Water and Nitrogen in Designed Ecosystems:

Biogeochemical and Economic Consequences

by

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A Dissertation Presented in Partial Fulfillment
of the Requirements for the Degree
Doctor of Philosophy

Approved November 2010 by the
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December 2010

ABSTRACT

More than half of all accessible freshwater has been appropriated for human use, and a substantial portion of terrestrial ecosystems have been transformed by human action. These impacts are heaviest in urban ecosystems, where impervious surfaces increase runoff, water delivery and stormflows are managed heavily, and there are substantial anthropogenic sources of nitrogen (N). Urbanization also frequently results in creation of intentional novel ecosystems. These "designed" ecosystems are fashioned to fulfill particular needs of the residents, or ecosystem services. In the Phoenix, Arizona area, the augmentation and redistribution of water has resulted in numerous component ecosystems that are atypical for a desert environment. Because these systems combine N loading with the presence of water, they may be hot spots of biogeochemical activity.

The research presented here illustrates the types of hydrological modifications typical of desert cities and documents the extent and distribution of common designed aquatic ecosystems in the Phoenix metropolitan area: artificial lakes and stormwater retention basins. While both ecosystems were designed for other purposes (recreation/aesthetics and flood abatement, respectively), they have the potential to provide the added ecosystem service of N removal via denitrification. However, denitrification in urban lakes is likely to be limited by the rate of diffusion of nitrate into the sediment. Retention basins export some nitrate to groundwater, but grassy basins have higher denitrification rates than xeriscaped ones, due to higher soil moisture and organic matter content.

An economic valuation of environmental amenities demonstrates the importance of abundant vegetation, proximity to water, and lower summer temperatures throughout the region. These amenities all may be provided by designed, water-intensive ecosystems. Some ecosystems are specifically designed for multiple uses, but maximizing one ecosystem service often entails trade-offs with other services. Further investigation into the distribution, bundling, and tradeoffs among water-related ecosystem services shows that some types of services are constrained by the hydrogeomorphology of the area, while for others human engineering and the creation of designed ecosystems has enabled the delivery of hydrologic ecosystem services independent of natural constraints.

DEDICATION

To my parents, Jean and Jeffry Larson, and my brother Chris Larson. I could not have asked for a better family.

ACKNOWLEDGMENTS

It is a pleasure to thank those who made this dissertation possible, and the many more who created the outstanding community and Arizona State University that enabled me to complete it. I owe my deepest gratitude to my advisor, Dr. Nancy Grimm, who served as an excellent mentor and role model. I thank her for welcoming me into her lab, challenging me intellectually, offering many wonderful opportunities, and supporting me throughout. My committee, Drs. Hilairy Hartnett, Stuart Fisher, Marty Anderies, and Kitty Lohse were invaluable in providing guidance and feedback that has substantially improved this work. I am grateful to Drs. Charles Perrings and Ann Kinzig for the opportunities to collaborate and be part of the EcoServices group. I wish to thank everyone in the IGERT in Urban Ecology program and CAP LTER for creating the best interdisciplinary environment for research and collaboration. There are numerous people to thank for their helpful contributions of all kinds: insight, time, labor, and emotional support. Although it is not possible to acknowledge everyone, I am indebted to Ryan Sponseller, John Roach, Jim Heffernan, David Huber, Tamara Harms, Rebecca Hale, Christina Wong, Xiaoli Dong, Melissa McHale, Sharon Hall, Rich Sheibly, Jennifer Edmonds, David Lewis, John Schade, Jill Welter, John Schampel, Jessica Block, Alex Buyantuyev, Jolene Trujillo, Kris Gade, Jessica Corman, Dan Childers, Michelle McCrackin, Elizabeth Cook, Kayla Graham, Bony Ahmed, Sarah Moratto, Harriet Van Vleck, Katie Meyer, Patrick Ortiz, Stevan Earl, Beth Hagen, Bethany Cutts, Nate Toke, Kate Darby and Mark Neff. I am especially grateful for the help with laboratory analyses provided by

Cathy Kochert, Natasha Zolotova, and Tom Collela. Thank you, Kevin McCulloch, for getting me through the end. I thank Barry Redmond from the bottom of my heart for being my companion through most of this, encouraging and supporting me, providing me with laughs and entertainment, and most of all, protecting my sanity.

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1.1 Introduction

Water is essential for life and civilizations flourish when they find efficient means both to reliably provide water and eliminate excess or waste water. As a result, "more than half of all accessible freshwater is put to use by humanity" and "one-third to one-half of the land surface has been transformed by human action" (Vitousek et al. 1997) These impacts are not independent of each other and are particularly acute in urban ecosystems, as municipalities strive to ensure an adequate supply of drinking water and commensurate wastewater disposal. Urbanization also contributes to greater impervious surface area, which increases storm runoff (Arnold and Gibbons 1996). Thus, municipal managers and planners must contend with heightened flood risk in addition to creating infrastructure for essential water needs.

