952, Ponnamperuma 1972). Soil structure and water holding capacity (among other characteristics) are affected by carbon content (Ponnamperuma 1972), which can form a gradient in wetland soils at the edge of standing water (Burbage 2004).

Plant zonation is most influenced by anaerobiosis caused by flooding. Numerous studies have demonstrated that plant zonation and community development are strongly tied to the depth, duration, and frequency of flooding (e.g. Squires and van der Valk 1992, van der Valk 1994, van der Valk et al. 1994, Lenssen et al. 1999a, Lenssen et al. 1999b, Casanova and Brock 2000, Rheinhardt and Faser 2001, Burbage 2004).

Aquatic wildlife can be severely impacted by low dissolved oxygen concentrations (Connolly et al. 2004). Such conditions can be naturally occurring; however, the decomposition of increased biomass in eutrophied systems can lower dissolved oxygen concentrations, resulting in fish kills (Parr and Mason 2003).

<u>Temperature</u>

The temperature of man-made reservoirs can be important factors in riparian restoration, affecting water quality in downstream reaches (National Research Council 1992). Thermal stratification of lakes and ponds occurs during warmer months when warm surface waters mix very little with cool, deeper water. These layers can mix (turnover) as temperatures decline in the fall and again, in icy climates, in the spring (Horne and Goldman 1994). Such stratification has important consequences in the availability and cycling of nutrients and the distributions of organisms in lakes. Chemical stratification of water bodies is common during warmer months, when dissolved nutrients at the surface can be depleted by autotrophic organisms. The sinking of these organisms, or their consumption by stronger-swimming heterotrophs, can move the nutrients into deeper water. Spring and fall turnovers redistribute these nutrients, sometimes resulting in algal blooms. Stratification can also result decreased oxygen concentrations at lower depths, due to the lack of mixing with oxygenated surface waters. Such stratification affects the survival and distributions of lake organisms (Horne and Goldman 1994). Oxygen concentrations are also affected by temperature in running waters; both of which influence aquatic wildlife distributions (de la Hoz Franco and Budy 2005). Water quality in downstream systems can decline if deep reservoir water is discharged; this water may have very low dissolved oxygen (DO) concentrations and elevated concentrations of phosphorus, ammonium, iron, manganese, and hydrogen sulfide (National Research Council 1992).

Acidity, alkalinity, and buffer capacity

The biotic communities of ecosystems are strongly influenced by pH and alkalinity (Clark 1986, Nicolet et al. 2004), as are certain enviro-chemical reactions. The pH of an ecosystem is regulated by the presence of calcium, magnesium, sodium, and other base cations in soils, sediments, and waters (Lyon et al. 1952). Although carbonic acid is naturally formed when carbon dioxide dissolves in water, the pH of the ecosystem is buffered by equilibrium reactions between this acid and base cations. As a result, the majority of aquatic systems maintain circum-neutral conditions. Base cations are supplied to aquatic systems via groundwater, into which they dissolve as the water passes through the soil (Horne and Goldman 1994). Buffering can also be facilitated by clay particles, which have a chemically charged surface (Lyon et al. 1952).

Some distinctive ecosystems are found in acidic and alkaline environments. The concentration of base cations can be very high where groundwater passes through alkaline soils (such as those containing limestone or dolomite), or in arid regions where very high evaporation rates concentrate the dissolved minerals. Photosynthesis in such alkaline waters can cause calcium carbonate to precipitate. The white precipitant may remain suspended, dramatically reducing light penetration. Marl soils are formed if the precipitant settles; it can also form a solid coating on rocks called tufa (Horne and Goldman 1994). Many plants have difficulty growing in calcareous soils; in part because many nutrients become sequestered (Rowell 1988). In addition, water hardness and pH can affect the toxicity of some metals and other substances on fish (Barron and Albeke 2000, Pyle et al. 2002).

Very acidic soils are toxic to most organisms. Consequently, naturally occurring acidic ecosystems are populated with specially adapted species, some of which can be quite rare. Peat bogs are an example of a community that thrives in very acidic water (Mitsch and Gosselink 1993). Such acidic conditions occur naturally in bogs and similar ombrotrophic systems, where the primary water source is direct precipitation. Since raindrops do not contain base cations, the resulting ecosystem has a very low buffering capacity (Vepraskas and Faulkner 2001). In addition to acidic conditions, the low availability of trace nutrients such as calcium, magnesium, and potassium, in these types of environments can strongly affect community composition (Mitsch and Gosselink 1993). Acidic conditions can also develop at the surface of very well drained soils, such as those of the Midwest, where dissolved carbonates have leached into lower levels (Rowell 1988).

The effect of soil acidity on iron and aluminum can also play an role in the distributions of plants between and within systems (Snowden and Wheeler 1993). In acidic soils, iron and aluminum can be solublized; high levels of which can cause plant toxicity and death (Tan 2000), especially near seeps (Whittecar and Daniels 1999). Conversely, the leaching of metals in old acidic soils can cause metal deficiencies (Cresser et al. 1993), affecting community composition.

Sulfidic soils (common in coastal areas, arid alkaline environments, and areas with high concentrations of organic matter (Mitsch and Gosselink 1993, Tan 2000) can pose a special problem in ecological restoration. Toxic to plants, sulfides can affect distribution and germination, depending on plant sensitivity (Seliskar et al. 2004). Sulfidic acidification is commonly seen as a result of coal mining and acid rain (Mitsch and Gosselink 1993) and can be caused by the oxidation of sulfidic soils during excavation (Whittecar and Daniels 1999).

<u>Nutrient cycling and availability</u>

The restoration of aquatic communities often requires restoration of the availability and cycling patterns of nutrients. These patterns can be critical factors in community development

(Bedford et al. 1999), and uptake rates of the various nutrient forms can differ between species. The total concentration of nutrients in an ecosystem is a function of their input from and export to adjacent environments and the atmosphere (Mitsch and Gosselink 1993). However, the concentration of a nutrient available for plant uptake is dependent on the chemical state of the nutrient, the location of the nutrient (e.g. in soils or tissues) and the nutrient turnover rate (Jonasson and Shaver 1999). As such, an understanding of the processes that affect nutrient states is necessary, so that these processes may be addressed, manipulated, and/or accounted for in restoration design.

Nitrogen. Nitrogen in often the most limiting nutrient in flooded soils, and is transformed between six main pools: ammonium, nitrite, nitrate, dinitrogen gas, organic N in living tissues, and organic N in dead tissues and organic compounds dissolved in the water column (Figure 2.2) (Mitsch and Gosselink 1993). While nitrogen bound in plant tissues is generally unavailable to other plants (Raven et al. 1986), the decomposition of biotic matter releases ammonium-N, which can be immediately taken up by plants and plankton. In aerobic sediments, nitrification converts ammonium to nitrate (with nitrite as an intermediate compound) (Mitsch and Gosselink 1993). Such sediments are usually present as a very thin layer at the surface of wetlands and other water bodies (Mitsch and Gosselink 1993, Horne and Goldman 1994). However, decomposition in anoxic soils can cause ammonium to accumulate. Transformation of this ammonium into nitrate occurs as it diffuses into the upper

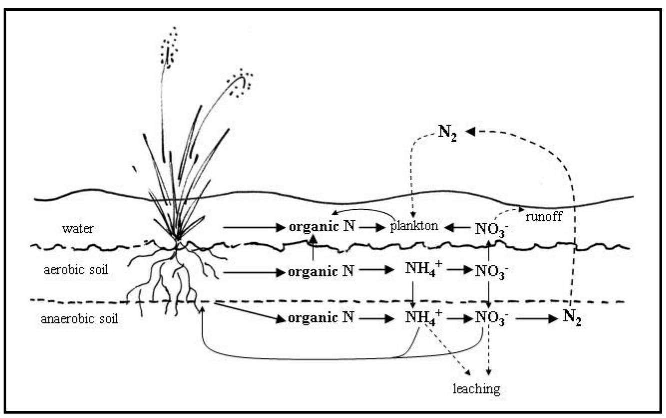

Figure 2.2: The nitrogen cycle in aquatic ecosystems

oxidized soil layer, becoming available for uptake by plants and plankton. Nitrate that diffuses down into the lower anaerobic sediments can be either taken up by plants or converted to nitrogen gas and lost from the system. Certain cyanobacteria and aerobic and anaerobic bacteria are able to fix nitrogen gas into organic nitrogen, which can be a considerable source of N in some wetlands (Mitsch and Gosselink 1993).

Nitrogen availability, cycling and transformations are controlled by a variety of influences (Mitsch and Gosselink 1993). Anoxic conditions can limit transformation rates, as decomposition occurs more slowly, and ammonium must then diffuse into oxidized soils for nitrification. In some aquatic systems, anoxia caused by the absence of water movement can result in the accumulation of ammonium in soils. Conversely, nitrification and denitrification rates can be increased in sediments with fluctuating water levels (Burbage 2004). Nitrogen transformations can also be affected by pH and leaf-litter quality (Mitsch and Gosselink 1993, Aerts et al. 1999).

Phosphorus. Phosphorus sources in aquatic systems include allochthonous materials, eroded sediments with bound P (especially clays), and discharged wastes such as that from animal production and wastewater treatment facilities (Horne and Goldman 1994). Once in the aquatic system, P occurs in four main pools: soluble orthophosphates, fixed mineral P, organic P in living tissues, and the organic P of dead tissues and dissolved organic compounds (Figure 2.3) (Vepraskas and Faulkner 2001).

Plants can only absorb P as dissolved orthophosphate (Wild 1988), making most phosphorus in freshwater systems biologically unavailable for plant growth (Horne and Goldman 1994). In addition, most orthophosphate is bound by sediments, particularly clays (Horne and Goldman 1994), by its adsorption to iron and aluminum oxides and hydroxides, as well as calcium and magnesium (Vepraskas and Faulkner 2001). However, soils have a finite P adsorption capacity, above which orthophosphate will remain in solution. Additionally, when iron is reduced in anoxic sediments, iron-bound orthophosphate is released into solution. Orthophosphate can then diffuse into the overlying water if anoxic conditions persist. This dissolved phosphorus is distributed throughout ponds and lakes during fall and spring turnover. These cycles of P sorption and release in lake sediments can cause internal loading of lake waters (Horne and Goldman 1994). Phosphorus can also be transported from sediments into the water column by its uptake in aquatic macrophytes and subsequent release in decaying above-ground portions (Horne and Goldman 1994).

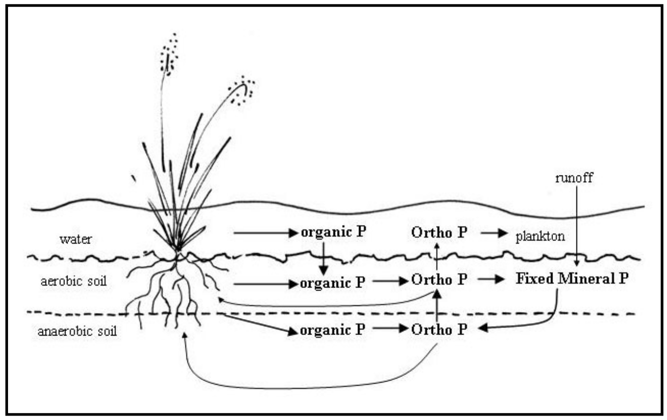

Figure 2.3: The phosphorus cycle in aquatic ecosystems

Lastly, P availability can be affected by the pH of aquatic systems. Where pH >7 (e.g. in calcareous areas), P precipitates as calcium and/or magnesium phosphates. At the low pH (<4) common to bogs and other ombrotrophic systems, iron and aluminum oxides and hydroxides dissolve and orthophosphate is released (Vepraskas and Faulkner 2001).

Nutrient availability, especially nitrogen and phosphorus, can strongly affect the community composition and spatial organization of plants within habitats types (Bridgham et al. 1996, Vaithiyanathan and Richardson 1999, Keddy et al. 2000, Svengsouk and Mitsch 2001). While some species (e.g. *Typha* sp.) can dominate in high nutrient environments, other species only escape competitive exclusion in low nutrient environments (Wisheu and Keddy 1992).

Alterations of chemical processes are common in urbanized and developed lands. Eutrophication is especially common in urbanized and agricultural areas due to nutrient inputs from fertilizers, wastewater treatment, and soil erosion (Allan et al. 1997, Paul and Meyer 2001, Brinson and Malvarez 2002). The loss of wetlands, which often function as nutrient sinks (van der Valk et al. 1978), can also result in increased nutrient loading of surface waters (National Research Council 1992). Increased concentrations of nitrogen and phosphorus often have dramatic effects on plant communities and other chemical processes. Although this usually results in increased primary production, species richness often declines as a result of competitive exclusion and the prevention of seedling establishment (Foster and Gross 1998, Bedford et al. 1999, Kirkman et al. 2001, Paul and Meyer 2001).

Eutrophication can cause a complete shift in community type (Bedford et al. 1999, Bayley and Mewhort 2004), and/or shifts in the dominant life forms in some systems (Roman et al. 2001). In ponds and lakes, increased nutrient loading can cause phytoplankton blooms and alter the composition of phytoplankton species (Horn 2003, Moraska-Lafrancois et al. 2003). It has also been shown that high nutrient loading can shift macrophyte-dominated aquatic systems to dominance by phytoplankton, with the extirpation of submerged macrophytes (Morris et al. 2003). The magnitude of such changes can depend on the previous nutrient status of the ecosystem, the types of nutrients added, and seasonal effects. For example, low nutrient systems such as bogs and fens are particularly sensitive to alterations in nutrient availability (Bedford and Godwin 2003). In addition, trophic status can be affected by the relative concentrations of different nutrients; aquatic systems can be limited by phosphorus, nitrogen, or both (Elser et al. 1990).

While changes in land use can result in nutrient loading, the removal of vegetation immediately adjacent to streams results in lowered detrital inputs (both leaves and coarse woody debris) (Allan 1995). The removal of these sources of carbon can alter the texture and lower the water holding capacity of soils (de Macedo et al. 2002), thus affecting plant communities (Thompson and Troeh 1957, Lenssen et al. 1999b). Plant communities can also be affected by increased scour (Keddy 1985, Roberts and Ludwig 1991) caused by the reduction of large woody debris. Dissolved and particulate carbon, however, are often increased in waters receiving wastewater treatment effluent (Paul and Meyer 2001), resulting in higher biological demands and lower oxygen concentrations (Allan 1995).

Chemical processes indirectly influence wildlife communities by their effects on the vegetation and algae that make up forage, cover, and breeding sites; changes in nutrient availability thus alter the structure and availability of these resources. In aquatic systems that exhibit bottom-up control of trophic interactions, eutrophication can result in dramatic changes in and destabilization of the food web (Hecky 1993). The chemical characteristics of the ecosystem even affect the nutrient quality of vegetation and algae (Kubín and Melzer 1996). Zooplankton and invertebrate community structure can depend on algal species composition and nutrient quality (Rosemond et al. 1993, Darchambeau and Thys 2005); changes in lake nutrient status has been shown to influence zooplankton communities by affecting phytoplankton (Gulati and van Donk 2002, Arhonditsis et al. 2003). The structure and diversity of vegetation also influence fish assemblages (Tonn and Magnuson 1982, Petry et al. 2003) and other wildlife that live in or frequent aquatic systems, such as amphibians and waterfowl (Edwards and Otis 1999, Parris and McCarthy 1999, Paracuellos and Telleria 2004).

The manipulation of chemical processes

A common goal in environmental management is to manipulate chemical processes in an attempt to support desired biota and/or change the trophic structure of an ecosystem (Clemente et al. 2004). This is often done to mitigate anthropogenic effects such as eutrophication and low oxygen conditions (National Research Council 1992).

Depending on project goals, an increase or decrease in the availability of oxygen may be desired. Management for oxygen-sensitive wildlife species may require an increase in dissolved oxygen concentrations. Conversely, the management of other chemical processes often requires the decrease of oxygen concentrations in soils and water so that the desired chemical transformations (e.g. denitrification) can be supported (James et al. 2004). Oxygen levels in water can be increased by decreasing biological oxygen demand (by decreasing decomposition rates) and/or temperature, or increasing the rate of photosynthesis (using care not to increase decomposition) and/or turbulence. Dissolved oxygen may be lowered by the opposite actions. Soil oxygen levels may be reduced by maintaining saturated conditions, or increased by draining, water level fluctuation, or the percolation of highly oxygenated water (Hammer 2000, Vepraskas and Faulkner 2001).

A reduction of water temperature can be an objective in projects addressing temperature- and oxygen-sensitive wildlife (National Research Council 1992). Temperature can be lowered by increasing the amount of shade overhanging the water body as well as in the entire catchment. Where increased temperatures are the result of runoff from heated pavement, temperature can be lowered by slowing the flow of runoff and directing it through shady buffers where the increased temperature can dissipate before entering the water body (National Research Council 1992, Allan 1995).

The manipulation of nutrient cycles and nutrient availability is required to reduce eutrophication and its effects, or to prevent and/or mitigate nutrient loading as a result of new development (Hammer 2000). In some cases, these processes could also be manipulated to manage for sensitive rare species that are generally restricted to oligotrophic environments. There are three strategies which may be used to influence nutrient cycling and availability: 1) nutrient management at the watershed level, 2) the use of natural systems to remove nutrients, and 3) the chemical treatment of the water body.

Sources of nutrients from the watershed include points sources such as combined sewerage overflows, wastewater treatment discharge, and industry discharge (Allan 1995, Ward and Trimble 2004), the management of which can be implemented at the source. More difficult is the reduction of nutrients from non-point sources (NPS). NPS pollution is a significant contributor to local and regional water quality problems (NVPDC, 1996). Common sources of NPS nutrients include fertilizer runoff from agriculture and silviculture, livestock waste, fertilizers and sediments from urbanized areas, and nitrogen and sulfur from atmospheric deposition (Neary et al. 1989). BMPs can be used to reduce nutrient loading from agricultural areas. The aims of such practices include increased infiltration and decreased runoff, livestock exclusion from surface waters, maintenance of forested riparian zones, and management of nutrient applications to prevent leaching and losses (Hairston et al. 2001). Nutrient loading in urbanized areas can be mitigated by controlling land use, maximizing vegetated areas and the infiltration capacity of soils (especially in riparian areas), the creation of stormwater wetlands, and public education about low impact landscaping, pet waste, and stormdrains (NVPDC, 1996).