Historical approaches to dealing with water supply, waste elimination, and urban flooding often involved extreme manipulation of urban aquatic systems: damming of rivers and reservoir creation to assure water supply, discharge of minimally or untreated wastewaters to waterways, and burial, channelization, and lining of streams to hasten stormwater removal. Rapid urban population growth was often unanticipated and has exceeded the capacity of historic hard-engineered water supply and removal structures. Many older cities have experienced both an increase in impervious surface area and more municipal sewage users relying on an aging system, resulting in increased flooding, cross-contamination between sewage and stormwater drainage systems, and stream down-cutting. These combined impacts are manifest in the "urban stream syndrome" (Paul and Meyer 2001, Walsh et al. 2005). This syndrome is characterized by changes in stream morphology, discharge timing and duration. Coupled with elevated nutrient and pollution inputs, this often results in degraded ecological functioning, as indicated by nutrient cycling, metabolism, and species diversity and richness metrics.

The structural and functional failures of older systems due to population pressure, along with the visible and unappealing impairment of urban streams, led many planners and engineers to reconsider approaches to meeting the water needs of urban residents. Over the past few decades, changes in aesthetic desires, increased environmental awareness, and a deeper ecological understanding have been reflected in environmental engineering and the increased popularity of urban stream restoration (Poff et al. 1997, Gleick 2000, Lake 2002). For example, there has been a transition from hard- to soft-engineering in some cases, e.g., incorporating natural materials as opposed to rectilinear concrete structures (Gleick 2002, Hayes 2004, Galloway 2005). Additionally, managers and planners have recognized that some natural ecosystems can provide the same or additional services as their hard-engineered counterparts as well as being more pleasing aesthetically (Larson and Plasencia 2001).

The impetus to restore urban streams was influenced greatly by the perception of how a "natural," pre-urban stream should appear and function. Most early research on the ecology of streams has been conducted generally in mesic climates where it was assumed that the reference (i.e., pre-settlement ecosystem) was forested. However, there is now general agreement among ecologists that the identification and definition of the pre-urbanization state is contestable and likely unachievable given the constraints of existing urban pattern and structure. Instead, focus has shifted to enhancing specific stream structures and functions, such as habitat heterogeneity and floodplain connectivity (Bernhardt and Palmer 2007). These in turn can provide ecosystem services such as flood abatement, water purification, education, recreation, and aesthetics. We assert that what is called restoration is in fact design. Urban populations must inherently alter their environment, but well-planned design can incorporate ecological understanding to meet basic needs *and* deliver other valuable ecosystem services that enhance quality of life. Additionally, much of the recent urban population growth has been in arid and semi-arid areas (Sutton and Day 2004, UNEP 2006), and this trend is expected to continue. Due to the constraints that drier climates place upon water resources in arid cities, the lessons and practices for stream restoration and management learned from cooler, wetter zones may be inappropriate. Further, public and civic perceptions that motivate restoration and dictate water allocation may differ significantly among contrasting biomes.

With these concerns in mind, in this chapter we propose that planners, designers, and managers use an alternative framework for determining "restoration" and management goals based on ecosystem services. We illustrate how ecological design has the potential to create valuable aquatic habitats within the context of the climate and water-resource demands of aridland cities. The first section describes the framework and related concepts. We then provide some background information of general aridland hydrology, including the important ecological functions that occur in desert streams and rivers. Next, a description of how a desert city has modified this hydrology is provided in a case study of the Phoenix, Arizona metropolitan area. Here we show how the constraints of development in an arid setting influence how society meets its demand for water supply, storm and waste water removal, aesthetics, and recreation via hydrologic manipulations. Then, given the Phoenix environmental and historical context, the following section describes two "restoration" projects currently underway or completed, documenting the design elements and evaluating the ecosystem services provided. Finally, we present a discussion regarding the benefits of viewing urban hydrologic alterations from the perspective of well-informed ecological design rather than attempts at restoration. Ultimately, we conclude that ecological design that incorporates adequate consideration of a variety of ecosystem services, determined by the values of multiple stakeholders, is the most appropriate approach.

1.2 Conceptual Framework: "Flowers" of Ecosystem Services

There are four types of aquatic systems that cities must create and manage (Table 1.1). In some cases, it is appropriate for these systems to be heavily engineered, such as the water supply system, but in many cases there is room for both engineered and ecological solutions, or some combination of both. For example, many wastewater treatment plants use wetlands as a final treatment step to reduce nutrient loading to recipient systems, or neighborhood stormwater retention basins can also be used as soccer fields. However, there are inevitably trade-offs between some services, so that maximizing one may mean detracting from another (Grimm et al. 2004); for example to provide a clean water supply efficiently, it is often necessary to limit recreational activities in reservoirs. Therefore, cities need multiple kinds of aquatic habitats to fulfill the needs and desires of its residents. Some may be focused on providing just one or two services to a very high degree, while others may be designed to provide moderate levels of four or five services.

To help illustrate this, we borrow from Foley at al.’s 2005 conceptual framework, which creates “flowers” of ecosystem services that can be used both for assessing existing systems as well as and planning ecosystem design (Figure 1.1). Foley et al. (2005) focused on globally important ecosystem services such as carbon sequestration, crop and forest production, and water flow regulation. Here, we create a similar flower, but with ecosystem services relevant to aquatic habitats as identified in Table 1. As in Foley et al. (2005), the petals on the flower are by no means exhaustive and, in this heuristic example, are not labeled nor normalized with common units.

Ideally, all stakeholders would contribute to constructing the list of relevant services to be considered. Planners and managers could conduct a regional assessment of existing aquatic systems, evaluating the magnitude of services along each of the axes, or could create individual flowers for each ecosystem of interest. This exercise may reveal specific services that may be under-provided, thereby guiding future planning and design. A further extension of this could entail spatially explicit evaluation, similar to the “Healthy Waterways” program in Southeast Queensland, Australia (http://www.healthywaterways.org/index.html), which would reveal potential geographic inequalities and assist in regional planning.

In the next three sections of the chapter, we will describe the range of aquatic ecosystems found in desert lands, and provide our own qualitative assessment of the types of ecosystem services provided by these systems via flower diagrams. We begin with depicting extant, non-urban desert aquatic ecosystems, and then move to describing the types of hydrologic modifications that have occurred in the Phoenix metropolitan area. Many of these systems were designed decades ago, predating increases in ecological understanding of aridland streams and general public concern for our environment. Thus, only in retrospect we can see that some services are missing or under-provided. In the third section, we examine two urban aquatic “restoration” projects and compare their stated vs. actualized goals.

1.3 Hydrology and Ecological Characteristics of Desert Streams

Arid and semiarid lands make up approximately one-half of the earth’s terrestrial surface (Middleton and Thomas 1997). Regions of aridity occur on most continents and include both hot and cold deserts. In this chapter, we use the terms desert and aridland interchangeably to describe regions that receive less than 500 mm of precipitation annually. In hot deserts, high temperatures often result in evaporation exceeding precipitation, which limits surface water features. Although perennial streams are not as prevalent on the landscape in arid regions relative to mesic areas, aridland streams and rivers collectively constitute a predominant aquatic ecosystem given the large proportion of the earth's surface that is arid. Despite this prevalence, the ecology of desert aquatic ecosystems has been less studied than aquatic systems in other biomes (but see Kingsford 2006). At the same time, desertification (Schlesinger et al. 1990) and population growth in arid regions (Sutton and Day 2004) are contributing to even greater areal coverage of aridlands and increased pressure on their limited aquatic resources. A better understanding of these critical yet fragile ecosystems is needed to facilitate conservation in appropriate areas and potentially enhance the ecosystem services they provide is urban settings.

There are several different types of desert aquatic ecosystems and this is determined largely by hydrology. Desert streams may be perennial, intermittent, or ephemeral. Perennial streams flow year-round and groundwater is their primary water source. Intermittent streams do not maintain surface water flow during dry periods. While intermittent streams may be seasonally connected to groundwater, storms have a stronger influence on the timing of surface water flows. Ephemeral streams are disconnected from groundwater and flow only for short periods of time following storms. Depending on the underlying geomorphology, desert streams may alternate between gaining and losing reaches, creating some reaches that are intermittent or ephemeral even in streams that are generally perennial. Groundwater-fed wetlands, or *cienegas*, can also occur at upwelling locations (Hendrickson and Minckley 1985). In general, aridland catchments tend to have high drainage network densities (Figure 1.2), although connectivity is low during dry periods (Gregory and Walling 1973). Additionally, many desert streams have experienced changes in their hydrology due to non-urban human activities such as agricultural diversions of surface water and groundwater pumping, so that stretches that were once perennial are now intermittent (Webb and Leake 2006).

The extent of all streams extends beyond the wetted channel and encompasses the stream-riparian corridor. The stream-riparian corridor consists of surface water as well as the alluvial sediments beneath the stream bed and the land surrounding the stream that is significantly influenced by the stream. The riparian zone consists of the land beyond the stream channel and represents a transitional zone between the aquatic environment and upland desert environment (Naiman and Decamps 1997). In aridlands, riparian zones along desert streams often support high productivity, provide habitat for wildlife, and are important sites for nutrient cycling (Dahm et al. 1998, Germaine et al.1998, Baxter et al. 2005, Lite et al. 2005, Baird et al. 2005).