Vegetated buffers and wetlands are natural systems that can be used to remove nutrients from runoff and surface waters. That wetlands act as nutrient sinks is well appreciated by land managers and ecologists, and the design of wetlands for wastewater treatment, stormwater, and NPS pollution receives much attention (Mitsch and Gosselink 1993, Davies 1995, Hammer 2000). Wetlands remove nutrients through plant uptake, trap nutrient-adherent sediments, and encourage denitrification. The placement of wetlands for nutrient removal depends on the nutrient source. Where runoff is the primary nutrient source, several small wetlands in the upper reaches of the watershed may be most effective (Marble 1992, Mitsch and Jørgensen 2004). Where a water body is eutrophied, flow directed into an in-stream or bordering wetland can reduce nutrients. Although expensive, water can be pumped from the main water body into treatment wetlands where natural flow is not feasible (Mitsch and Jørgensen 2004); this type of constructed wetland has shown promise in some studies (Nairn and Mitsch 2000, Jing et al. 2001).

When designing wetlands for nutrient removal, site hydrology must receive specific attention. Permanently flooded and/or saturated soils are most desirable, as is a gradual basin gradient, and low water velocity (Marble 1992). Organic soils are necessary for nitrogen removal (Marble 1992), as are anoxic soil conditions. However, anoxic conditions in overlying waters should be avoided, as denitrification can be interrupted and accumulation of ammonium in sediments can result (Burbage 2004). The adsorption of phosphorus is best effected by ferric or clay soils with a minimum alkalinity of 20 mg/l (Marble 1992). Since adsorption of phosphorus by soils is finite, regular maintenance by dredging and replacement of soils may be required (Hammer 2000). Where uptake of nutrients by plants is desired, sheet flow is preferred to maximize uptake (Marble 1992) and vegetation harvesting may be necessary to prevent detritus (and nutrients) from re-entering the system (Hammer 2000).

Vegetated buffers are an easy, cost-effective way to improve water quality, especially in agricultural areas. Similar to wetlands, buffers remove nutrients through plant uptake, encouraging denitrification, and trapping sediments; they are also very effective at reducing surface runoff. Buffer types must vary depending on which types of vegetation are appropriate to the region (grasses, shrubs, or forest) and climate (cool-season or warm-season) in order to maximize permanent nutrient removal (Dosskey 1998, Lee et al. 1999, Perry et al. 1999). The width of buffer strips is also important; wider strips can remove considerably more nutrients (Lee et al. 1999).

In some lentic systems, internal nutrient loading prevents sufficient water quality improvement, even when runoff is adequately managed (Cooke et al. 1993, Horne and Goldman 1994, Présing et al. 2001, Søndergaard et al. 2003). In these cases, chemical treatment of the water may be appropriate to immobilize phosphorus, and iron and/or aluminum compounds can be used (Cooke et al. 1993). However, careful consideration of other water quality parameters is necessary to achieve positive results (Hansen et al. 2003). Where iron additions are used, phosphorus can be re-released into the water under anoxic conditions. As a result, surface sediments must be oxidized via aeration or nitrate concentrations must be increased to prevent iron reduction (Hansen et al. 2003). While aluminum sulfate can bind phosphorus ions permanently, the duration of phosphorus reduction from a single treatment is unknown, especially if external loading continues (Steinman et al. 2004). In addition the use of aluminum in softwater lakes can cause acidification and the formation of toxic aluminum compounds, and the use of aluminum sulfate can cause the formation of pyrite in iron-rich waters (Hansen et al. 2003). Lastly, the effect of chemical additions to surface waters on biota must be considered (Reitzel et al. 2003).

The acidification of freshwater ecosystems is a continuing concern. Lime is commonly used in both catchments and surface waters as a means to increase buffering capacity and alleviate the effects of acid rain and acid mine drainage (Covert 1990, Dorland et al. 2005, Petty and Thorne 2005); wollastonite (a silicatious mineral) can also be used (Likens et al. 2004). As is the case for chemical additions for phosphorus control, the effect of altering the buffering capacity of the ecosystem on all resident or potential organisms must be considered. This may be of special importance in areas where peat bogs are present and the protection of rare endemic species should be ensured.

Biotic Processes

Biotic processes and their effects

While water balance, fluvial, and chemical processes can lay the physical foundation in a restoration plan, it is the actions and interactions of organisms that establish the community. Life-cycle processes, community, and trophic dynamics influence the abundance of each species, its distribution, and its role in the ecosystem. In addition, organisms alter the physical environment through the very acts of living, eating, breeding, and dying. As a result, the processes of organisms affect both the structure and functioning of ecosystems (Chapin et al. 1997).

Population dynamics

Population size is a function of the rates of birth, death, immigration to and emigration from the population area (Begon et al. 1990). The dynamics of plant populations are somewhat more complex, due to the indeterminate size and vegetative reproduction of many species, as well as seed dormancy (Barbour et al. 1987). A species reproductive rate is affected by its life-history strategy and resource availability. Two life-history strategies have evolved to accommodate competition for resources and resiliency to high-mortality events (such as physical environmental disturbance or predation). Species of stable populations tend to be resource-limited, enduring high levels of intraspecific competition (k-selected species). These species tend to be larger, older at maturity, and produce larger young with more parental investment; reproductive rates are consequently lower. In contrast, species of fluctuating populations reproduce quickly to become re-established after disturbance (r-selected). These species often mature at smaller sizes and produce smaller, easily transported offspring with less parental investment (Whittaker 1975).

While in natural systems k- and r-selection strategies promote long-term community stability, communities often shift when frequent anthropogenic disturbances favor r-selected species. Community shifts can also occur as the result of exotic r-selected species introduction. These exotics may become invasive not only because they are released from predation but because the population is not subjected to the disturbance patterns of its original environment, allowing it to increase towards carrying capacity (McMahon 2002).

Death rates are affected by the life-history traits of the species, predation or herbivory, and resource limitations. Some organisms do not live for more than one season, and an entire population of adults may die at roughly the same time. Predation/herbivory rates can be variable depending on the behavior, type, and abundance of predators. Resource limitation can lead to higher death rates or even the competitive exclusion of a species (Barbour et al. 1987, Begon et al. 1990).

Factors that influence immigration can be especially important in restoration, where the establishment of additional species is desired (or, in the case of exotics, undesired). Immigration and emigration rates depend on the species life-history and migratory habits. While some wildlife species migrate for long distances annually, some move between habitats daily, and others remain within a relatively small territory (Begon et al. 1990). Local migration can be inhibited by habitat fragmentation. Fragmentation generally leads to lower species richness, and some species need rather large contiguous areas to maintain viable populations (Nilsson 1978, Saunders et al. 1991). Aquatic fragmentation can occur in the form of anthropogenic structural changes (such as dams); these features can result in genetic isolation and/or the extirpation of some species from upstream areas (Scruton et al. 1998). Plants must rely on external forces such as wind, water, animals, and gravity; some species are particularly dependent on specific wildlife species for dispersal (Barbour et al. 1987, Traveset and Riera 2005). Effective seed dispersal of many fruiting plants requires animal digestion to encourage germination (Begon et al. 1990), and specific animal species may be required for pollination (Barbour et al. 1987). For these reasons, if immigration is required to establish a restored population, care must be taken to ensure that the mode of migration is available (Bond and Lake 2003).

Community dynamics

Processes affecting community structure include interspecific competition, mutualism, facilitation, and trophic interactions (e.g. predation and parasitism). The succession of species over time within a habitat is caused by these forces coupled with disturbance. Competition for resources limits species distributions and abundances, while mutualism and facilitation can extend a species distribution by the creation of favorable microhabitats by other species (Begon et al. 1990).

Plant communities form the basis of habitat for many wildlife species. The plant community determines the types and amounts of a variety of different forage foods, breeding sites, and cover (Hammer 2000). Plant community composition is dependent on the potential species pool and interspecific competition. The potential species pool is affected by reproductive rate and dispersal, as well as the ability to survive the environmental conditions of the habitat (van der Valk 1981). The distribution and/or competitive exclusion of species in the pool is determined by competitive interactions (for space, light, and nutrients) (Wisheu and Keddy 1992). Community composition after disturbance can be also affected by the order of species re-introduction (Keddy 1999). The introduction of exotics alters both the potential species pool and the structure of competitive interactions. Coupled with anthropomorphic disturbances, r-selected exotics can sometimes outcompete native species and form monospecific stands (Galatowitsch et al. 1999, McMahon 2002). Similarly, the introduction of exotic animal species or re-introduction of extirpated species can affect the native wildlife population and affect trophic interactions (O'Dowd et al. 2003, Fortin et al. 2005).

Trophic dynamics. While the energy in all food webs originates from primary production, the source of organic matter in aquatic systems may originate within the system from cyanobacteria, algae and/or plants (autochthonous matter) or from the surrounding environment as organic matter carried by wind, water, or animals (allochthonous matter) (Horne and Goldman 1994, Allan 1995). The rate of primary production and/or input, types of plant materials, and nutritional quality are important in aquatic trophic dynamics (Rosemond et al. 1993, Meerhoff et al. 2003); these factors are influenced by chemical processes, such as nutrient levels and pH (Hayati and Proctor 1990, Kubín and Melzer 1996, Bedford et al. 1999, Lenssen et al. 1999b, Keddy et al. 2000), physical environmental characteristics such as light, soil texture, and soil water, and disturbance (Keddy 1982, Keddy and Constabel 1986, Decocq et al. 2004).

The processes of various trophic levels are interrelated. For example, plant and/or algal community composition can strongly affect herbivore communities (Hecky 1993, Wallace et al.

1997), while grazing impacts plant communities (Taylor and Grace 1995, Duffy 2002, Gordon et al. 2004). Similar is the relationship between the types and availability of prey and predator communities (Petchey 2000). Predators indirectly influence plant communities by affecting herbivores populations (Scasso et al. 2001, Duffy 2002). Lastly, the role of detritivores and invertebrate soil organisms in trophic dynamics should not be ignored (Wallace et al. 1997). This group of consumers is responsible for aiding in the decomposition of biotic material and thus affects nutrient cycling and availability, as well as soil texture (Pringle et al. 1999, Gessner and Chauvet 2002). Many detritivore species are also prey items (Wallace et al. 1997).

The net effects of production, herbivory, and predation on trophic structure depends on a delicate balance of the influences of the component organisms. In some environments, trophic dynamics are controlled by the types and abundances of primary producer input (both autochthonous and allochthonous). Trophic dynamics in these 'bottom-up' communities are largely determined by the type and availability of forage for herbivores and, indirectly, predators (Rosemond et al. 1993, Horne and Goldman 1994). Consequently, the alteration of plant and/or algal communities in these systems can affect the populations of higher trophic levels (Meerhoff et al. 2003). The trophic dynamics of 'top-down' communities are regulated by predation pressure on herbivores, subsequently influencing plant and algae communities and even the physical structure of the habitat (Horne and Goldman 1994, Duffy 2002). As such, competition between plant species is often reduced and less-competitive species can attain greater densities than would occur without herbivory (Fortin et al. 2005)

Environmental modifiers. Many organisms modify their environments, both directly or indirectly. To support their needs, some animal species considered to be 'environmental engineers' actively manipulate the environment to the extent where entire habitats may be altered. The creation of beaver ponds is an example of such modification, where a wooded streamside area can be converted to an open pond containing submerged and/or floating aquatic plants (Ray et al. 2001), fish and other lentic wildlife (Ray et al. 2004), and wetland-associated terrestrial organisms (Edwards and Otis 1999). Such modification also affects the downstream hydrology, soil structure, and chemical processes (Johnston 2001). Other forms of environmental modification are more subtle, such as the building of dens or nests. These smaller modifications can impact the biotic community at the microsite scale by changing hydrology, soil texture, fertility, and chemical characteristics (Berg and Kangase 1989, Duffy 2002).

Vegetation influences land forms, stream morphology and water quantity and quality. The use of plants for soil management is common in both aquatic and upland systems (Whisenant 1999). Plants encourage soil deposition (Sear 1996) and protect soils from the forces of water and erosion, through stabilization with roots, rain interception, and increased surface roughness (Knighton 1998, Mallik et al. 2001). In areas with dense vegetation, runoff is slowed and the effects of storms on water velocity, volume, and turbidity in receiving waters is considerably lessened. Evapotranspiration removes water from both upland areas in the watershed as well as from within wetlands and streambanks, and rates can have a surprisingly great effect on water quantity, depending on the amount of vegetation and species present (Dunne and Leopold 1978). Conversely, plant shade increases the amount of water at the soil surface by providing protection from both sun and wind (Ward and Trimble 2004). The influence of vegetation on stream and river morphology, the hydrograph, and sheer stress affect fish and amphibian communities (Ward and Trimble 2004). Consequently, the removal of vegetation due to development and/or agriculture has tremendous impacts (Knighton 1998).

Environmental modification as the result of plant succession can be an important issue in ecological restoration (Kirkman et al. 2000, Kellogg and Bridgham 2002). While successional trajectories are constrained by species immigration and interspecific competition, the success of many species can be impacted through facilitation by previous species. For example, organic matter accumulation in soils can be dependent on the community composition of the vegetation (Wigginton et al. 2000). An increased water-holding capacity of soils associated with increased organic matter (Lyon et al. 1952) could consequently support more drought-sensitive species during periods of drought (Sturtevant 1889).

Plant communities can also have a strong impact on nutrient dynamics, the effect of which depends on the individual species and the limiting factors of the environment (Marble 1992, Tanner 1996, Bachand and Horne 2000). Vegetation can serve as either a sink or source of soil nutrients in aquatic systems. For example, plant uptake of nutrients often serves as a nutrient sink in wetlands receiving runoff (Mitsch and Gosselink 1993, Tanner 1996). Conversely, plants serve as a nutrient source in densely shaded headwater streams through the input of leaf litter (Allan 1995). Lastly, plants affect soil chemical processes by the oxidation of the rhizosphere. Anoxia is prevented in small areas around the root due to the leaking of oxygen, consequently influencing nutrient (especially nitrogen) transformations (Chasar et al. 2000, Vepraskas and Faulkner 2001). Therefore, alterations of the vegetation dynamics of an aquatic habitat can disrupt both source/sink and aeration processes, altering nutrient concentrations in both soil and water (Mitsch and Gosselink 1993, Allan 1995). The loss of nutrient sinks in the headwaters of a system can result in increased nutrient loads (Whigham and Jordan 2003, Yeakley et al. 2003) potentially altering downstream vegetation dynamics and wildlife communities. For example, increased nutrient availability can allow species such as *Typha* sp. to become dominant competitors (Richardson et al. 1999). In such circumstances, nutrient-facilitated competitive exclusion can lead to a monoculture of the dominant species. Increased nutrient availability in lentic systems can cause intense algae production and increased turbidity; such algal turbidity can sometimes be directly controlled by zooplankton grazing and indirectly by planktivorous fish (Horne and Goldman 1994).

In addition to providing forage, plant communities provide the physical structure for wildlife cover, nesting, and nursery habitats (Lehtiniemi 2005). Fish, amphibian, and invertebrate communities also benefit from lower water temperatures provided by riparian shade (Ward and Trimble 2004, Watanabe et al. 2005). Increased stream, river, and lake temperatures are often caused by riparian tree removal (Allan 1995).

Special considerations

Keystone species. Certain species are considered to be critical to the functioning of certain environments, and have greater effects on ecosystems than would be predicted by their biomass. These species may have strong trophic impacts or may be influential environmental engineers. In all circumstances, keystone species are thought to directly and indirectly impact many aspects of their habitat, such as trophic dynamics, physical structure, and fluvial and chemical processes (Mills et al. 1993). Many wildlife species can be specific in their habitat requirements, and the lack of a required plant species can result in the inability of an ecosystem to attract or sustain the desired wildlife (Hammer 2000). The effect of the removal of keystone species on environmental processes can lead to a host of adjustment interactions that destabilize the entire community. For example, the extirpation of the red land crab (*Geocarcoidea natalis*) on Christmas Island (due to crazy ant [*Anoplolepis gracilipes*] introduction) caused what has been called “an invasional ‘meltdown’” of community dynamics (O'Dowd et al. 2003).

Biodiversity. Of great concern to many environmental managers is the topic of biodiversity. Within individual habitats, high biodiversity is often thought to be beneficial due to the functional redundancy of the component species (the insurance hypothesis) (Yachi and Loreau 1999). However, of some debate is the question of whether a diversity of species is always desirable, or whether a goal of diversity of habitats is more appropriate (Kemp et al. 1999, Ward et al. 2001).

The manipulation of biotic processes

Biotic processes are often manipulated in an attempt to affect a specific population or environmental characteristic (Morrison et al. 1994). Unfortunately, this narrow focus has the potential to negatively affect non-target communities and characteristics by overlooking the intricate network of interactions between species and trophic levels. In order for a restoration project to yield a high probability of success, all interactions of an ecosystem's biotic components should be considered. Although common management goals include addressing production, herbivory, and predation rates, as well as individual population sizes, the effects of modifying one or more of these processes on other processes must be considered. While the improvement of a population of a rare species may be desired, a stable community can only be established when all other biotic effects are in balance (Tockner et al. 1998). Therefore, a well-designed restoration project will carefully analyze the existing biotic community and attempt to predict potential changes that may occur due to biotic manipulation.

The vegetation of the ecosystem is often addressed as a means to stabilize soils and affect soil and water quality. Such manipulations can include the establishment of stream and wetland buffer zones, as well as the establishment of upland vegetation, to retain nutrients and slow runoff (Parkyn et al. 2003, Ward and Trimble 2004). Particularly common is the introduction of plant materials along stream banks (FISRWG 1998). When this course of action is taken, the processes necessary to sustain the new plant materials must also occur to prevent restoration failure. Gaining in popularity is the use of wetlands to act as treatment systems for nutrients (Mitsch 1995). In such systems, vegetation must often be harvested in order to remove detritus that would otherwise serve as energy and nutrient sources for downstream organisms. Lastly, vegetation is commonly manipulated to provide forage and habitat for wildlife species (Morrison 2002).