Desert streams and rivers are characterized by high interannual flow variability and thus some unpredictability in water availability. Hydrographs show from periods with little to no surface water flow and periods in which discharge is several orders of magnitude above baseflow. While many flash floods tend to occur within distinct wet seasons, it is not uncommon for desert streams to exhibit high seasonal and interannual variability in the timing of storms. Consequently, desert stream hydrographs tend to be much more variable relative to mesic systems (Sabo and Post 2008). The hydrological template of desert streams has a strong influence on the ecological characteristics of these streams. Organisms that can survive in desert stream systems must be able to withstand highly variable flow conditions, and have evolved different strategies to cope with the threat of desiccation and flood disturbances (Gray 1981, Grimm and Fisher 1989). Desert streams support productive and diverse algal and invertebrate communities. Algal communities include a myriad species of green filamentous algae, diatoms, and cyanobacteria. Algal species exhibit physiological adaptations (e.g., extracellular mucilage to increase water retention) to withstand drying and flooding by producing spores, cysts, and zygotes, and thus are able to rapidly recolonize following flooding (Grimm and Fisher 1989). Desert stream invertebrates also exhibit adaptive life history characteristics, including short development times, timing of emergence to occur prior to flood disturbance, and an aerial recolonization where eggs are deposited in sections of streams that may contain water for longer periods of time (Gray and Fisher 1981, Lytle 2002, Stanley et al. 1994). Desert streams support both native and non-native species of fish, whose adaptations include large reproductive efforts, short development times, and the ability to withstand low oxygen conditions (Meffe and Minckley 1987, Olden et al. 2008). Unlike mesic streams, the diversity of fish species in desert streams is quite low. Desert streams and adjacent riparian areas also support high vertebrate diversity (Soykan 2007). Many terrestrial vertebrates are dependent on desert streams, including several hundred species of birds, reptiles, and mammals ranging from bats to elephants (Kingsford et al. 2006).

In addition to supporting high biodiversity, desert streams and adjacent riparian areas may be important locations of nutrient cycling. Desert streams tend to be nitrogen limited, thus uptake of nitrogen by algae tends to be high (Grimm and Fisher 1986). Stream invertebrates also contribute to nitrogen cycling by excreting and recycling up to 70% of the inorganic nitrogen in desert streams (Grimm 1988). Rates of nutrient cycling in riparian areas tend to be temporally and spatially dynamic due to water availability and soil patchiness (Harms and Grimm 2008). Nutrients tend to accumulate in riparian plants and soils during dry periods. During storms, both surface and subsurface hydrologic flow paths may transport large quantities of particulate and dissolved nutrients from the uplands to the stream (Welter et al. 2005). This pulse of nutrient input results in rapid rates of nutrient cycling in desert streams. In fact, these short-term periods of rapid nutrient cycling may account for a significant proportion of nutrient cycling that occurs within desert streams and riparian areas on an annual time scale (Belnap et al. 2005).

The primary ecosystem services provided by natural desert streams potentially include wildlife habitat, sense of place (which includes existence value), water quality, aesthetics, and recreation. Because they are outside of the city extent and value of these services will vary considerably on the size, type, and location of the streams. For example, the San Pedro River, a perennial semiarid river in southeastern Arizona, has an extensive riparian gallery forest that supports over 400 species of birds, including over 250 migratory species, and over 80 mammal species. Thus, the San Pedro a popular place for birders and other naturalists, and there are several preserves in the area offering educational opportunities (The Nature Conservancy http://www.nature.org/initiatives/freshwater/work/sanpedroriver.html). Consequently, ecosystem services that are most appreciated by humans include wildlife habitat, aesthetics, and recreation opportunities for bird watchers (Fig. 1.3A). In contrast, Sycamore Creek is a much smaller, intermittent desert stream with a less extensive riparian gallery forest in central Arizona. Further, bird and mammal diversity is substantially lower in comparison to the San Pedro River. Wildlife habitat is still provided by Sycamore Creek but not to the extent as along the San Pedro River (Fig. 1.3B). The close proximity to Phoenix, however, contributes to Sycamore Creek's being a popular location for off-road vehicle use (http://www.fountainhillsguide.com/rec_offrd_sycam.html). Thus the primary ecosystem services provided by Sycamore Creek are recreation and sense of place.