Wildlife populations are sometimes manipulated with the goal of effecting top-down control on ecosystems. Such manipulations can include the re-introduction of predators to control herbivores. For example, the re-introduction of wolves in Yellowstone National Park has affected herbivore populations and behaviors, which in turn have restored typical native plant communities to stream sides (Fortin et al. 2005). Similarly, the introduction of piscivorous fish has been attempted in some lakes to improve water clarity; in these cases, the expected result was the control of planktivorous fish populations, which would allow zooplankton populations to increase, and control of algal populations by grazing (Gulati and van Donk 2002). The introduction of aquatic wildlife species must be considered with great hesitation in aquatic ecosystems. Unlike plant species which may be easily transported over long distances, aquatic animals can be restricted to particular basins, and their introductions into basins they do not normally occupy can have serious unintended effects on trophic interactions.

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CHAPTER 3

ECOLOGICAL RESTORATION OF THE LOWER TRUCKEE RIVER, NEVADA: A CASE STUDY

Introduction

Recent restoration efforts in the lower Truckee River, Nevada, exemplify the successful use of process in ecological restoration. Projects have addressed altered processes in each of the four categories (water balance, fluvial, chemical, and biotic). As are other rivers in arid environments, the Truckee River is a vital resource to area ecosystems, and historically supported a wide diversity of species. However, the value of this freshwater resource resulted in many societal demands on the river, which is highly regulated by several dams and diversions. These demands, in turn, have resulted in considerable degradation of both the river and its supported ecosystems; the lower reach was said to have reached a state of "ecosystem collapse" (Rood et al. 2003). The restoration of some altered environmental processes has already caused marked improvement of the ecosystem, and additional planned projects that include environmental processes are likely to further effect positive results. In addition, this case study shows how the restoration of a single process (in this case the flow regime) can result in dramatic ecological improvements and the indirect restoration of other processes. Recent recognition of the role the river plays in supporting area ecosystems, as well as an increased appreciation of natural resources, has led to a number of efforts aimed at restoring the river's functions and associated riparian areas. As is common in many large restoration endeavors, projects have not been coordinated into a master plan but have developed as separate projects by different interest groups. Projects range from structural (such as in-stream manipulations) to non-structural (such as regulation and educational initiatives). Many of the projects, both completed and still being planned, attempt to address the underlying processes that are causing ecological damage. These projects have not been faultless, however, as some contain components that do not adequately address these processes and/or fully examine their potential ecological effects. Some altered processes, although acknowledged, are completely overlooked in management plans. The results of such efforts will not be self-sustaining and will require constant management. However, the reestablishment of some processes has resulted in a remarkable recovery of many aspects of the system, and much planned work is expected to promote considerable ecological improvement.

Background

The Truckee River (Figure 3.1) originates in California from Lake Tahoe at 6223 feet above mean sea level (MSL) in the Sierra Nevadas (Dawson et al. 2000). Flowing 105 mi north and east into the Great Basin, it ends at Pyramid Lake in Nevada, a terminal basin lake. Although the catchment is approximately 3,060 square miles, precipitation occurs mostly in the Sierras. The bulk of water enters the river as snow melt, which then flows in the losing river through mountain canyons and valleys in the Sierra Nevada rainshadow. The river continues through Truckee Meadows, a historic floodplain that is now developed by the cities Reno and Sparks. The lower Truckee River is a low- to moderate-gradient stream that flows through desert canyons and irrigated agricultural areas. A large dam (Derby Dam) downstream of Sparks

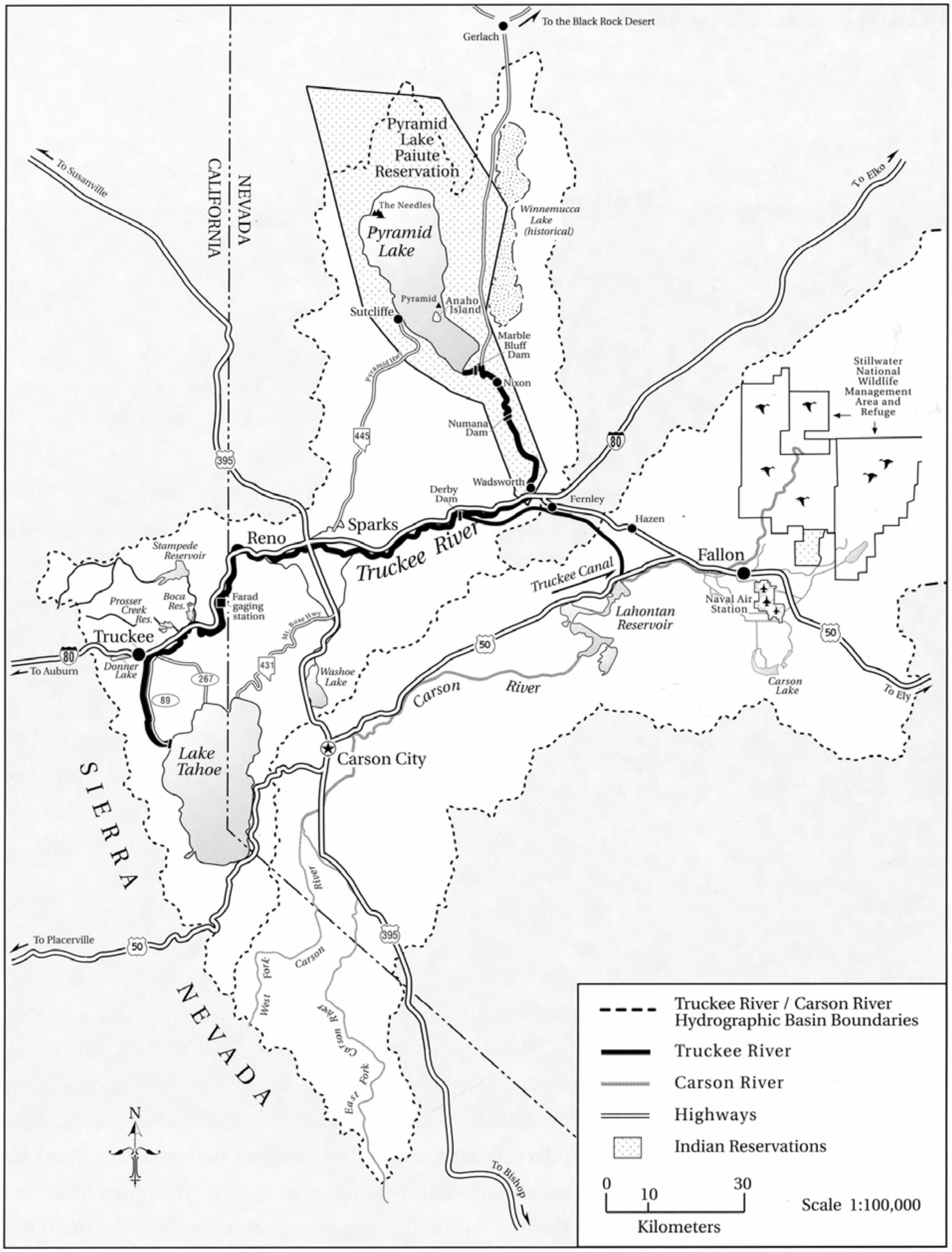

Figure 3.1: The Truckee River hydrographic system (Dawson et al. 2000)

diverts water to the Carson River. While water flowing from Lake Tahoe was historically highly oligotrophic, nutrients and dissolved solids concentrated slightly as water evaporated enroute. Evapotranspiration in Pyramid Lake is estimated to be about 40,000 acre-ft per year; this continual concentration in the lake creates alkaline, slightly saline waters (Horton 1997).

In the mid 1800s, the Truckee River was bounded by lush floodplain forests of Fremont cottonwood (*Populus fremontii*) and sandbar willow (*Salix exigua*), with wetlands formed by oxbow lakes. Truckee Meadows (in which Reno and Sparks later developed) was a green valley surrounded by arid desert, range, and narrow rocky canyons. Marshes and lowlands bordering the river covered the eastern third of the meadows, forming low, boggy fields with grasses, sedges, cattails, willows and cottonwoods. This area naturally flooded in the spring, facilitated in part by the obstruction of a large bedrock formation called the Vista Reefs (Horton 1997). The riparian system supported a large and diverse avian population and was a migratory bird corridor (Klebenow and Oakleaf 1984). The Truckee River and Pyramid Lake are the only habitats of the federally endangered cui-ui lakesucker (*Chasmistes cujus*) (U.S. Fish and Wildlife Service 1992). They also support the federally threatened Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*) (U.S. Fish and Wildlife Service 1994).

<u>Recent Environmental Concerns</u>.

Many recent environmental concerns stem from the historic use and alteration of the Truckee River and its watershed. A notable concern is the protection and recovery of the endemic cui-ui (Figure 3.2). with an adult population estimated to be from 90,000 to 200,000 adults between 1982 and 1986 and 300,000 adults in 1991 (U.S. Fish and Wildlife Service 1992). Maturing at 23-28 inches and up to 8 lbs, the cui-ui was once the mainstay of the Pyramid Lake Paiute Indian Tribe. The fish lives in Pyramid Lake most of the year and migrates into the lower

Truckee River in the spring to spawn in fresh water. While spawning adults have been noted to migrate up to 45 mi upstream (Scoppettone et al. 1983), migration is currently restricted to the lower 10 mi of the river by barriers (U.S. Fish and Wildlife Service 1992). Eggs hatch after one to two weeks and larval cui-ui generally return immediately to the more alkaline Pyramid Lake, although some remain in river backwaters for several weeks. Threats to this species have been the reduced and altered flows of the Truckee River (which do not stimulate spawning), dams preventing migration, and degraded water quality (low DO and high nutrient concentrations). Altered flows and migration barriers prevented spawning in most years since 1920 (Scoppettone et al. 1983); the long life-span of the cui-ui (up to 40 years) and hatchery operations have been credited for the species' survival during this period (U.S. Fish and Wildlife Service 1992).

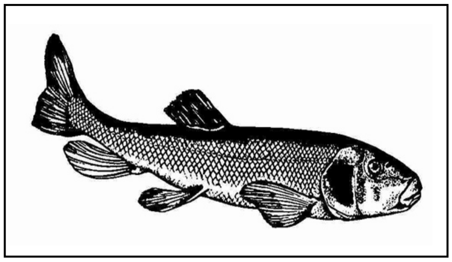

Figure 3.2: Cui-ui lakesucker
(Pyramid Lake Fisheries 2005)

The Lahontan cutthroat trout lives in the Truckee River basin and other regional river basins including those of the Carson, Walker, and Humboldt Rivers (Figure 3.3). These river basins once made up the basin of ancient Lake Lahontan; the drying of which separated the Lahontan cutthroat trout into three distinct populations (U.S. Fish and Wildlife Service 1994). The lower Truckee River was once the spawning and nursery habitat for the Pyramid Lake strain; however, this strain has been extinct since 1940. Other strains of Lahontan cutthroat trout have been re-introduced into Pyramid Lake in the 1950s (Hoffman and Scoppettone 1988) and have been

Figure 3.3: Lahontan cutthroat trout
(U.S. Fish and Wildlife Service 2005)

maintained using hatchery stock (U.S. Fish and Wildlife Service 1994). These two- to four-foot lake fish were historically a main part of the Paiute Indian fishery, especially during spawning runs which occurred in both spring and fall (U.S. Fish and Wildlife Service 1994). The historic persistence of this species in the Truckee River system has been attributed to connectivity among metapopulations in the river and lake (TRBRIT 2003). So abundant was this fish, it was noted in 1869 as being "cheaper than beef in the market at Reno and Wadsworth" (Horton 1997). Threats to the Lahontan cutthroat trout population have been overharvesting, dams preventing migration, water quality degradation (low DO and high nutrient concentrations), competition and hybridization with exotic trout species (especially kokanee salmon [*Onocorhynchus nerka*], brook trout [*Salvelinus fontinalis*], and brown trout [*Salmo trutta*]), and poor spawning and rearing habitat in the lower Truckee River (TRBRIT 2003, U.S. Fish and Wildlife Service 1994).

Riparian forest is a relatively rare habitat in Nevada. The Truckee River is an exceptionally important area for the support of Fremont cottonwoods, the primary riparian tree species in this region (Klebenow and Oakleaf 1984). While the lower reach once supported vast areas of rich riparian forest, it has experienced serious declines, leaving only thin bands of forest (Klebenow and Oakleaf 1984, Rood et al. 2003). These small bands consist mainly of willows with cottonwoods occurring only as mature trees in occasional clumps (Lynn et al. 1998). The decline in riparian forests has been attributed to impacts caused by flow regulation and extensive channelization. Such impacts include disconnection from the floodplain and wetlands, clearing of banks, and a lowered water table. Other human activities blamed include logging, clearing for agriculture and pasture, and severe overgrazing (Klebenow and Oakleaf 1984, Caicco 1998, Rood et al. 2003). Due to these disturbances, only 40% of the corridor from Vista to Wadsworth, NV, is in its natural state (Figure 3.4) (Caicco 1998). Two major factors in the

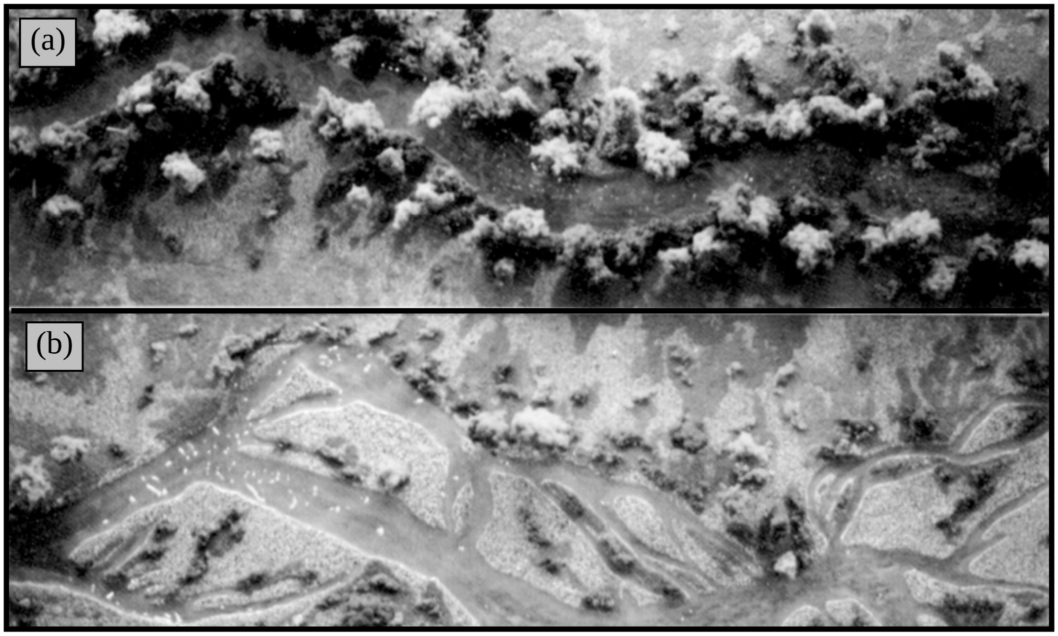

Figure 3.4: Models of the Truckee River in (a) estimated pre-development conditions and (b) degraded conditions in the 1980s (Rood et al. 2003)

decline of riparian forests are the severely reduced regeneration of floodplain cottonwoods and willows, with recruitment failure through most of the 1900s (Rood et al. 2003), and the increasing dominance of exotic species. Exotic grasses and herbs represent 33% to 72% of the total understory cover (Caicco 1998). Although planting of cottonwoods and willows has been attempted, it has been recognized that such planting is insufficient for recovery of the riparian ecosystem (Rood et al. 2003).

Concomitant with riparian forest decline was a decrease in the abundance and diversity of birds. While 91 species were observed along the lower Truckee River in 1868, bird counts conducted between 1972 and 1976 showed dramatic reductions (Klebenow and Oakleaf 1984). Forty-two species had been extirpated from the area, and 26 species have declined in abundance. A 1993 study determined that the highest bird species richness near the river occurred in riparian scrub and that both riparian scrub and Fremont cottonwood habitats were used with greater frequency than predicted by their availability (Lynn et al. 1998).

Water quality in the Truckee River has been of recent concern, and the river does not support its designated uses under the Clean Water Act (USEPA 1994). While the nitrogen-limited river once had pristine, cold-water flows, it is now is plagued by increased concentrations of nitrogen and phosphorus, elevated temperatures, and lowered dissolved oxygen (DO), as well as some industrial contamination including mercury (Stamenkovic et al. 2004). Water diversions lowered the surface of Pyramid Lake and caused elevated concentrations of total dissolved solids (TDS), which increased from 3,500 mg/l in 1882 to 5,100 mg/l in the 1990s (Horton 1997). Such water quality degradation led to increased algae growth in the river, algae blooms in Pyramid Lake, and reduced spawning and egg survival of cui-ui and Lahontan cutthroat trout (Hoffman and Scoppettone 1988, U.S. Fish and Wildlife Service 1992, USEPA 1994, TRBRIT 2003). Concentrations of nitrogen, phosphorus, and TDS at Vista, NV, increased dramatically between 1987 and 1993 and are much higher than they are 13 miles upstream at Farad, CA (Warwick et al. 1997). These water quality changes have been attributed to wastewater effluent, agriculture, urban runoff, groundwater discharge, and gold and silver extraction operations (USEPA 1994, Stamenkovic et al. 2004), as well as low dilution caused by reduced river flows (U.S. Fish and Wildlife Service 1992).

Causes of Environmental Degradation and Associated Impacts

Dams, water rights, and river blockages

Perhaps the largest issue affecting the Truckee River ecosystem is water use. As a principal source of water in the region, the river has been repeatedly dammed and diverted for agriculture, power generation, and municipal uses. Water diversion in the Truckee River basin first began in the 1850s with the discovery of nearby gold and silver deposits. Rapidly developed mining operations began diverting water from the upper Truckee River basin to cool mines, drain geothermal water, and create log flumes to transport timber for mining operations (Horton 1997). Local agricultural diversions followed in the 1860s and intensified in the 1870s. By 1882, the number of irrigation ditches and other water diversions caused a severe water shortage in the Truckee River below Reno, causing it to nearly dry (Horton 1997). Even in 1990, 59% of the water used in the basin was for irrigation (Kilroy et al. 1997).