1.4 Case Study: Water Features in the Phoenix, AZ Landscape

The unpredictability of desert hydrology creates problems for cities with respect to both water supply and flooding. Developing a secure, long-term water supply is perhaps the most pressing issue, especially considering predictions that some desert areas will become drier with more inter-annual variability in precipitation (US Bureau of Reclamation 2003, Seager et al. 2007). Once basic needs are met, numerous questions then arise about the allocation to various supplemental uses, such as landscaping, pools, creation and/or maintenance of recreation and aesthetic features, and support of natural systems within the city (also potentially for recreation and aesthetics). To manage flood risks, there are numerous options that span a range of hard- to soft- engineering management practices. Depending on design, aquatic systems have the potential to provide both basic necessities and enhanced quality of life via recreational and aesthetic properties. In arid systems, it may be especially to society's advantage to incorporate more than one function into these systems given limited water availability.

Located in the northern Sonoran Desert of the southwestern USA, the Phoenix metropolitan area receives approximately 180 mm of precipitation a year, with an average January temperature of 12º C and an average July temperature of 34º C (Baker et al. 2004). Most rain is concentrated in two seasons: a summer monsoon season with short, intense, localized thunderstorms and a winter rainy season characterized by frontal storms of longer duration and lower intensity. Given its hot, dry climate, the area experiences an average potential evapotranspiration of two meters annually. The city is situated in an alluvial valley surrounded by rugged mountain ranges typical of Basin and Range topography (Jacobs and Holway 2004). It sits at the confluence of two major rivers, the Salt and the Gila, and there are several other smaller tributaries and washes (Figure 1.4).

Phoenix is one of the most rapidly growing cities in the USA, increasing in population size from approximately 300,000 in 1950 to greater than 4 million inhabitants spanning more than 20 municipalities as of 2006. Models predict that by 2025, the population will exceed 6 million, representing a 280% increase since 1980 (Jacobs and Holway 2004), and nearly all of the currently undisturbed and agricultural lands within the metropolitan area will be developed into urban land uses (Jenerette and Wu 2001). With few geographical barriers to expansion, growth has been largely in an outward direction, estimated at approximately 0.8 km per year (Gober and Burns 2002). Most new construction has been the result of conversion of agricultural to residential use but, increasingly, housing developments are built in native desert areas.

Water supply and distribution

Though the climate of the greater Phoenix metropolitan area is distinctly arid, surface water, some of which is imported great distances, constitutes approximately half the water supply for the burgeoning population (Arizona Department of Water Resources http://www.azwater.gov/dwr/). To meet the water demands of greater Phoenix, and throughout much of Arizona, surface water is collected in reservoirs and transported to end users through extensive delivery systems. Because of the tremendous demand for water resources exerted by the growing population, agriculture, and, to a lesser extent, industry, much of the available surface water is used in its entirety, and formerly perennial systems often flow only during flood conditions. The Gila River is largely dewatered for municipal and mostly agricultural purposes before reaching Phoenix. Immediately upstream of Phoenix, the Salt and Verde Rivers are impounded in six reservoirs with a combined storage capacity of 2.8 x 10^9 m^3 (Gooch et al. 2007), and the entire flow of these rivers is appropriated for municipal and agricultural purposes. Water originating from the Salt River's 34,000-km^2 catchment comprises a substantial portion of the available surface water, and the Tonto National Forest was established in large part to protect this critical resource.

Central Arizona's portfolio of surface water was given a considerable boost in 1985 with the opening of the Central Arizona Project (CAP) canal, which pumps Colorado River water (uphill) from Lake Havasu in western Arizona to Phoenix and, ultimately, to Tucson via a 554 km concrete-lined canal. The CAP canal is designed to deliver 51% of Arizona's 3.4 x 10^9 m^3 allotment of Colorado River water afforded by the Colorado River Compact (Jacobs and Holway 2004). Water from the CAP canal may be stored temporarily, depending on season and demand, behind Waddell Dam, which also impounds the Agua Fria River, another north-south trending desert river that contributes to the surface-water portfolio of greater Phoenix.

Corresponding to the tremendous water-storing capacity of the reservoirs is an equally impressive water-delivery system that moves stored water to where it is needed. CAP canal water and surface water collected in reservoirs along the Salt and Verde Rivers is delivered to and distributed throughout the greater Phoenix area by way or more than 2,100 km of canals (Gooch et al. 2007). There is a long history of modifying river flows in the Phoenix area for anthropogenic purposes, and segments of the canal infrastructure in place today follow canal paths established by the ancient Hohokam Civilization of ca. 500-1400 A.D. (Fitzhugh and Richter, 2004). Given the extensive development of water resources for the Phoenix and Tucson metropolitan areas, these canal systems now compromise a considerable portion of lotic habitat in central Arizona (Marsh and Minckley 1982). In the early 1900s, the canals were clay-lined, flanked by large cottonwood trees, and considered desert oases (Wenk 2002). However, the canals were lined with concrete and the bank-side trees were filled in the 1950s in response to safety concerns, maintenance considerations, and a general unwillingness to share water with the riparian flora (Wenk 2002). In recent years, as municipal demand for water usurps agriculture (historically the largest water user in central Arizona), initiatives are emerging that would incorporate the canals into parks, trail systems, water-front property, and other civic-minded features (Wenk 2002, Gooch et al. 2007).

Stormwater removal

Although Phoenix receives an average of only 180 mm of rain annually, precipitation is concentrated in two wet seasons occurring in the winter and summer. This means that despite a generally dry climate, stormwater systems must be designed to accommodate the large flows that these brief but intense storms generate. Streets, large channels, natural rivers (i.e. Salt River, New River, Agua Fria), and floodplains (Indian Bend Wash -- IBW) constitute the major stormwater systems in the Phoenix metropolitan area. Because of these large flows much urban runoff is diverted to large flumes which direct flows to large retention basins or natural river beds. These large concrete flumes provide the important service of flood control but provide little in the way of other services such as aesthetics, recreation, wildlife habitat, or water quality. In 2002 the Flood Control District of Maricopa County, which oversees all major flood control projects, explicitly added to its mission and goals the multi-use, aesthetic qualities of stormwater designs (FCDMC 2002).

Since 1985, stormwater projects have been planned within Area Drainage

Master Plans, developed by municipal engineering departments for each watershed, which provide the minimum criteria and standards (for flood control and drainage) for land use and development. Drainage regulations of Maricopa County apply to all development of land and conditions that may affect drainage systems and patterns; that is, no development is allowed to create an increase in the peak discharge, volume or velocity of runoff or change the point of entry of drainage onto other property during storms. Additionally, designs must conform to Best Management Practices to control erosion and sediment transport (usually with grass, concrete and/or rip rap). Since most of the Phoenix area is relatively flat, a common stormwater feature is the retention basin. These basins must have the capacity to hold a 2-hour/100-year flood. The landscaping and use of these basins is open. Some are xeriscaped, i.e., planted with drought-tolerant species (although these are often drip-irrigated) and covered with non-organic mulch. Others are grassy, and thus are irrigated and often fertilized as well (Figure 1.5).

The poster child for multi-use stormwater management is the Indian Bend Wash (Figure 1.6), developed on a small tributary of the Salt River (~500 km^2 watershed area). After a devastating flood in 1972, the City of Scottsdale teamed up with the FCDMC and Army Corps of Engineers to create a greenbelt capable of containing a 100 year flood (Matthews 1985). Although the primary goal of the IBW flood control project was to protect the city of Scottsdale from the severe flash flooding that is a characteristic feature of southwestern streams (Baker 1977), the flood plain was also specifically designed for recreational and aesthetic purposes and contains artificial lakes, irrigated turf floodplains, recreational trails, sports fields, picnic ramadas, and tennis courts (Roach et al. 2008). A recreational path meanders up the belt, and a series of lakes provide fishing opportunities for residents. Additionally, this large swath of green land may provide some relief from the urban heat island effect (Ca et al. 1998).

IBW has been designed to function as a floodplain capable of handling a 100-year storm; however, in order to fulfill this role, as well as serve recreational amenities to the community, it is a very different stream than it once was. The floodplain is dominated by turf and other vegetation that depends on irrigation, and the lakes are connected to a groundwater pumping system that feeds them with a continuous water supply. Furthermore, this urban stream is functionally very different from its natural counterpart due to changes in geomorphological structure and groundwater-surface flow dynamics (Grimm et al. 2004, Roach and Grimm 2010).

IBW was named one of the 10 most outstanding engineering projects in 1974 (Matthews 1985). Although it bears little resemblance to a past state, it is a functioning ecosystem capable of providing services to the surrounding community (Roach 2005). A simple internet search on this wash in Scottsdale reveals a number of sites dedicated to advertising the area as a hot spot for rollerblading, jogging, bike riding, etc., with comments from residents claiming the area is “great for viewing the beauty of the AZ desert when you add water” (http://gocitykids.parentsconnect.com).

The success of this system at offering an enjoyable recreational area has not prevented its providing some ecological services to the community as well. Research has shown that this area is a hot spot for processing nitrogen (Roach and Grimm *In review*). Very high denitrification rates characterize IBW, which can reduce nitrogen loads to receiving waters. Although the massive influx of N from groundwater and fertilizer to this floodplain-lake-stream complex may be overwhelming the its N-removal mechanisms, this project is evidence that designed ecosystems can provide many benefits to urban residents even when "natural" functioning has not been restored (Grimm et al. 2004, Roach and Grimm *In review*).

Not all desert washes within the area have been eliminated. Some washes are more heavily designed than others (Figure 1.7). Currently, we do not know the frequency and distribution of these different types of designs for stormwater systems. Although water quantity and quality are monitored by the FCDMC and municipalities for National Pollutant Discharge Elimination System permit requirements, work on the ecological functioning of these diverse stormwater systems is in its nascence. To date, there has been work on the biogeochemistry of the artificial lakes, streams, and floodplain of Indian Bend Wash (Roach and Grimm *In review*), and of retention basins (Zhu et al. 2004, Chapters 2 and 4).