There are currently five major dams along the Truckee River and six reservoirs in the basin; Lake Tahoe is also dammed for water supply (Table 3.1) (Kilroy et al. 1997). Many other dams have been constructed and removed as water demands changed over time. The first dam on the Truckee River was constructed in 1870 at the outlet of Lake Tahoe, to create flumes for timber transport. By the mid-1870s, few trout could travel upstream to spawn due to the number of dams and fish traps. In 1875, construction of a large dam near Verdi ended trout migration into the upper Truckee River; by 1879, four dams were in place between Wadsworth and Reno, further preventing fish migration (Horton 1997).

Table 3.1: Major dams along the Truckee River

Dam Name	**Location**	**State**	**Use**
Lake Tahoe Dam	Tahoe City, CA	CA	Power, flow regulation
Floriston Dam	Floriston, CA	CA	Power
Derby Dam	Slightly downstream of Clark	NV	Diversion
Numana Dam	14.5 mi below Wadsworth	NV	Diversion for irrigation
Marble Bluff Dam	4 miles below Dixon	NV	To reduce erosion and promote cui-ui spawning runs

At the beginning of the 20th century, rivers in the Great Basin were considered to be wasted resources if they were not put to human use. The Reclamation Act of 1902 was enacted to promote irrigated agriculture in these arid lands to 'reclaim' the desert. Championed by Nevada Representative Francis Newlands, the act (which became known as the Newlands Project) authorized the creation of the U.S. Reclamation Service. One of the first undertakings of the Newlands Project was the 1905 construction of Derby Dam (Figure 3.5) in the lower Truckee River, for the purpose of diverting water to the Carson River for farmland irrigation, as well as the creation of upstream storage reservoirs (Rowley 2002). From Derby Dam, the Truckee Canal travels 32.5 miles to the Carson River and has a capacity of 900 cfs or 1,785 acre-feet per day (Horton 1997). Capacity has since increased and about 258,300 acre-ft of water was diverted at Derby Dam in 1990 to irrigate about 68,000 acres of land (Covay et al. 1996).

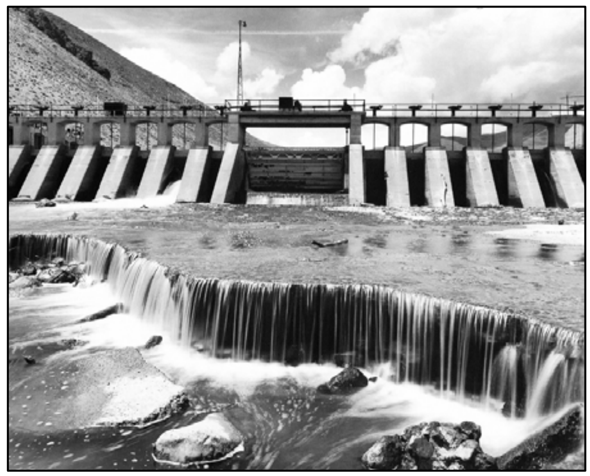

Figure 3.5: Derby Dam (Dawson et al. 2000)

Rapid growth of the Reno/Sparks area has placed an additional demand on water for municipal uses. For example 36% of the water used in the Truckee River Basin was for public supply in 1990 (Kilroy et al. 1997). In addition to that required for household use, large quantities of water are used for aesthetic and recreational purposes. Most homes and new developments sport green, irrigated lawns and shade trees. Private and public pools, as well as man-made ponds, are common. The many casinos in Reno put an additional demand on the water system by their egregious water use for entertaining tourists (Dawson et al. 2000). The conservation of water has not been historically encouraged; rather, beginning in 1919, many laws were passed banning the use of water meters. This was done, in part, to encourage growth in

Truckee Meadows (Horton 1997). Instead, residents pay a flat $49 per month fee regardless of usage (Dawson et al. 2000).

The use and diversion of water intensifies shortages caused by drought (Figure 3.6). Such shortages were already present in the late 1800s, before the construction of Derby Dam, when the river nearly dried. In 1912 (after construction of the Newlands Project), the entire flow of the Truckee River was diverted at Derby Dam to the Carson River. As a result, flow ceased in the river below the dam and the channel was reportedly "clogged with dead and dying trout". More recently (in 1992 and 1994), a nearly four mile segment of the Truckee River dried completely between the Glendale Water Treatment Plant (a diversion) and the Reno-Sparks Sewage Treatment Plant (a discharge). Flow shortages have also been seen where the flow ceased below the Lake Tahoe Dam in the Upper Truckee River (Horton 1997).

Figure 3.6: Dry channel at Derby Dam (Dawson et al. 2000)

The cumulative impact of the many diversions and dams affect fish populations, water quality, and the water balance of the basin. Dams prevent fish, especially cui-ui and Lahontan cutthroat trout, from migrating from Pyramid Lake into the river to spawn. Similarly, the migration necessary between available habitats for the persistence of Lahontan cutthroat trout metapopulations was obstructed (U.S. Fish and Wildlife Service 1992, 1994). Although fish passages have been required at some dams, they have not been successful in supporting fish movements. For example, the Numana Dam fish passage is not conducive to cui-ui migration

(U.S. Fish and Wildlife Service 1992). The many dams and diversion on the river have also resulted in a highly regulated hydrograph with little variation. Normally, river flows would be the highest during the spring following rain and snow melt in the Sierra Nevadas (Warwick et al. 1997). This flow regime would prompt the migration and spawning of cui-ui, and support riparian habitat by submerging floodplains. The loss of the natural flow regime removed the stimulus for cui-ui spawning (U.S. Fish and Wildlife Service 1992) and reduced floodplain inundation, preventing the germination and regeneration of cottonwoods (Rood et al. 2003).

In addition to affecting water levels in the river, diversions have affected levels in Pyramid Lake, where the water level and quality are the direct result of the balance between evapotranspiration rates and Truckee River flow rates (Figure 3.7). From 1929 to 1967, less than half of the Truckee River water entering Nevada flowed to Pyramid Lake. As a result, an estimated water deficit of 135,000 acre-ft per year occurred in the lake (Horton 1997). Cumulative deficits caused a 83-ft decline in water level (from 3,870 ft MSL in 1910 to 3,783.9 ft MSL in 1967). The lowered lake levels exposed the outflow delta of the Truckee River, preventing fish migration into the river. As lake levels continued to drop, differences in elevation between the lake and the Truckee River caused the channel in the delta to incise. Lowered lake levels also threatened nesting white pelicans on Anaho Island in Pyramid Lake (now a National Wildlife Refuge) by nearly exposing a land bridge to the mainland. In addition, the evaporation

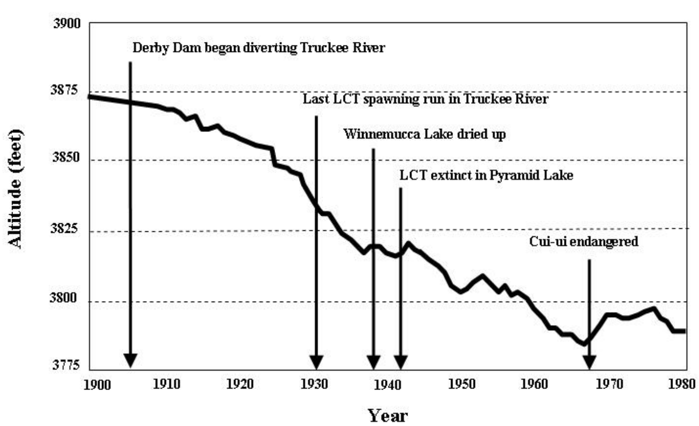

Figure 3.7: Pyramid Lake surface elevations, 1900-1980. Adapted from Pyramid Lake Paiute Tribe (2005)

of lake water without adequate inflows concentrated dissolved solids, which increased to near 519,000 ppm TDS. TDS loading of the river and lake has also become a concern, as irrigation activities in Washoe County have increased the discharge of highly mineralized groundwater into the river (Doherty 2002).

Lastly, decreased water flows in the Truckee River also affected adjacent wildlife habitats. Lake Winnemucca (also known as Mud Lake) was a large depression east of Pyramid Lake, with water depths varying from inundated to marshy to dry. The area, about half the size of Pyramid Lake, supported a wide diversity of plant and animal species and was important habitat for nesting and feeding white pelicans and other migratory waterfowl. Lake Winnemucca was designated a National Wildlife Refuge in 1936; however this designation was abandoned in 1938 when the lake completely dried due to insufficient water flow. It is estimated that in the absence of diversions, Truckee River discharge would have connected Pyramid Lake to Lake Winnemucca up until 1930. (Horton 1997).

Channel and bank alterations

The cities of Reno and Sparks developed in the Truckee Meadows, an area above the confluence of Steamboat Creek and the Truckee River. The verdant landscape at Truckee Meadows likely attracted the early pioneers who settled there (Rowley 1984). However, in a region where rainfall is less than 7 inches per year, flooding of the Truckee Meadows is not uncommon. The first extensive flood after the development of Reno occurred in 1890. Major flooding subsequently occurred in the Truckee Meadows (including Reno and Sparks) in 1907, 1937, 1950, and 1955 (Horton 1997).

As a response to these floods, the U.S. Army Corps of Engineers (USACE) completed flood control projects in the mid 1960s. These projects included enlargement and channelization of the river channel from Truckee Meadows to 7.5 miles south of Reno, as well as other intermittent channel alterations, to allow for increased water flow (Horton 1997). Selected reaches of the river were straightened between Wadsworth and Dead Ox Wash, resulting in increased channel gradient, decreased sinuosity, and isolation of the river from its former floodplains (Warwick et al. 1997). In addition, banks were cleared, cottonwood trees were harvested, oxbows were largely eliminated, and several small islands in the Truckee River were obliterated (Horton 1997, Warwick et al. 1997). Within 6 months of these changes, the riparian zone of the lower Truckee River was severely damaged (Horton 1997). The loss of connectivity between the river and its floodplain resulted in the loss of many remaining riparian forests and wetlands and contributed to the failure of cottonwood and willow regeneration. As such, only highly fragmented patches of older forest remained (Warwick et al. 1997, Rood et al. 2003).

As part of the USACE flood control projects, modifications were made to Steamboat Creek, the largest tributary flowing into the Truckee River. The Vista Reefs, a bedrock formation just downstream of the confluence of Steamboat Creek and the Truckee River, were removed in 1963. This action resulted in the draining of considerable amounts of wetlands, as well as massive amounts of upstream erosion in the creek (Horton 1997). The subsequent lateral instability of the banks caused the channel to migrate, especially downstream of Wadsworth and upstream of Vista Reefs. Currently, many areas of Steamboat Creek are stabilized by riprap (Warwick et al. 1997).

The transportation and deposition of eroded sediment (caused by the removal of the Vista Reefs and other projects) into Pyramid Lake prograded the delta, which has extended the length of the river by 4 km since 1900 (Warwick et al. 1997). The enlarged delta completely blocked the migration of cui-ui and Lahontan cutthroat trout (Warwick et al. 1997). The channelization projects also altered the characteristics of both the river substrates and water flow (Warwick et al. 1997), likely affecting cui-ui egg survival by impeding the subsurface flow of water (U.S. Fish and Wildlife Service 1992). As such, the spawning habitats of cui-ui and Lahontan cutthroat trout, as well as the habitats of many benthic organisms, were severely damaged. Additionally, channel alterations caused a lowering of the water table, resulting in the encroachment of more drought-tolerant upland plants (Warwick et al. 1997)

The loss of riparian and wetland habitats caused by flood control activities greatly affected the river's water quality, as elimination of these areas removed their important ecosystem buffering actions. As a result, nutrients and sediment in runoff can directly enter the river, and water temperatures have increased due to the loss of shade. The increased nutrients and sunlight stimulated the growth of algae, thereby increasing decomposing biomass and lowering DO levels (Kilroy et al. 1997). The loss of floodplain forest also removed vital habitat for riparian wildlife, such as piscivorous birds and waterfowl (Lynn et al. 1998).

Ironically, these and other upstream flood control projects have not prevented floods in Reno and Sparks. Major flooding occurred again in 1963, resulting in catastrophic erosion and scouring of the newly altered downstream channel. In 1997, a massive flood in the entire Truckee River basin occurred (Figure 3.8), and six counties (including Washoe) were declared federal disaster areas (Horton 1997).

Figure 3.8: Flooding in downtown Reno, 1997 (Dawson et al. 2000)

Land use and management

The use and management of land has been the greatest contributor to water quality degradation in the Truckee River Basin. Early uses consisted of mining, logging, and agriculture, while urban areas grew in the 1900s. These activities introduced nutrients, solids, and chemical contaminants into the river and its tributaries.

<u>Logging</u>

Extensive logging started in the 1850s in the upper Truckee River basin near Lake Tahoe to supply timber for the gold and silver rush (Horton 1997, Otis Bay Ecological Consultants 2004). Completion of the Central Pacific Railroad in the 1860s enabled further development of the timber industry (Horton 1997). Logging, paper, and pulp mills prospered, resulting in intense deforestation of the area in the 1880s. By 1896, 60% of all mature trees in the Lake Tahoe Basin were harvested (Otis Bay Ecological Consultants 2004). Massive quantities of sawdust, discharged from the logging and sawmill operations, thickly coated the banks and bed of the river and formed bars where the river flowed into Pyramid Lake (Horton 1997, Otis Bay Ecological Consultants 2004). These sawdust bars blocked fish migration, preventing spawning, and the slurry of sawdust in the river resulted in fish kills. Sawdust discharge continued until the 1890s (Horton 1997).

In the 1850s, prospectors and mine workers, attracted by gold and silver deposits in the Comstock Lode, settled near Truckee Meadows in what was later called "the Rush to Washoe". Reno began to develop in the 1860s with the construction of the Central Pacific Railroad, becoming a center for commerce and transportation (Rowley 1984). Agricultural development of the region, encouraged by the Newlands Project, dramatically changed the landscape. Vast areas were cleared, plowed, and irrigated or converted into cattle range (Rowley 1984, Horton

1997, Warwick et al. 1997). This clearing included sloughs, moist meadows, and much of the floodplain, which was particularly used for extensive cattle grazing (Klebenow and Oakleaf 1984, Warwick et al. 1997). As recently as 1990, cattle range occupied 53% of the Truckee River watershed in 1990, while forest accounted for 27%. Irrigated agriculture has been reduced to 2% in recent times (Kilroy et al. 1997). Major crops are alfalfa, cantaloupe, onions, and garlic (USEPA 1994).

The clearing of floodplain forests allowed the direct runoff of fertilizer and cattle waste into the river. These types of activities adjacent to the river and its tributaries continue to affect water quality today. Drainage from agricultural lands and irrigation return flows have contributed to increased nitrogen and phosphorus concentrations in the river (USEPA 1994, Warwick et al. 1997), and the removal of understory vegetation from severe overgrazing has eliminated the dense shrub understory and prevented cottonwood regeneration (Klebenow and Oakleaf 1984).

Urbanization

Rapid urban development of Reno began in the 1950s (Covay et al. 1996) as it grew in popularity as a city for entertainment, gambling, drinking, prostitution, quick marriage and easy divorce (Rowley 1984). As urban development continued, additional floodplain areas were cleared (Warwick et al. 1997). Although urban areas comprised only 3% of the Truckee River watershed in 1990, the development is concentrated in the Reno/Sparks area. About 290,000 people lived in the basin in 1990, with 200,000 concentrated in the Reno/Sparks urban area (Kilroy et al. 1997).

The urbanization of Reno and Sparks has taken a toll on water quality. Currently, agricultural areas in Truckee Meadows are being developed into urban and suburban areas at a rapid pace (Kilroy et al. 1997, Dawson et al. 2000). This construction exposes soils to erosion, and new landscape activities contribute nutrients (through fertilizers) and pesticides. Urban runoff near Reno and Sparks greatly increases nitrate concentrations and is a likely partial cause of increased orthophosphate concentrations (Kilroy et al. 1997).

Sewage disposal from developing areas in the Truckee River watershed has consistently been a major cause of water quality degradation. Sewer lines, first constructed in Reno in 1860, dumped raw sewage into the river (or, during late summer drought, onto dry lake beds). Subsequently, algae, which was never before observed, was noted to be covering boulders in the river (Horton 1997). Presently, two large wastewater treatment plants serve as the main wastewater sources affecting the Truckee River. The Truckee Meadows Water Reclamation Facility (TMWRF), constructed in 1966 (as the Reno-Sparks Sewage Treatment Plant), has a current capacity of 40 mgd for sewage and 60 mgd for stormwater treatment (Horton 1997). This discharge to Steamboat Creek is responsible for most of the increased nitrogen and phosphorus concentrations in the lower Truckee River (USEPA 1994, Kilroy et al. 1997, Warwick et al. 1997). The treatment plant can have an especially large impact on downstream water quality in summer, when it makes up a higher percentage of river flow (Kilroy et al. 1997). This percentage can sometimes be extreme, as it was in 1992 and 1994, when the entire flow downstream of the TMWRF consisted of wastewater effluent (Horton 1997).

The Tahoe-Truckee Sanitation Agency, formed in 1972, constructed a sewage treatment plant in the upper Truckee River Basin (Sacramento Regional Research Institute 2004). Although treated effluent from this facility is applied to land, nitrogen-enriched groundwater drains into a tributary of the upper Truckee River and contributes to river nitrogen concentrations (Kilroy et al. 1997). Other waste sources include accidental sewage spills and leaks over time

(Horton 1997, Kilroy et al. 1997). All told, the Truckee River Basin received approximately 43,000 acre-ft of treated sewage in 1990 (Kilroy et al. 1997). As a result of nutrient loading from municipal and agricultural sources along the Truckee River, algae blooms have occurred in Pyramid Lake. The reduced volume and increased salinity of the lake exacerbates this problem, as inflowing fresh water floats over the denser alkaline lake water, and nutrients are retained in the photic zone (Horton 1997).