Wastewater removal

Most of the waste generated in the Phoenix metropolitan area is treated by the two largest treatment plants (91st Avenue and 23rd Avenue). In contrast to cities in wetter climates, effluent is not discharged to perennial rivers. Due to the scarcity of water in the Phoenix area, most treated waste water is reused or recharged to groundwater aquifers. Effluent is used to support a diversity of ecosystems, all designed to provide a wide range of services, from food production and water quality and storage to recreation and wildlife habitat (Table 1.2). During peak growing season, most effluent is diverted via the Buckeye Canal to be used for agricultural irrigation. Effluent is also used, to a lesser degree, to irrigate golf courses and parks and to fill small lakes. Some developers have even built small reclamation plants within their projects to treat water for golf courses and lakes. A small portion is used to recharge groundwater through dry stream beds, such as the Agua Fria River (Greely and Hansen Engineers 1998). The town of Gilbert has created a unique designed ecosystem with its effluent, the Riparian Preserve at Water Ranch. Here, all of the effluent from the town is recharged through 28 ha of recharge basins. The water supports a range of plant communities, including marshlands, riparian and upland vegetation areas, which in turn support a diversity of birds, insects and amphibians. The Preserve also functions as a recreational park, with trails and a lake which is designated as an urban fishing resource (http://www.riparianinstitute.org).

When irrigation demand is low, effluent is discharged to the Salt River (Greely and Hansen Engineers 1998). Stricter water quality regulations by the Arizona Department of Environmental Quality for discharges to waterways in 1990 sparked the development of the Tres Rios project, discussed in detail in the next section. In 2000, a larger restoration design was approved by Congress, and will include 194 ha of emergent wetlands and a 6.8 km-long levee for flood control.

Aesthetic value and recreation

The Salt River, the major natural water feature in the Phoenix metropolitan area, was dry due to damming and diversion for much of the past century, thus residents have mostly relied on artificial water bodies for cooling, aesthetics, and recreation. As mentioned above, the first few decades of the resurrection and expansion of the pre-historic canals, many of them were tree-lined and provided public places to meet, retreat from the heat, and even swim (Baggetta 2004, Yabes et al. 1997). However, in the 1950's the trees were removed from the canal banks to reduce water loss due to transpiration, and people relied more on air conditioning in their homes for cooling and private pools for recreation. There has been a recent increase in the appreciation of native rivers, riparian areas and washes, as evidenced by the inclusion of recreation and aesthetics as state goals of the restoration project described below.

Lakes are another type of artificial aquatic system commonly found in the Phoenix area. Since the early 20th century, Phoenix boosters have touted the area as an oasis in the desert (Gober 2006), known not only for its lush, sometimes tropical vegetation, but an abundance of golf courses as well, which include the requisite water hazards. Currently there are >150 golf courses in the valley. To further bolster claims of desert oases, many housing and commercial developments have built lakes as well. Many municipalities have created lakes within city parks, and the Arizona Game and Fish department runs an urban fishing program, stocking 21 lakes in 11 valley cities with trout, catfish, largemouth bass, and sunfish (http://www.azgfd.gov/h_f/urban_fishing.shtml).

In order to keep all of these lakes filled, owners and managers previously relied on ground water, or, in the case of agricultural lands newly converted to residential, excess surface water rights. However, serious over-drafting of the valley aquifer resulted in the state passing the “Lakes Bill” in 1987, which prevented the creation of any new lakes that relied on groundwater (ARS 1987). Most of the surface water rights have already been appropriated, so newly created lakes must rely on treated effluent. It is estimated that, since 1987, approximately 180 new lakes have been created using this water source (Mullins, pers. comm.). Currently, the total estimate for the Phoenix valley is approximately 1000 lakes (See Chapter 2).

These artificial aquatic bodies exist in a nebulous area with respect to environmental regulation and ecological understanding. The Arizona Department of Environmental Quality (ADEQ) does not even have a complete list of all water bodies and “although not included in the surface water definition, ADEQ is interested in tracking water quality data for ‘urban lakes’ or man-made lakes created for recreational purposes,” but has no program underway to collect these data itself (ADEQ 2004). Many of the lakes are lined to prevent loss to groundwater, and some are dredged periodically to prevent sediment build-up. Additionally, many of the lakes are treated with algaecides to improve clarity, and have aerators and fountains (Figure 1.8).