Although not severe, some pollution (acids and mercury) of the Truckee River has occurred as a result of industrial discharges. The Floriston Pulp and Paper Company in California began discharging up to 150,000 gallons daily of acidic waste in 1899. This discharge severely degraded water quality and killed thousands of fish until it was discontinued in the 1930s. (Horton 1997). Mercury waste from the processing of gold and silver ore in the 1800s led to contamination in the sediments of Washoe Lake, Steamboat Creek, and nearby in the Truckee River. It has been determined that Washoe Lake continues to be a significant source of mercury (Stamenkovic et al. 2004). Like many developed areas, the more recent history of the Truckee River and its watershed has been checkered with gas, chemical, and sewage spills (Horton 1997). However, elevated nitrogen and phosphorus levels are the main water quality issues in the river.

Wildlife and vegetation

Occasionally, ecological problems along the Truckee River have been brought about or aggravated by the manipulation of wildlife, both animals and vegetation. Most of such alterations can be categorized as either a wildlife addition (as an exotic species) or wildlife removal (as population reduction or extirpation).

Exotic species

Exotic fish species (not native to the Truckee River Basin) were introduced as early as 1875 in response to declining fish stocks. Introduced species included brook trout, rainbow trout (*Oncorhynchus mykiss*), brown trout, mackinaw (lake) trout (*Salvelinus namaycush*), kokanee trout, whitefish (*Coregonus* spp.), and catfish (*Ameiurus catus*). By the 1890s, annual restocking was necessary to maintain the Lahontan cutthroat trout fishery, as was the establishment of a closed season (Horton 1997, TRBRIT 2003). These introduced species, especially trouts, compete for resources and hybridize with Lahontan cutthroat trout, contributing to the decline of the native population (U.S. Fish and Wildlife Service 1994).

Exotic weeds were identified as a major, growing problem along the Truckee River in 1996 (Horton 1997). One such species, tall whitetop (*Lepidium latifolium*) is both allelopathic and strongly competitive, and has displaced other plant species as it has colonized over 12,000 acres of the lower Truckee River basin (Donaldson 1997). Growing near water, this species is difficult to control since the season for applying herbicide coincides with that of spawning cui-ui (Horton 1997). In addition, tall whitetop growing along streambanks leads to bank instability during flood flows, as its root system is easily broken apart. Unfortunately, new plants can also grow from each broken root segment. Both extensive soil erosion and range expansion from root fragments occurred during the 1997 flood in Truckee Meadows (Donaldson 1997). Tamarisk (*Tamarix* spp.) is a problematic exotic shrub in the Truckee River floodplain. Although it is a heavy user of water, its long taproot is able to access groundwater from depths of 3 m or more. As a result, the water table can be lowered in areas colonized by these shrubs. Tamarisk is easily spread by seed or fragmented stems (Donaldson 1997). Other problematic exotic plants include yellow starthistle (*Centaurea solstitalis*), Canada thistle (*Circium arvense*), puncture vine

(*Tribulus terrestris*) and whitetop (*Cardaria draba*) (Horton 1997), and small stands of purple loosestrife (*Lythrum salicaria*) have recently established along the lower Truckee River (Donaldson 1997). The spread of these species is contributing to the decline of floodplain forest habitat, as native understory plants and cottonwood seedlings are outcompeted. Additionally, the lowered quality of floodplain habitat has exacerbated the problem of declining bird populations (Lynn et al. 1998).

Wildlife removal

Severe overharvesting of fish contributed to the decline of cui-ui and Lahontan cutthroat trout populations. The construction of the Central Pacific Railroad in the 1860s enabled an enormous expansion of the fishing industries. Extensive harvesting of trout, sometimes entire spawning runs, quickly decimated the population. The Truckee River, once seen as a bountiful fishery, became depauperate (Horton 1997).

The reduction in forested riparian areas (by clearing or hydrological change) resulted in the loss of an important migratory bird corridor. Entire guilds of birds have been eliminated, including almost all the marsh birds, and many of the remaining species have smaller populations (Klebenow and Oakleaf 1984, Warwick et al. 1997). Piscivorous birds were also lost, most likely due to changes in fish populations and assemblages caused by channel alterations (Warwick et al. 1997).

Passive wildlife effects

Changes in wildlife populations caused by the alteration of environmental resources created chains of effects throughout the ecosystem. For example, low DO concentrations caused by the proliferation of aquatic plants and algae (caused by high nitrogen and phosphorus concentrations), as well as elevated temperatures (caused by the removal of the tree canopy), have caused fish kills (USEPA 1994). Similarly, the blocked migration of cui-ui and Lahontan cutthroat trout reduced piscine biomass in the river, affecting both predators, such as river otters and bald eagles, and scavengers such as aquatic insects, and disrupting nutrient cycling patterns between the river and Pyramid Lake (Warwick et al. 1997).

Restoration Initiatives

Within the past 30 years, there has grown tremendous interest and effort regarding the Truckee River ecosystem and its conservation and restoration. While some of the effort has been required as the result of legal battles and settlements, much of it has come from a better informed public. An array of agencies and groups have been involved with management, planning, and restoration, including federal, tribal, state, regional, and local governments, research institutions, non-profit agencies, and groups representing citizens, businesses, and utility companies (ECO:LOGIC Consulting Engineers 2004). Their efforts have largely focused on the issues of water rights and supply, restoration of fisheries and floodplains, channel reconfiguration, land use, and exotic species management.

Water rights adjudications, settlements, and supply initiatives

The Truckee River is one of the most litigated rivers in the nation (Cobourn 1999). Principal parties have included the U.S. Department of the Interior (as the U.S. Bureau of Reclamation, the Bureau of Indian Affairs, and the U.S Fish and Wildlife Service), the Truckee-Carson Irrigation District, the Pyramid Lake Paiute Tribe, water and electric companies in the Truckee Meadows, the cities of Reno, Sparks, and others, the U.S. Environmental Protection Agency, and the states of Nevada and California (Horton 1997). The subject of litigation has primarily been water rights, but water quality has also been of issue. Such issues have taken shape as disagreement over the control of dams, minimum flow rates, and dates of first appropriation. Water rights in Nevada are based on a ‘prior appropriation doctrine’, in which the first entity to use water for beneficial use has a higher priority than subsequent users (Horton 1997). The ecological system has no intrinsic right to the water supply.

Of critical importance in restoration of the Truckee River have been several lawsuits brought by the Pyramid Lake Paiute Tribe in an effort to restore water flows to Pyramid Lake. The tribe was historically supported by fishing, especially for cui-ui and Lahontan cutthroat trout, which were especially plentiful during spawning runs. However, the decline of fish populations caused by upstream water diversions forced the tribe to become agrarian. A U.S. District Court decision in 1973 (*Pyramid Lake Paiute Tribe of Indians v. Morton)* recognized that the Paiute Indians had prior water use rights for the survival of their people and fishing-based culture. These new limits required that all water at Derby Dam in excess of valid Newlands Project water rights be diverted to Pyramid Lake. Annual diversions at Derby Dam were expected to be reduced by 79,000 acre-feet per year (Horton 1997, United States Department of the Interior 2004). However, the Truckee-Carson Irrigation District, which operates Derby Dam, refused to comply for some time (Horton 1997, Seney 2002).

A major water supply turning point occurred in 1982 after a court decision (*Carson-Truckee Water Conservation District v. Secretary of the Interior*) ruled that that the waters of Stampede Reservoir were to be used solely for the restoration of Pyramid Lake and its fishery (Horton 1997). For the first time, water storage was set aside specifically for the management of Pyramid Lake and its ecosystem. This decision was bolstered in 1989, when the Nevada Legislature determined that water use for wildlife, wetlands, fisheries and wildlife habitat constituted a ‘beneficial use’ that could be granted water rights (Horton 1997, Seney 2002).

Enacted in 1990, the Truckee-Carson-Pyramid Lake Water Settlement Act (Public Law [P.L.] 101-618) included provisions to promote the enhancement and recovery of endangered and threatened fish species in Pyramid Lake and to prevent further degradation of Lahontan Valley wetlands (Horton 1997, Seney 2002). The act requires that a new Truckee River Operating Agreement (TROA) be developed (United States Department of the Interior 2004) and stipulates that water rights be purchased to support the restoration efforts. Approximately 225,000 acre-ft are expected to be purchased from willing rights holders of the Newlands Project (Seney 2002). A program to purchase an additional 24,000 acre-ft of water rights below Vista was established by the 1996 Truckee River Water Quality Agreement (between Reno, Sparks, Washoe County, the Pyramid Lake Paiute Indian Tribe, and the U.S. Department of the Interior). The decrease in diversions would permit sufficient dilution of increased wastewater discharge from an expanded TMWRF. Consequently, both instream flow in the lower Truckee River and supply to Pyramid Lake would increase (Horton 1997).

To maximize the available water for users with water rights, court decisions and agreements often included water conservation elements. For example, one goal of the Operating Criteria and Procedures (OCAP), established in 1969 for the Newlands Project, was to minimize diversions from the Truckee River at Derby Dam. These regulations also required water conservation in agriculture and the discontinuation of diverting Truckee River water for power generation in the Carson River (Horton 1997, United States Department of the Interior 2004).

Municipal supply currently constitutes much of the water demand. Water conservation is being encouraged through both regulation and public education. As a result of such initiatives, TMWA customers have reduced their consumption by 25% (Regional Water Planning Commission 2005). A water meter retrofit program in Truckee Meadows started in 1995 will require metered water billing; however, it was estimated that the program would take 12 years to complete (Horton 1997). In the interim, strict ordinances in Reno and Washoe County have been adopted to eliminate the waste of water by regulating and limiting outdoor uses. For example, the pooling or running of water into the street is prohibited, and leaks found in pipes or hoses must be fixed within 24 hours. More stringent rules are applied during water emergencies. Penalties for violation of these conservation ordinances can be severe; ranging from $25 for a first offense to $500 for a third offense, depending on the type of violation (Washoe County 2004, City of Reno 2005).

Household water conservation is promoted by the use of water-conserving fixtures. All new construction and remodels in the region must use efficient plumbing fixtures, and efficient plumbing design is encouraged. To help existing consumers reduce their water consumption, retrofit programs have been established to provide free low-flow toilets and faucets (Regional Water Planning Commission 2005). Nearly 10,000 toilets have been installed under the program, saving 86 million gallons per year (Voyles 2004).

Public education about water conservation is provided by a number of groups. The Truckee Meadows Water Authority has developed excellent education tools aimed at encouraging water conservation, offering guidance on indoor and outdoor water use (using specific, easy to understand examples), water efficient landscaping, instructions for finding and repairing leaks, and school lesson plans (Truckee Meadows Water Authority 2005).

Increased flows have improved spawning run conditions at the Truckee River delta (U.S. Fish and Wildlife Service 1995). In addition, conditions of downstream wetlands have improved, and protection of Anaho Island by increased water levels in Pyramid Lake has resulted in an increase in white pelican colonies (Horton 1997). These benefits are likely to increase, due to additional flow from the purchase of water rights.

Fisheries

The placement of cui-ui and Lahontan cutthroat trout on the Endangered Species List prompted the formation of plans for the conservation and recovery of these species. Such plans are also required by P.L. 101-618 (U.S. Fish and Wildlife Service 1994). The major issues affecting cui-ui and Lahontan cutthroat trout populations are recognized as declining population, migration barriers, degraded or eliminated riparian habitat, and water quality.

To support the populations of cui-ui, hatchery larvae and juveniles have been stocked into Pyramid Lake since 1972, and harvesting is prohibited (U.S. Fish and Wildlife Service 1992). Small self-sustaining populations of Lahontan cutthroat trout are located in the upper Truckee River and upper tributaries; however, the Pyramid Lake population is supported solely by hatcheries (United States Department of the Interior 2004). An innovative project of in-stream incubators in the lower Truckee River is attempting to imprint fry on the river location so they will return there to spawn (United States Department of the Interior 1998). Lastly, fishing regulations limit the number of Lahontan cutthroat trout that can be harvested (U.S. Fish and Wildlife Service 1994).

The main migration barrier for cui-ui and Lahontan cutthroat trout is the delta formed at the mouth of the Truckee River. To prevent further erosion and sedimentation at the river's mouth, the Marble Bluff Dam was constructed in 1975. The Pyramid Lake Fishway was constructed simultaneously to direct migrating fish around the dam when river access is blocked by the delta. Initially, the fishway was not effective; its design and operation has been modified several times to improve performance. When water levels and/or flows permit fish passage through the delta, a river trap and elevator located near the base of the dam corrals and transports the fish into the river. The river trap is the preferred method of migration, as it results in lower stress to the fish and fewer mortalities (U.S. Fish and Wildlife Service 1995).

High spring and early summer flows (minimum 1,000 cfs) are required to initiate cui-ui spawning runs, and more fish migrate if flows increase above the minimum needed. Beginning in the 1980s, water has been released from Stampede Reservoir every spring to provide these flows (U.S. Fish and Wildlife Service 1992). In addition, three abnormally wet years in 1983 raised the elevation of Pyramid Lake by 23 feet, allowing cui-ui to pass through the delta (the route preferred by environmental managers) (U.S. Fish and Wildlife Service 1995). Further plans for the recovery of both species focus on increasing water flow, restoring riparian habitat, improving water quality, and improving spawning run management. Water flow is expected to increase with the help of the TROA, water conservation in Reno and Sparks, and better irrigation efficiency (U.S. Fish and Wildlife Service 1992, United States Department of the Interior 2004). Management of populations and spawning runs includes improvements to the Marble Bluff fish facilities to alleviate problems caused by insufficient capacity, construction of a fish bypass at Numana Dam, and continued use of hatcheries (U.S. Fish and Wildlife Service 1992, 1994, TRBRIT 2003). While the Lahontan cutthroat trout recovery plan includes the restoration of metapopulations, this goal is unlikely to be achieved due to the many dams along the river.

As a result of recent conservation measures, migration of cui-ui at the Marble Bluff Dam has increased from no individuals in 1976 to 112,682 in 1995 (Figure 3.9). However, very few Lahontan cutthroat trout have been recorded passing through the fishway (U.S. Fish and Wildlife Service 1995). Increased spring river flows have been successful in initiating the cui-ui spawning run (U.S. Fish and Wildlife Service 1992). In addition, increased flows to Pyramid

Lake have increased populations of tui chub (*Gila bicolor*), the primary food of Lahontan cutthroat trout. As a result, the growth rate of Lahontan cutthroat trout has greatly increased (Horton 1997).

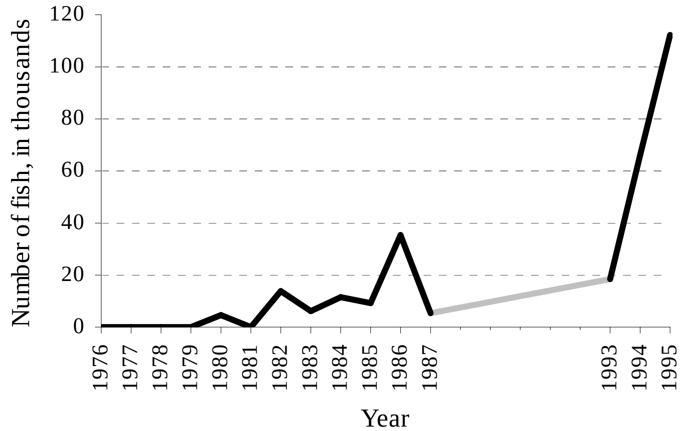

Figure 3.9: Number of cui-ui passing through the Marble Bluff fish facility. Data from U.S. Fish and Wildlife Service (1995).

Floodplain forest ecosystem

Management of the flow regime for cui-ui provided an unexpected benefit for riparian cottonwood forests. The 1987 flow regime produced collateral recruitment of Fremont cottonwoods and willows in bands lining the river (Rood et al. 2003). Analysis of that year's hydrograph revealed conditions that matched those of a cottonwood "recruitment box model" proposed by Mahoney and Rood (1998). The model, based on studies of cottonwood hydrologic requirements, predicts the required characteristics of flow regimes in early spring to summer for cottonwood germination and establishment. As cottonwood seeds are viable for only about one month, the correct regime must match the period of seed release. The peak of the spring flood must both submerge the river banks and occur slightly before seed release, so that germination can occur during the falling limb of the hydrograph. Recruitment occurs if the flood stage declines slowly, at a rate of about 1.0 inch per day or less, so that slow-growing cottonwood roots can stay in contact with the receding moisture. Seeding mortality occurs where flood stage declines more rapidly (Mahoney and Rood 1998). Seedling survival likely increases if flows are low over the next few years, preventing scouring. Additionally, recruitment is likely enhanced if flows in the year preceding germination are high, producing barren sites for recruitment (Rood et al. 2003).

In response to these observations, an instream flow regime was implemented in 1995 to intentionally recruit cottonwoods. The suggested regime consisted of a flood stage 24 to 60 in above base flow, followed by a rapid decline to expose germination sites, then a slow decline of 1.0 inch or less per day. This prescription was again followed from 1996 to 1999, and was remarkably successful. Extensive bands of cottonwood seedlings were formed, often with initial seedling densities exceeding 418 per square yard, and grew to 6.5 to 9.8 ft tall within 4 years (Figure 3.10). Willow seedlings also appeared, and quickly spread by suckering (Rood et al. 2003).

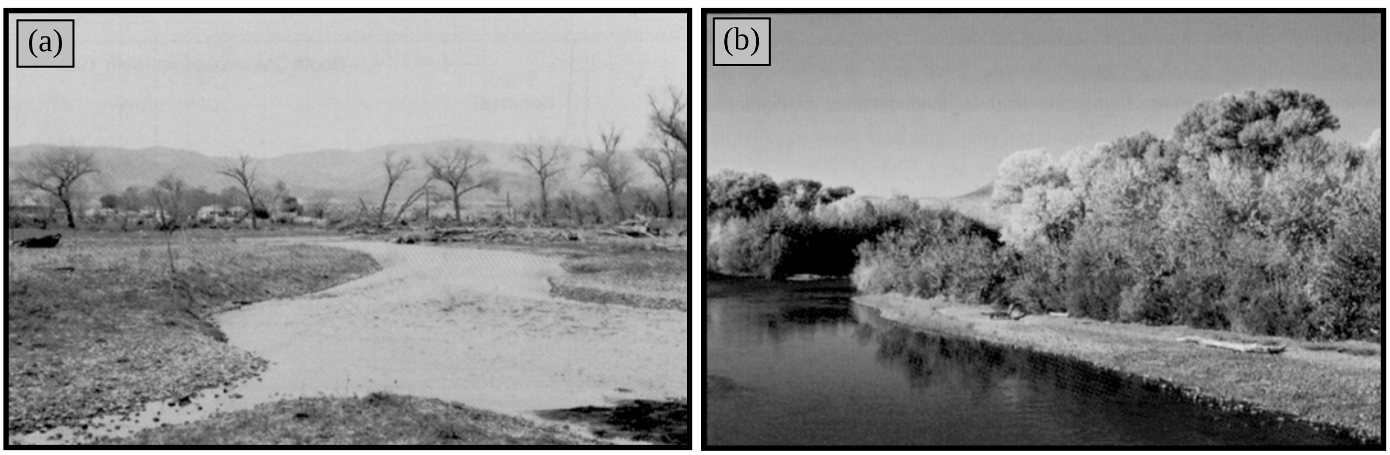

Figure 3.10: Truckee River conditions in (a) winter 1977 and (b) autumn 1997 at adjacent locations (Rood et al. 2003)

The reestablishment of cottonwoods and willows along the river cascaded into a number of additional habitat improvements. The river channel has narrowed and deepened as sediments have been deposited along the bands of saplings. Stream temperatures have lowered as water depth and shading increased, resulting in the notable observation of trout throughout the summer of 1999. A remarkable recovery of bird populations has occurred following the increase in streamside heterogeneity. Rare and extirpated species have returned, and observations have indicated that many species are breeding. While these ecological improvements are extraordinary, physical alteration of channels is required in areas that have been channelized, as restoration from only the naturalized hydrograph and forest regeneration would take decades or centuries (Rood et al. 2005).

Channel/bank restoration

As was clear from the 1997 flood in Reno and Sparks, flood control work is still needed to protect infrastructure. To do so, projects have been planned and partially completed along the river from Reno to Wadsworth. Unlike previous flood control projects, plans for future work are attempting to enhance and restore aspects of the Truckee River while protecting the cities from floods (USACE 2003). Another new aspect is that a community-based planning process was used. The Community Coalition, a group of citizens interested in the Truckee River ecosystem, has proposed alternatives to restore parts of the ecosystem while preventing flood damages. The central argument to their recommendations is that there needs to be a place for the river to flood. To accomplish this, the group has proposed the creation of benches along the channel in some areas instead of levees. If constructed, a low terrace would create a floodplain at the two-yr flood level, and a higher terrace would be created two feet above the low terrace. This plan would create a floodable area of about 172 acres, along 5.9 miles of the south bank and 1.1 miles of the north bank (Montgomery Watson Harza 2002b). The Community Coalition proposal is the most comprehensive alternative under consideration.

The USACE and the Community Coalition share a common goal of promoting a living river concept that would preserve and improve habitats, water quality, geomorphic characteristics, and restore environmental resources (Montgomery Watson Harza 2002a). Specific goals include those to increase the amount of riparian forest and wetland habitat, increase the structural diversity within these habitats, reduce accelerated rates of bank erosion, restore the hydrogeomorphic structure of the river and its connectivity to the floodplain, restore instream habitats, protect fish migration, improve water quality, and create a greenbelt and other recreational opportunities (Montgomery Watson Harza 2002a, USACE 2003).

In order to achieve these goals for both flood protection and river restoration, current project objectives are to recreate an ecologically functioning floodway, using artificial flood control measures (e.g. floodwalls and levees) only where necessary. Methods to accomplish these objectives include the restoration of geomorphic meanders and floodplain connections, removal of artificial bank stabilization material, creation of floodplain terraces, placement of necessary levees and floodwalls back from habitat and vegetation, creation of floodplain forest by planting and/or encouraging regeneration, use of wetlands to improve water quality, removal of exotic plant species, and restoration of flows by reducing diversions (USACE 2003).

Rigorous planning methods are being used to ensure hydrogeomorphic stability of the project and avoid downstream impacts. Project planners recognize that while attempting to control excess erosion, lateral migration should occur as a natural feature of the river system. Therefore, computer models are being applied to integrate hydraulics, geomorphology, and water quality to predict the effects of proposed channel modifications (Montgomery Watson Harza 2002a).

Riparian projects that have received specific attention to date are the restoration of the McCarran Ranch and Lockwood properties on the Truckee River and of lower Steamboat Creek (USACE 2003). Work sponsored by The Nature Conservancy at McCarran Ranch has already begun, involving five miles of the channelized river. Entrenchment of the channelized river had resulted in the degradation of the adjacent floodplain, creating a more upland habitat. To restore natural conditions, a narrower, meandering channel was created. About 18,000 tons of river rock were used to elevate the entrenched channel by approximately one meter and to create riffle habitat. The reshaped banks and elevated bed were designed to allow connection to an adjacent floodplain, increase infiltration, and support newly-planted floodplain vegetation. In addition, two wetland areas were created to be used to raise endangered frogs and support other aquatic life (DeLong 2003, The Nature Conservancy 2003, Voyles 2005).

Work planned for Steamboat Creek is similar to that performed and planned at McCarran Ranch. The channelized and incised lower 1.1 mile section of the creek (owned by Washoe County and the University of Nevada) is to be replaced by a new low-gradient, meandering stream channel of 2.2 miles (Figure 3.11). The new channel would be designed for overbank flooding in a 700-ft wide meander belt and a 1,770-ft wide riparian corridor. As the historic

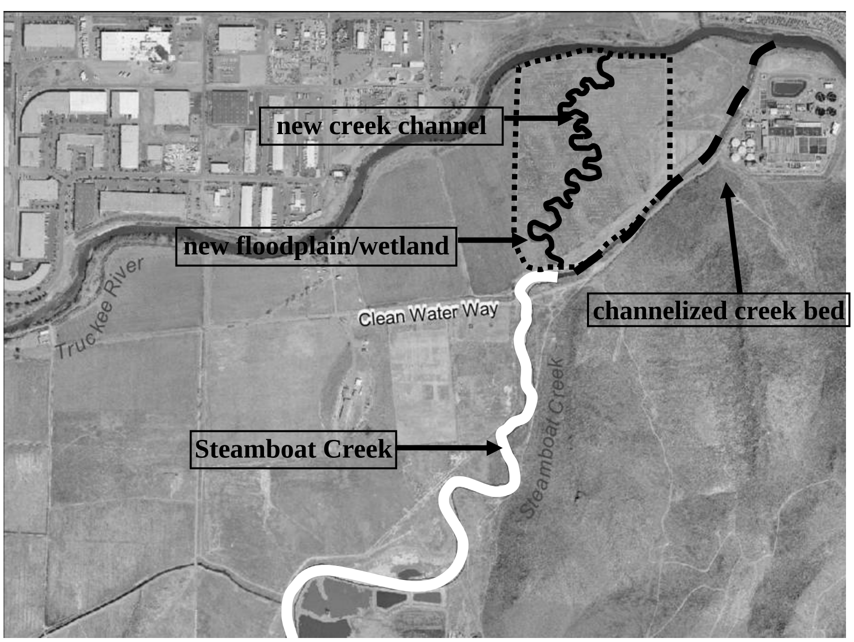

Figure 3.11: Conceptual plan for Steamboat Creek restoration. Adapted from USACE (undated).

creek floodplain included large wetland areas, the project includes the creation of approximately 80 acres of wetland in the meander belt and 200 acres of forest in the riparian corridor. As such, the project is expected to provide flood attenuation and wildlife habitat (Washoe-Storey Conservation District 2001).

Due to hydraulic changes from the removal of Vista Reefs in the Truckee River, an entirely new hydrogeomorphically appropriate system must be designed for lower Steamboat Creek. To help ensure project success both geomorphologically and ecologically, an extensive academic and peer review process has been established including faculty from the University of Nevada, Nevada Department of Environmental Protection (NDEP), U.S. Fish and Wildlife Service, USACE, the Natural Resources Conservation Service, USEPA, and other agencies (Washoe-Storey Conservation District 2001).

While plans for the Lockwood property on the Truckee River are still preliminary, Washoe County has purchased the land and removed most of the homes present to prevent flooding problems and facilitate restoration efforts (USACE 2003).

Land use and management initiatives

To combat ecological problems in the Truckee River caused by agricultural practices and urban development, strategies have been developed to reduce their impacts and improve water quality. These strategies include those that manage point sources and non-point sources (NPS), and projects to sequester nutrients already in the waterway. Water quality standards and attainment programs have been developed by both the NDEP and the Pyramid Lake Paiute Tribe. The major pollutants regulated by these programs include nitrogen, phosphorus, and TDS (NDEP 1994, Pyramid Lake Paiute Tribe 2004). Special attention has been paid to nitrogen loading, due to the nitrogen-limited status of Pyramid Lake (Pyramid Lake Paiute Tribe 2004).

Discharge permits are used to regulate point sources, and permit requirements can be modified as necessary to achieve water quality objectives (NDEP 1994, Pyramid Lake Paiute Tribe 2004). Stormwater discharge permits have been issued to Reno, Sparks, and Washoe County. These permits provide governmental authority to regulate activities that affect stormwater quality, as well as enforce compliance (NDEP 1994). Other NPS management plans and attainment programs are presented by these programs (NDEP 1994, Pyramid Lake Paiute Tribe 2004)

The largest point source affecting the lower Truckee River is the TMWRF. Major improvements have been made to this facility, including a phosphorus removal system installed in 1985 and nitrification/denitrification towers installed in 1988. While these systems have been very effective in removing nutrients, nitrogen removal efficiency has been affected by the establishment of exotic pouch snails, common in aquaria, in the towers. Consumption of the nitrogen-removing bacterial film by these snails caused nitrogen discharge limit violations in 1995. Control of these snails has required expensive control measures at the facility. Discharge loads have also been reduced by a 1996 project using treated effluent to irrigate sports complexes (Horton 1997).

Several programs have been developed to reduce NPS by regulating land uses and activities. The comprehensive plan for Washoe County contains a conservation element to guide land use practices for the protection of water quality and riparian habitat. The plan uses zoning regulations to protect wetlands and the floodplain from development, and requires that all development be consistent with the primary uses of riparian areas, e.g. wildlife habitat, floodways, and water quality protection. Development design and construction practices must also protect water quality, reduce erosion, and preserve natural drainage and riparian habitats (Washoe County Planning Division 1991). To reduce NPS pollution in stormwater discharge, a regional stormwater management program has been developed for the Truckee Meadows. Components of the program include land use planning and ordinances to reduce runoff, requiring structural controls such as buffer strips and grassy swales to improve runoff quality, regulating industrial and construction site stormwater management, reducing illegal discharges, and improving municipal street operations. The program places a large emphasis on public outreach, to both educate the public about water quality issues and to provide a complaint and reporting system (Kennedy/Jenks Consultants 2001).

The recent shift from large-scale agriculture to urban development has resulted in a growing number of small ranches and hobby farms. This new breed of farmer has presented an opportunity for introducing environmentally friendly land management practices. To this end, the University of Nevada Cooperative Extension created a small ranch manual aimed at new ranch owners. Available in both print and over the internet, the manual discusses the importance of and methods for agricultural best management practices, animal waste management, irrigation efficiency, erosion control, the use of streamside buffers, fertilizer and pesticide use, exotic plant control, and water efficient landscaping (University of Nevada 2002). Additionally, programs have been developed to protect waterways from cattle grazing by providing watering facilities and installing fences and stream crossings (Klebenow and Oakleaf 1984, Pyramid Lake Paiute Tribe 2004).

A few innovative projects have been developed to investigate the use of treatment wetlands for improving the quality of water already in the drainage system. As Steamboat Creek is the largest contributor of non-point pollution to the Truckee River (Horton 1997), it has been targeted for water quality improvement efforts. One such project evaluates redirecting a portion of the instream flow into constructed wetlands for nutrient removal; some initial positive results have been observed (Dennett and Spurkland 2001). The creation of wetlands and floodplain forest associated with the restoration of lower Steamboat Creek is expected to provide ancillary water quality benefits, as are similar projects on the Truckee River (Washoe-Storey Conservation District 2001, The Nature Conservancy 2003, USACE 2003).

Exotic wildlife and vegetation control

The control of exotic vegetation is a component of many of the programs aimed at restoring ecological integrity to the Truckee River system. Ambitious removal projects have been included in floodplain restoration plans for Steamboat Creek and McCarran Ranch (Washoe-Storey Conservation District 2001, The Nature Conservancy 2003), as well as other preliminary plans for Truckee River restoration. Restoration plans proposed by the Washoe County Community Coalition include not only the removal of exotic vegetation from the riparian zone, but removal of nearby seed sources outside the project area (Montgomery Watson Harza 2002a). Of particular interest are tall whitetop and tamarisk, both of which are able to competitively exclude native floodplain vegetation. Volunteer 'Weed Warriors' are being trained to eradicate these species in the Truckee River Basin (Donaldson 1997) and a full-time steward will control exotic plants at the restored McCarran Ranch (Gourly, C. *personal communication* September 2005). While mechanical removal and/or chemical control will most likely be required in many locations, the restoration of the natural flow regime may be able to control these species, especially tamarisk, in the floodplain (Levine and Stromberg 2000). Biological control may also reduce tamarisk populations; the Chinese leaf-eating beetle (*Diorhabda elongate deserticola*) was approved for release in 1999 to control the exotic tree (SER International 2004).

The U.S. Fish and Wildlife recovery plan for the Lahontan cutthroat trout calls for the removal of exotic fish, especially exotic trout species, in some Truckee River Basin streams (U.S. Fish and Wildlife Service 1994). However, such removal programs are not likely to be planned due to sportfishing interests (Gourly, C. *personal communication* September 2005).

Analysis

Changes in environmental processes

Human activities in the Truckee River Basin have directly and indirectly altered many of the processes that shape the Truckee River ecosystem. The most influential changes have been to water balance and fluvial processes. Water diversions throughout the system severely disrupted the ecosystem, especially below Derby Dam. The loss of nearly half the river flow stranded entire communities (e.g. floodplains and wetlands) that were dependent on water supply. The geomorphology of the channel below Derby Dam and the river's interaction with Pyramid Lake were dramatically changed as their erosional, depositional, and chemical processes became those of a much smaller river.

Modifications to the river channel for flood control purposes were no less benign. By creating a wider and deeper channel, most interactions between the river and the surrounding landscape were lost. In addition, the new channel shape was not sustainable by the river flows, causing excess erosion and entrenchment in some locations and sedimentation in others. The resultant alteration of bed substrate characteristics degraded the habitat of many organisms, especially invertebrates and fish larvae. Lowering of the water table caused by entrenchment, combined with the loss of flood flows, resulted in a remarkable loss of floodplain vegetation and wildlife.

The damage to biotic communities caused by the disruption of these processes causes a feed-back system of continuing ecosystem decline. Floodplain forests and wetlands provided services such as streambank stabilization, flood dissipation, and nutrient uptake. The loss of these services has resulted in additional streambank erosion and entrenchment, more forceful flood flows, and further eutrophication of the system. Agricultural and urban development, in addition to removing habitat areas, altered the chemical balance of the river system. Increased nutrient loading changed the trophic structure of the river, affecting production and food web dynamics, further altering the habitat for aquatic organisms. In addition, the concentration of salts in Pyramid Lake threatens to create toxic conditions for lake-dwelling organisms.

The main ecological concerns of environmental managers on the lower Truckee River are degraded water quality, reduced cui-ui and Lahontan cutthroat trout populations, and reduced riparian forest area. These conditions are the result of the alteration of four main processes: evapotranspiration, flow regime, geomorphology, and nutrient supply. Other processes that likely affect the ecosystem are competition, predation, and hybridization by aquatic fish, competition by aquatic plants, and cattle grazing. Alteration of these 'source processes' caused a cascade of effects where other environmental characteristics and processes were affected; multiple causes and effects of each altered process are common., The complexity of these interactions can be illustrated as a process web (Figure 3.12), and a complete listing of altered processes and their modes of action is given in Table 3.2.

The principal processes affecting water quality are increased evapotranspiration (represented by the removal of river water for agriculture), increased nutrient supply, an altered flow regime caused by dams, and altered flow characteristics caused by channelization. Several

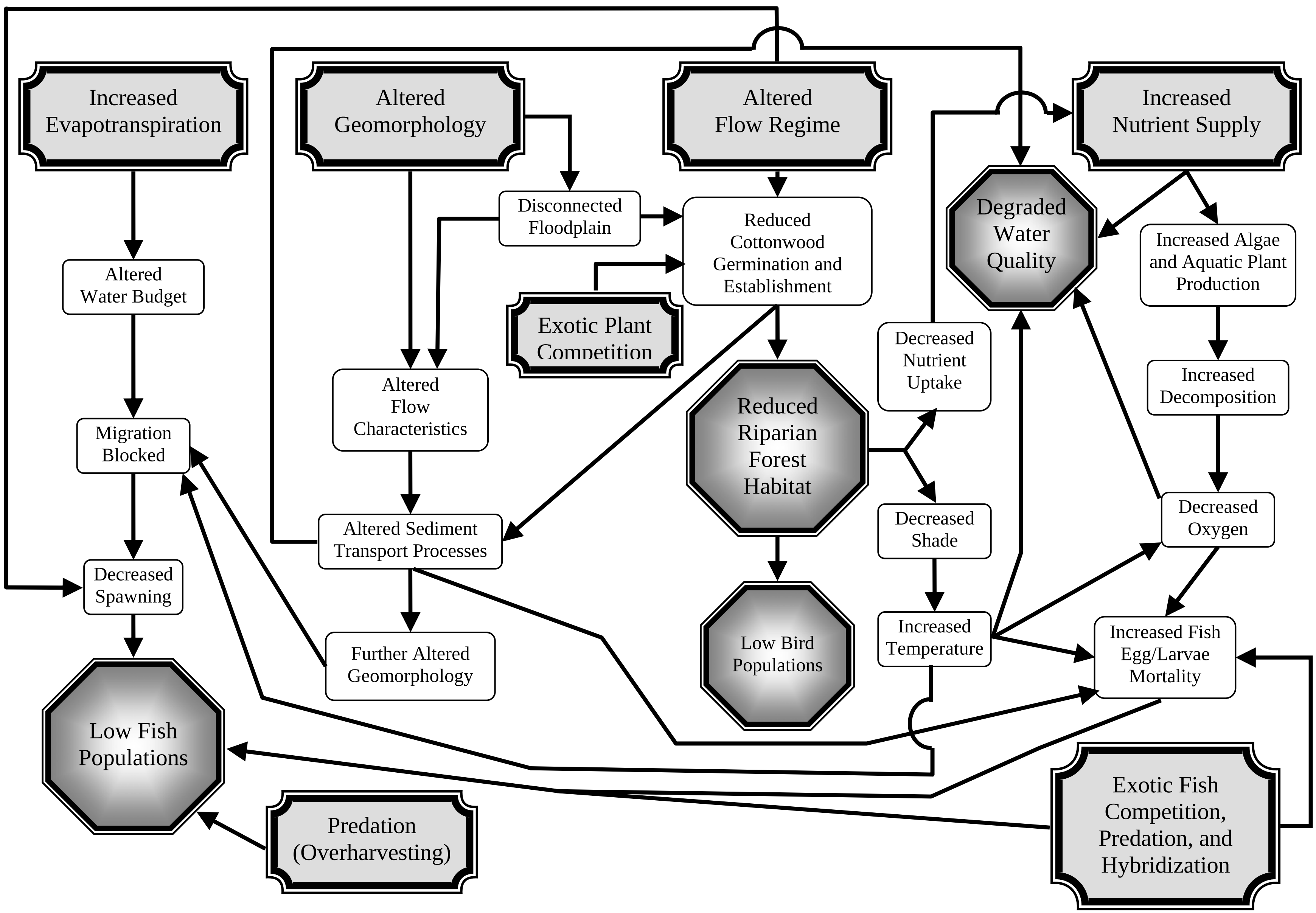

Figure 3.12: Web of altered processes and effects for the lower Truckee River, Nevada.

Table 3.2: Altered processes causing ecological degradation

Ecological Problem	Altered Process	Mode of Action
Reduced cui-ui population	Migration prohibited	Prevents access to spawning grounds
	Flow characteristics from channel alterations and barriers	Prohibits flow regime necessary to stimulate spawning runs, alters sediment transport patterns
	Sediment transport	Degrades quality of spawning substrate, creates delta at mouth of river
	Increased nutrient inputs	Increases primary production
	Decreased shading	Increases temperature and primary production
	Increased primary production	Increases biomass for decomposition
	Increased decomposition	Lowers oxygen levels in river sediments, thereby killing eggs and larvae
	Increased temperature	Lowers oxygen levels
	Competition/predation from exotic species	Exotics consume prey items, may consume young fish
Reduced Lahontan cutthroat trout population	Migration prohibited	Prevents access to spawning grounds; prevents survival of metapopulations
	Increased death rate (overharvesting)	Reduces population
	Flow characteristics from channel alterations and barriers	Alters sediment transport patterns
	Sediment transport	Degrades quality of spawning substrate, creates delta at mouth of river
	Increased decomposition	Lowers oxygen levels in river sediments, thereby killing eggs and larvae
	Increased temperature	Lowers oxygen levels
	Competition/predation from exotic species	Exotics consume prey items, may consume young fish
	Spawning behavior and hybridization	Hybridization between exotic trout and Lahontan cutthroat trout

Table 3.2: continued.

Ecological Problem	Altered Process	Mode of Action
Degraded water quality	Decreased water input to lake	Lake water becomes concentrated
	Increased evapotranspiration in river (in the form of agricultural diversions)	Decreases inflow to lake
	Increased nutrient inputs to river	Increases nutrient load in lake
	Increased radiation (temperature)	Lowers dissolved oxygen
	Riparian forest regeneration prevented	Increases radiation
	Altered flow velocity (from channel modifications)	Alters erosion and sedimentation rates, turbidity
Loss of riparian forest	Increased herbivory (cattle)	Removes understory and seedlings
	Inhibited cottonwood germination and establishment	Prevents regeneration of riparian forest
	Decreased flooding	Inhibits cottonwood germination
	Altered flow regime	Prevents cottonwood seedling establishment
	Competition from exotic plants	Exotics prevent reestablishment of native species.

other processes have been altered due to their interactions with these main processes. The increase in evapotranspiration reduced the water budget for the lake, which concentrated dissolved compounds in the water, affecting both river and lake chemical processes. An increased nutrient supply resulted in increased algae and aquatic plant production, causing increased decomposition of dead plant material, thereby lowering dissolved oxygen concentrations. The altered flow regime reduced riparian forest habitat, thereby reducing shading, increasing temperature, and further decreasing oxygen concentrations. Lastly, altered flow characteristics resulted in altered erosion and sedimentation rates, causing increased turbidity.

The principal processes affecting cui-ui and Lahontan cutthroat trout populations are increased evapotranspiration, an altered flow regime and flow characteristics, increased nutrient supply, and competition and/or predation by exotic fish species. The reduced water budget caused by increased evapotranspiration blocks the migration of these fish out of Pyramid Lake by exposing the delta at the outlet of the Truckee River. In addition, the altered flow regime prevents cui-ui migration by failing to stimulate spawning runs. The altered flow characteristics affected the channel substrate (by altering sediment transport processes), degrading spawning substrates. The degradation of water quality caused by an increased nutrient supply increased the mortality of cui-ui and Lahontan cutthroat trout eggs and larvae, as well as prevented the migration of Lahontan cutthroat trout through warmer river waters during the summer. Predation (as overharvesting) was a major contributor to the decline of the Lahontan cutthroat trout population. Lastly, competition and predation by exotic fish species increased fish mortality and decreased the birth rate of Lahontan cutthroat trout due to hybridization.

The principal processes affecting the floodplain forest are the altered flow regime, reduced flooding, increased herbivory (cattle), and competition from exotic plants. The altered flow regime and reduced flooding prevented the germination and establishment of Fremont cottonwoods. Competition from exotic plants likely lowered recruitment as well, and likely reduced willow abundance. Herbivory from cattle removed many of the seedlings that were able to germinate. Bird populations plummeted due to decreased forage, cover, and breeding sites resulting from floodplain forest loss.

Restoration of source processes along the Truckee River

An ideal process-oriented restoration project would restore the altered source processes (in this case evapotranspiration, geomorphology, flow regime, and nutrient supply) to support the former ecosystem. However, more concrete goals, reflecting the more easily observed symptoms of ecosystem decline, have often been emphasized in the planned projects to gain public support. As a result, planners and interest groups have focused on cui-ui and Lahontan cutthroat trout populations, the amount of riparian forest and floodplain loss, and degraded water quality. In addition, comprehensive restoration throughout the system has not been planned due to the overwhelming size of the affected area. However, completed and planned restoration projects to achieve managers' goals often included localized management of one or more of the source processes.

Evapotranspiration

Evapotranspiration due to agricultural irrigation reduced the water budget of the lower Truckee River by more than half, lowering the surface elevation of Pyramid Lake. A major accomplishment towards restoring the water budget of the Truckee River has been the acquisition of water rights. These rights were obtained by reducing the amount diverted at Derby Dam (as required by court decree) and the purchase of water rights from willing farmers. Projects promoting water use efficiency for agricultural and municipal uses are hoped to further reduce the demand for water withdrawal. While agricultural uses are limited by water rights, the total volume of water used by households and other municipal entities is not regulated. As a result, even though ordinances prohibit leaks and dictate when outdoor water use is permitted, consumption for any size of irrigated landscape, pool, and/or water feature is allowed. Consequently, municipal water use in Reno and Sparks is likely to remain high, due to the demand for large, lush landscapes and pools.

Since societal restrictions prevent the full restoration of the water budget for the Truckee River, water levels at Pyramid Lake are not likely to rise to their former levels. Although wetter than average conditions in the early 1990s did raise the water level at the lake, future dry years should be expected. Since evapotranspiration and water budget processes can not be fully restored, the Marble Bluff fish facility attempts to mitigate their effects on fish migration processes. Recent improvements to the facility have been successful in promoting cui-ui migration through the Truckee River delta, but not that of Lahontan cutthroat trout. Planned facility improvements may alleviate this shortcoming in the future.

Evapotranspiration processes were manipulated mainly for the purposes of cui-ui and Lahontan cutthroat trout migration. The possible negative effects of the decreased water budget on other aquatic species (e.g. invertebrates and other fish species) were not considered. While it is possible that the Marble Bluff fish facility may mitigate low flow effects on fish migration, it would not improve conditions for other species. To truly restore the Truckee River ecosystem, it should be determined if the decreased water budget affects other aquatic species, as well as physical environmental characteristics (e.g. groundwater recharge). If they prevent the restoration of the ecosystem, such effects should be addressed in future projects.

Flow Regime

The many dams on the Truckee River created a static hydrograph, affecting many organisms and processes that require temporal variation. Improved management of the flow regime during the spring months resulted in striking improvements to the ecosystem; this management strategy alone has reestablished a network of environmental processes that positively affected channel morphology and water quality in addition to wildlife and plant communities. The new flow regime has been very successful in stimulating the cui-ui spawning run, and provides the flooding depths required for Fremont cottonwood regeneration and willow thicket growth. The return of these floodplain plants near the river's edge restored aspects of sediment erosion and deposition processes, which in turn restored characteristics of river morphology and flow characteristics (width and depth). The mitigation of temperature and DO concentrations by these process changes allowed Lahontan cutthroat trout to travel along the river during summer months. The increase in riparian bird populations that occurred during these changes suggests that the regenerating floodplain forest restored many migratory, breeding, and foraging processes. Although not documented, it is likely that nutrient uptake by the floodplain forest will increase, and that sediment transport patterns may provide more favorable spawning sites for cui-ui and Lahontan cutthroat trout.

While the benefits of the restored springtime flow regime are considerable, the impact of the flow regime on the ecosystem during the remainder of the year has not been studied. Since this flow remains static, it is likely to have had negative impacts on other, less conspicuous, aspects of the ecosystem (e.g. amphibian populations or soil chemical processes). The degree of any such impacts will not be understood until adequate studies have been performed; it may be discovered that additional flood regime restoration is required.

Geomorphology

The channelizing of the Truckee River and Steamboat Creek negatively affected flow characteristics, consequently affecting sediment transport processes and further altering the geomorphology of the system. This in turn degraded both aquatic and floodplain habitats. Projects aiming to restore geomorphology focus on channel restoration (e.g. meanders, depth, width, and elevation) and floodplain connectivity. Hydrologic and ecologic computer modeling has been used extensively to aid in the design of a morphology that supports the desired river processes and characteristics, and more successful projects will likely result. However, few projects addressing geomorphology have been implemented to date, and sufficient time has not passed to determine their effectiveness. Other projects are in the planning stage. In addition, since projects are designed as small channel segments, complete hydrologic and ecologic models for the whole ecosystem can not be created. This piecemeal approach, however, might not significantly impede restoration of the river since channel alterations (e.g. bedrock removal) prevent the restoration to its exact historic form. Plans for restoring the geomorphology of the river will thus need to focus on the creation of an analogous, but not replicated, system.

Lahontan cutthroat trout populations have been greatly affected by morphological barriers created by the dams above Derby Dam. Currently, no projects have been planned that restore the process of fish migration over these dams. As a result, the metapopulations required to sustain Lahontan cutthroat trout can not be maintained. Consequently, this species will require continued hatchery support for its survival.

Nutrient Supply

Increased nutrient availability in the Truckee River has led to its eutrophication, causing trophic changes and lowering DO concentrations. Improved wastewater treatment facilities have reduced a major nutrient source. The promotion of improved agricultural and streamside practices, as well as the implementation of stormwater quality programs, will likely lower NPS inputs. However, NPS reductions are difficult to predict as many of the recommended land use practices are voluntary.

The restoration of the floodplain forest and river geomorphology may further reduce nutrient concentrations. Restored floodplain forests and new wetlands created by a restored geomorphology can result in increased nutrient uptake, as well as denitrification in saturated soils. Such denitrification may prove especially beneficial, as the Truckee River ecosystem is thought to be nitrogen limited.

A complete restoration of nutrient inputs is not likely using current technology as historic nutrient inputs in the lower Truckee River were exceptionally low. As a arid system, the watershed of the lower river contributed very little runoff, and hence, very few nutrients. The addition of agriculture and the growth of urban centers in the watershed have resulted in both increased irrigation runoff and increased nutrient supply. The reduction of nutrient concentrations in water entering the river to historic levels may never be accomplished.

Exotic Species

While having less impact than the aforementioned source processes, exotic species have affected aspects of the ecosystem. Hybridization between introduced trout species and Lahontan cutthroat trout may have a considerable effect on recruitment. In addition, it is generally agreed that competition and predation from the introduced trout are negatively impacting the native trout populations. Predation may also be affecting cui-ui populations, as well as those of other aquatic species. Unfortunately, the demand for sport fish is likely to result in continued stocking of exotic trout. Consequently, processes altered by these species will not likely be addressed.

The effect of exotic plant species, however, is recognized, and these plants are regarded as nuisances. The restoration of the river's flow regime has resulted in a decline in tamarisk populations, likely due to an increase in the duration of spring flooding. Currently, mechanical removal is the best technique for controlling tall whitetop in areas adjacent to the river. This labor-intensive method, as well as constant monitoring and maintenance, will be required in perpetuity unless the processes that encourage tall whitetop invasion are understood and can be managed.

A summary of altered environmental processes and associated restoration initiatives is provided in Table 3.3. Many of the restoration goals are likely to be achieved due to approaches that target the degraded source processes of the ecosystem. To further this effort, it would be beneficial if planners began emphasizing the term 'process' in restoration plans, to encourage deliberate attention to the underlying natural forces shaping the system. This would enable stakeholders and planners to recognize where environmental processes are not addressed by proposed solutions. Of course, replications of original processes are not being attempted due to societal water and infrastructure requirements. However, for the planned restoration projects to be feasible and supported, certain societal impacts must be allowed. Therefore it is critical that these impacts be accounted for in the new ecosystem design.

Table 3.3: Restoration initiatives to repair damaged environmental processes.

Ecological Problem	**Altered Process**	**Restoration Initiatives**
Reduced cui-ui population	Migration prohibited	Increased river flows to provide passage through the river outflow delta; improved fish passage at the Marble Bluff Dam
	Flow characteristics from channel alterations and barriers	Restoration of channel characteristics such as gradient, meander, and floodplain connection
	Sediment transport	Restoration of channel characteristics and sediment retention in floodplains
	Increased nutrient inputs	Management of both point and non-point sources; nutrient uptake through reestablishment of floodplain forest and wetlands
	Decreased shading	Restoration of cottonwood riparian forest through flood regime management
	Increased primary production	Management through decreased nutrient loads and increased shading
	Increased decomposition	Alleviated by decreased algal production
	Increased temperature	Alleviated by increased shading and the restoration of a deeper, narrower channel
	Competition/predation from exotic species	Exotic vegetation control programs and restoration of the natural flow regime

Table 3.3: continued.

Ecological Problem	Altered Process	Restoration Initiatives
Reduced Lahontan cutthroat trout population	Migration prohibited	Increased river flows to provide passage through the river outflow delta; improved fish passage at the Marble Bluff Dam
	Increased death rate (overharvesting)	Improved fishery management and hatcheries
	Flow characteristics from channel alterations and barriers	Restoration of channel characteristics such as gradient, meander, and floodplain connection
	Sediment transport	Restoration of channel characteristics and sediment retention in floodplains
	Increased decomposition	Alleviated by decreased algal production
	Increased temperature	Alleviated by increased shading and the restoration of a deeper, narrower channel
	Competition/predation from exotic species	Not addressed
	Spawning behavior and hybridization	Not addressed
Degraded water quality	Decreased water input to lake	Acquisition of water rights to support ecosystem
	Increased evapotranspiration in river (in the form of agricultural diversions)	Initiatives to decrease irrigated land and increase irrigation efficiency
	Increased nutrient inputs to river	Management of point and non-point sources; nutrient uptake through reestablishment of floodplain forest and wetlands
	Increased radiation (temperature)	Restoration of floodplain forest
	Riparian forest regeneration prevented	Natural flow regime implemented
	Altered flow velocity (from channel modifications)	Restoration of channel characteristics such as gradient, meander, and floodplain connection
Loss of riparian forest	Increased herbivory (cattle)	Improved cattle grazing management
	Inhibited cottonwood germination and establishment	Restoration of the natural flow regime
	Decreased flooding	Channel modifications to provide floodplain connection
	Altered flow regime	Restoration of the natural flow regime
	Competition from exotic plants	Exotic vegetation control programs and restoration of the natural flow regime

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CHAPTER 4

THE ROLE OF THE LANDSCAPE ARCHITECT IN PROCESS-ORIENTED ECOLOGICAL RESTORATION

Introduction

Environmental processes are the underpinnings of freshwater ecosystems. However, the understanding and manipulation of these processes in and of themselves do not constitute an ecological restoration plan. A restoration plan, to be complete, must incorporate the whole of the physical environment and its temporal variations, and be compatible with societal needs. To encompass these aspects, Boon (1998) proposed five dimensions of river restoration: conceptual, spatial, temporal, technological, and presentational. The conceptual dimension represents the purpose of the restoration effort and the overall approach that is taken. Aspects include goals and values, scope, and ecological theory. The spatial dimension represents the physical elements of the project area and its surrounds, as well as their arrangement in the landscape. Examples of these aspects include connectivity, scale, and geomorphology. The temporal dimension recognizes that a landscape is a product of its past and that it will continue to change in the future. As such, historic characteristics, future management, and temporal variations intrinsic to the ecosystem should be included in the design process. The technological dimension is comprised of the myriad of techniques for both the gathering and processing of information and the installation of project features. The tools used encompass the fields of science, engineering, architecture, economics, sociology, and others. Examples include the modeling of hydrology, geomorphology, and ecology, GIS analysis, and recent structural innovations. The presentational dimension acknowledges that projects can not succeed without the support of a wide audience. This audience includes conservationists, scientists and engineers, politicians, business groups, and the general public. These groups must be educated about both the need for restoration as well as societal benefits (such as water quality, flood control, and recreation) to justify the effort and expense involved. As these groups have varied backgrounds and interests, presentational materials must be tailored for each audience.

The Role of the Landscape Architect

The field of landscape architecture is innately suited to address the five dimensions of restoration. Indeed, these dimensions are routine aspects of design. The landscape architect approaches the conceptual dimension by generating a design concept, defining the project's purpose and intent, developing a program and style, and maintaining a sense of place. The spatial dimension is clearly demonstrated by the analysis and design of the layout, topographical shape, and patterns of landscape elements. The temporal dimension plays a critical role as the designer addresses a landscapes history, imagines how it is to be used, anticipates its continuing development, and develops a management and maintenance plan. Environmental fluctuations are even considered, as a good designer not only works around the effects of season and weather, but uses them to his advantage. The arena of the technological dimension includes aspects of site engineering, construction, and the adoption of new materials, as well as computerized advances such as drafting and imaging software. Lastly, the landscape architect is keenly aware of the presentational dimension. Projects and plans must be presented in a manner that will attract, engage, and convince the client and general public to adopt the proposed project. At the same time, the presentation must assure the client of the landscape architect's competency while retaining an understandable context.

Recognition of the many interconnected facets that comprise ecological restoration, and the intricate level of sensitivity at which they operate, has led to a recent emphasis on the need for an interdisciplinary approach (Franklin 1997, Clifford 2001, Musacchio 2004). Good restoration planning requires attention to scientific, technical, political, and socio-economic issues (Clifford 2001); each may require a set of experts. However, these groups often find it difficult to collaborate, or even communicate, due to differences in terminology, scale, and focus (Benda et al. 2002). In addition, project success requires that the stakeholders (politicians and general public) understand and are supportive of the restoration plan. The inputs and concerns of the public must be addressed, and the expense of the project is made more palatable if benefits are shown for both infrastructure and recreation (Higgs 1997). Since communication of specific, ecological, social, and economic concerns is difficult for many citizens, meaningful discourse between the public and the planners, scientists, and engineers involved in the project can be nearly impossible to achieve.

Despite these wide communication gaps between the various project entities, incorporating scientific understanding and public needs into a concrete restoration plan is essential. As such, the landscape architect can play a pivotal role through his unique abilities to communicate with each group and translate the information into a restoration program, design, and plan. In this manner, the landscape architect can act as an information conduit, integration center, and project coordinator. The sharing of information between project entities can be facilitated by its translation into commonly understood language and terminology, thus allowing the various groups to respond to each others recommendations, concerns, and requirements. Recommendations, ideas, and pertinent information can then be distilled and integrated into a strategy that adequately addresses both environmental and social concerns. Lastly, the landscape architect is in an excellent position to coordinate the entire restoration process, from preliminary assessment through plan finalization, project installation, and even post-installation monitoring and adaptive management. A conceptual notion of the landscape architect's role and relationship with other project entities is shown in Figure 4.1.

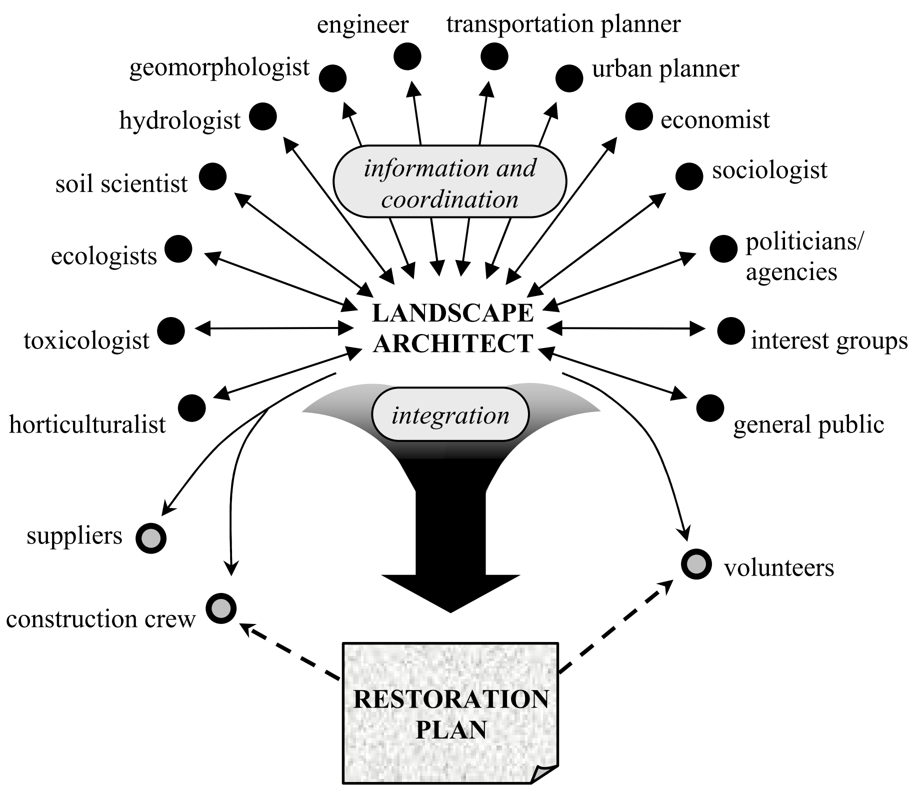

Figure 4.1: The role of the landscape architect in the ecological restoration design process.

Ecological restoration can be viewed as an expansion of traditional landscape architecture into the larger landscape. Broken into their basic components, the two forms of design show striking similarities (Table 4.1). In their fundamentals, both address the form and design of

Table 4.1: Comparison of the traditional landscape architecture design process to ecological restoration planning.

Components of Traditional Landscape Architectural Design	Components of Ecological Restoration Planning
Client Interaction	
• Communicate with client	• Communicate with stakeholders
• Determine client's underlying conceptual needs	• Determine stakeholders underlying conceptual needs
• Educate client about their site's characteristics and potential	• Educate stakeholders about their interaction with their landscape
• Understand clients needs and incorporate them into the design	• Understand stakeholders needs and incorporate them into the design
• Generate design alternatives that satisfy the underlying client needs	• Generate design alternatives that repair the underlying ecological processes
• Generate a final design desirable to the client	• Generate a final design desirable to most stakeholders
• Present design to client	• Present design to stakeholders
Baseline Evaluation	
Determine: • Current site conditions • Temporal variability • Historical site conditions • Future landscape uses	Determine: • Current site conditions • Temporal variability • Historical site conditions • Future societal needs and ecological development
Design Planning	
• Understand integration of site into landscape (sense of place) and incorporate into design	• Understand site and landscape connectivity (interactions/influences) and incorporate into design
• Create a site program	• Create a site program
• Determine soil conditions and how they may need to be altered to support desired landscape	• Determine soil conditions and how they may need to be altered to support desired ecosystem
• Determine the landscape's water needs and incorporate into the design	• Determine landscape and regional hydrology and incorporate into design
• Understand the influence of seasonal conditions and incorporate them into the design	• Understand the influence of temporal influences and incorporate them into the design
• Determine how the site morphology needs to be changed to accommodate the new landscape	• Determine how the site morphology needs to be changed to accommodate the new landscape
• Understand different type of landscape communities (e.g. sun/part shade/shade; wet/moist/arid; tropical/temperate/boreal)	• Understand different ecological communities (e.g. open/closed canopy; upland/floodplain/wetland; lotic/lentic; tropical/temperate/boreal)
• Understand and account for the requirements of vegetative landscape components	• Understand and account for the requirements of vegetative landscape components
• Understand and account for the requirements of animal landscape components (e.g. birds, butterflies, insect pests)	• Understand and account for the requirements of animal landscape components (e.g. wildlife, domestic)
• Understand potential needs for fertilizers/soil conditioners	• Understand chemical interactions of the ecosystem
• Understand hardscape elements' potentials limitations	• Understand the local and regional infrastructure needs

Table 4.1: (continued).

Components of Traditional Landscape Architectural Design	Components of Ecological Restoration Planning
Final Design	
• Incorporate information from specialists (e.g. engineers, horticulturalists)	• Incorporate information from specialists (e.g. scientists, engineers, economists, social scientists)
• Plant specification	• Plant specification
• Locate suppliers and/or contractors	• Locate suppliers and/or contractors
• Develop maintenance program	• Develop adaptive management program
• Draft final plan	• Draft final plan
Implementation	
• Create implementation schedule	• Create implementation schedule
• Allocate and maintain budget	• Allocate and maintain budget
• Communicate with contractors	• Communicate with contractors and/or volunteers
• Adjust plan to unexpected problems	• Adjust plan to unexpected problems
• Report to client	• Report to stakeholders
• Provide future consultation	• Provide future consultation and monitoring

landscapes and draw on similar methods of site evaluation, design planning, implementation, and client interaction. Differences between the two design types can include project scale, complexity, values (including aesthetics and design principles), and the intended beneficiaries. Project scale becomes complicated by the fact that the aquatic system is a product of everything in its catchment. Therefore, to effectively coordinate and design a restoration project, the practitioner must consider the effects of the greater landscape, often at the watershed scale. As such, the design process utilizes landscape ecology in the examination of landscape characteristics and environmental processes throughout the watershed. This permits the evaluation of the ecosystem using a systems approach, in an effort to incorporate all of the environmental characteristics and processes, as well as process interactions and feedback, that shape it.

Precepts for Process-Oriented Ecological Restoration

The multifaceted nature of ecological restoration can complicate the planning process. To guide project development, a set of eight design precepts is proposed in Table 4.2. The first precept is that ecological recovery remains the primary goal of the project. Should other goals supercede this aim, the project is likely to slide into the realm of ecological enhancement or aesthetic improvement. The second precept is that project assessment and planning should focus on the environmental processes that shape the ecosystem. The third is that a system analogous to the original be designed, if possible. Although exact ecosystem replication is never possible, the practitioner should not take license to, for example, design an herbaceous marsh where a floodplain forest once stood. While the establishment of other, perhaps diminishing, habitats can be beneficial for a region, such activities result in habitat creation instead of restoration. Of course, this is a scale-based interpretation; the rearrangement of habitats within a project area (due to constraints such as infrastructure or severely altered morphology) could constitute restoration if the resultant ecosystem exhibits its original overall functions and features.

Precepts four and five pertain to the coordination of interdisciplinary teams to provide specialized information, and the integration of that information into the design process.

Table 4.2: Eight precepts for process-oriented ecological design

1. Ensure that ecological recovery is the primary goal
2. Ensure that the project is process-oriented
3. Strive for the creation of an analogous ecosystem
4. Coordinate an interdisciplinary restoration team
5. Integrate expertise from restoration team into design process
6. Maintain a high level of public education and involvement
7. Maintain a holistic viewpoint, addressing the catchment, reach, and microhabitat scales
8. Include floodplains and wetlands as integral components of river ecosystems

Similarly, the sixth precept maintains that all stakeholders be invited to participate so that their opinions and needs may be addressed. The development of educational materials may be necessary to facilitate this involvement and to help stakeholders make informed choices. Such community involvement is necessary for gaining the support required for restoration projects. In addition, local residents, having a more vested stake in the outcome, often push for comprehensive solutions. Such was the case in Nevada's lower Truckee River, where the local community developed and demanded the most ecologically-oriented and sustainable alternative (Montgomery Watson Harza 2002). The ability to instill pride in the local community of their natural resources can create tremendous support for the project, as well as encourage future environmentally sensitive planning decisions. Depending on the project, specialists such as economists and/or sociologists may be needed on the interdisciplinary team to address stakeholder concerns.

The seventh and eighth precepts pertain to ecosystem integrity, connectivity, and complexity. To adequately address the problems of the project area, the catchment, reach, and microhabitat scales must be considered. At the reach scale, all adjacent aquatic, hydric, and floodplain areas must be considered as integral parts of the system.

A Process-Oriented Ecological Restoration Protocol

While each project is unique, precluding the use of formulaic solutions, a generalized protocol can be used to guide the planning process. Table 4.3 presents a protocol in four phases, each divided into its requisite activities. While the protocol presents a sequence of these tasks, many of them likely overlap in time, or may be revisited later in the planning process.

Table 4.3: Generalized protocol for process-oriented ecological design

Assessment

1. Identify the primary symptomatic attributes of the ecosystem
2. Assess current site characteristics, history, and land use
3. Identify project constraints
4. Define and assess appropriate reference site(s)
5. Acquire a basic understanding of each attribute of the ecosystem
6. Determine desired ecosystem trajectory
7. Re-evaluate symptomatic attributes of the ecosystem
8. Determine all possible causes of degradation, focusing on environmental processes
9. Evaluate possible causes, determine the influence of each, and how they are influenced, possibly using computer models

Design development

10. Determine measurable goals and objectives
11. Generate conceptual program
12. Refine concept with input from all involved parties
13. Generate alternative plans based on concept and program
14. Refine alternative plans with input from all involved parties, using computer models

Design finalization

15. Evaluate alternative plans with input from all involved parties
16. Develop restoration plan (document and possibly construction plans)
17. Design adaptive management plan

Implementation and monitoring

18. Oversee implementation and installation or provide guidance for project oversight
19. Participate in post-installation monitoring
20. Continue relationship with client; document outcomes and project adjustments

The design sequence itself is quite similar to that of traditional landscape architecture, with familiar site assessment, design development, plan finalization, and project implementation phases. The principal difference may be that the site boundaries are somewhat indistinct, or even unknown, at the outset. The effects of environmental activities and processes are not dampened by property lines and political boundaries. While the targeted environmental symptom may be limited to one location (although such confinement is unlikely), it may be caused by the alteration of environmental processes elsewhere in the watershed. Similarly, any proposed changes to environmental processes will affect downstream areas. As such, restoration projects must be approached with the recognition that boundaries are especially pervious.

Assessment phase

The nature of environmental problems is usually not well understood at the start of a project. However, some type of primary environmental symptom likely drew attention to the degraded system. Because of this, a client may approach the restoration designer with a request for a particular solution. To ensure a process-oriented solution, it is necessary to (gently) put the requested solution aside, in order to step back and determine the problem that caused the client to act. It is quite possible that the requested solution is not the answer to the problem (or may even worsen it). For example, catastrophic flooding along the Truckee River led Washoe County officials to contact a consultant with the request for an improved levee system (Gourley, C. *personal communication* September 2005). By gently stepping back, the consultant can investigate the nature of the true problem and propose alternative solutions that treat the underlying causes. In the case of the Truckee River, the consultant was able to convince the county that the river needed an area to flood into. As a result, the preferred alternative recreates floodplains in areas that will not damage the city. Residents are immensely happy and proud of the plan.

Of course, the primary environmental symptom provides the impetus for identifying the processes underlying ecological degradation, and is a good starting point for directing the focus of investigation. As the site's history, characteristics, and land use, as well as project constraints, are identified, appropriate specialists should be brought into the team to assist in evaluation and planning. Potential reference sites should be determined at this point so that a basis of comparison can be made. The coordinating landscape architect, with the help of the interdisciplinary team, should acquire a basic understanding of each environmental attribute of the project and reference site. In doing so, not only can the various components of the project be better integrated, but, perhaps more importantly, elements with complex interactions requiring additional expertise can be recognized.

By using the reference site as a guide, the interdisciplinary team can establish the desired ecological trajectory (direction of ecological development and progression) for the project area. A static endpoint is both undesirable and impossible to maintain; rather, a dynamic, living ecosystem should be intended. After the site is well understood and the desired trajectory is determined, the symptoms of environmental degradation should be reevaluated to create a more complete understanding of the ecological issues. All potential causes of each degraded aspect should be catalogued by the interdisciplinary team. These potential causes can then be evaluated, determining the influence of each and any possible interactions. The use of computer models may be appropriate to estimate the previous roles of environmental processes, their interactions, and ecological effects.

Design development phase

The development of the ecological restoration plan resembles that of traditional landscape architecture, except that environmental processes are the focal points of the design. Thus, measurable goals and objectives relating to the reestablishment or adjustment of altered processes should be defined. From these goals, the conceptual design program can be developed. Those stakeholders not already involved should be encouraged to participate at this stage and throughout the remainder of the project. Using suggestions and feedback from the interdisciplinary team, the landscape architect should generate several alternatives based on the conceptual program. This process entails much consultation and dialogue with the various team members and the client. Their critical review of the resultant alternatives is necessary to ensure project feasibility and self-compatibility. Computer models that incorporate the processes forming the ecosystem of interest, their interactions, and effects are likely to be required during design development, especially to evaluate hydrologic processes and effects. Such modeling can help ensure that the desired ecological trajectory is reached as well as reduce the risk of unforeseen impacts.

Design finalization

Once the alternative designs are developed, both the client and stakeholders can participate in final design selection. Part of this process should estimate whether the environmental processes affected in each alternative can restore and support the desired ecosystem. While final approval always lies with the client, stakeholder opinions should be weighed as appropriate for the project. The selected alternative can then be crafted into the final restoration plan, again with the critical review of the interdisciplinary team to ensure that the components are both technically sound and compatible. Computerized modeling may again be appropriate at this stage. The resultant plan takes on written form, possibly including construction documents. At a minimum, the plan should detail the various components of the project, their implementation, and expected costs. The rationale and relationships between components should be explained so that the overall aim of the project remains clear. As part of the document, an adaptive management plan should provide criteria for project monitoring and possible maintenance, as well as additional strategies should the restoration effort need potential adjustment.

Implementation and monitoring

The degree to which the landscape architect participates in project implementation and monitoring depends on the individual contract. Projects without a construction component, such as those consisting solely of regulatory and/or management changes, can be implemented without oversight. However, projects containing a large construction component can naturally benefit from the continued participation of the landscape architect so that any unexpected complications can be addressed. A complex project with an extended timetable may require long-term involvement so that restoration activities can be tailored according to project developments. At a minimum, the landscape architect should document the outcome of the project, its post-implementation development, and any modifications or adjustments made to the plan.

Concluding Remarks

The restoration protocol described above may appear to be expensive and difficult, requiring an army of specialists and planners. This is not necessarily so. The level of effort required in ecological restoration depends on the nature of the project. The restoration of a severely degraded large river that flows through a city center may indeed require the involvement of many specialists from several disciplines (not to mention the involvement of governmental agencies). However, smaller projects might involve just a handful of planners and one or two stakeholders. The ability to recognize when and where additional expertise is needed is the matter of importance, so that the root cause of ecological degradation can be appropriately remedied and the client enjoy a high likelihood of project success.

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