1.5 Examples of Restoration in the Phoenix Metropolitan Area

Restoration-based agendas are highly influenced by societal needs and values, and thus these are reflected in their design. The National River Restoration Synthesis (2006) shows a majority of restoration projects in Arizona were dedicated to riparian management, with a high concentration of projects focusing on water quality management and flow modification (Table 1.3). Although there were a low number of projects with their primary intent as aesthetics/recreation/education, many of these projects had more than one objective. Below we describe two case studies of restoration specifically in the Phoenix metropolitan area, and although the primary objectives of these were to enhance riparian areas, treat wastewater effluent, or manage large flood events, the secondary goal in each case was to increase aesthetic value and recreation opportunities. These examples in Phoenix have been labeled “restoration” projects, yet in their final form their structure and function is different from a typical desert stream.

The Salt River was once a perennial river that supported an extensive riparian ecosystem. However, in the early 1900’s the U.S. Bureau of Reclamation began constructing several dams and reservoirs along the Salt River to provide a steady, year-round water supply. These dams and reservoirs also provided protection from floods and hydropower to a growing Phoenix metropolitan area. These hydrological alterations to the Salt River were necessary to support the economic development of the Phoenix metropolitan area, but left behind a dry river and devastated the riparian habitat along the river. In the past twenty years, the City of Phoenix, in partnership with the U.S. Army Corps of Engineers and the Maricopa County Flood Control District initiated two major restoration projects along the Salt River: the Rio Salado Habitat Restoration Project (hereafter, Rio Salado Project) and the Tres Rios Constructed Demonstration Wetlands Project (hereafter, Tres Rios Project).

The primary goal of the Rio Salado Project (Figure 1.9) was to restore native riparian and wetland vegetation along an 8-km (241 ha) section of the Salt River through Phoenix, Arizona (City of Phoenix 2007). A major component of the Rio Salado Project was to plant native cottonwood (*Populus fremontii*) and willow (*Salix goodings*) along the banks of the Salt River. So far, 17 ha of cottonwood-willow habitat have been planted (City of Phoenix 2007). Additional habitats that have been created include 56 ha of mesquite bosque habitat, 32 ha of saltbush (*Atriplex spp.*), quailbush (*Atriplex spp.*), and burrobush (*Hymenoclea salsola*), 26 ha of Lower Sonoran Desert habitat, including palo verde (*Cercidium spp.*) and mesquite (*Prospois spp.*), 21 ha of aquatic strand, 6 ha of wetland marsh, and 81 ha of open space (City of Phoenix 2007). While the Rio Salado Project has successfully created several acres of riparian habitat and returned flows to previously dry sections of the Salt River, the naturally flashy hydrological regime has not been restored. Thus, in order to maintain the planted riparian habitat, 5 wells were built which provide the main source of water to the created habitat. Pumps draw groundwater, and pipes and canals distribute the water to different areas of the Rio Salado Project. Storm drains are also used to redirect runoff to the habitat restoration areas.

The Rio Salado Project is not an example of ecosystem restoration, but rather an example of habitat design. The project does provide multiple ecosystem services, including habitat for wildlife, high biodiversity, and flood control. As a result of habitat creation and high biodiversity, the Rio Salado Habitat Restoration Area has become a hotspot for bird watching and wildlife photography. This is enhanced by an ongoing environmental education program and a 16-km trail system with interpretive signs. There are also recreational opportunities provided by the Rio Salado Project ranging from biking, jogging, hiking, horseback riding, and picnicking. Due to the recent completion of the Rio Salado Project, scientific studies evaluating the ecosystems services provided by the habitat restoration project have not been completed. However, since the Rio Salado Habitat Area opened to the public in November 2005, the area has proved to be a success in terms of providing recreation opportunities, improving the aesthetics of the Salt River, and contributing to downtown Phoenix revitalization.

Tres Rios Project is located at the confluence of the Salt, Gila, and Agua Fria Rivers, downstream of Rio Salado Project. The initial goal of the Tres Rios Project was to determine whether the constructed wetlands, in an area approximately 14 km long by 1.5 km wide mile wide, are able to treat effluent from a waste water treatment plant and meet discharge levels and National Pollutant Discharge Elimination System permit requirements. Secondary goals included providing wildlife habitat, recreational areas, and environmental education.

The project created perennial wetlands at two sites: a former agricultural field on a terrace adjacent to the Salt River, which was subject to flooding only during flows in excess of 100-year floods, and in the channel of the Salt River. Restoration included installation of sedges, primarily bulrush (*Schoenoplectus spp*.) and trees, such as cottonwood and willow, and removal of nonnative saltcedar (*Tamarix ramosissima*). Public facilities such as recreational trails, picnic areas, bird blinds, interpretive signs, and a butterfly garden have also been constructed.

Similar to the Rio Salado, the Tres Rios Project is an example of a designed ecosystem, rather than a restored ecosystem, providing key services by enabling wastewater removal, creating habitat for wildlife, and providing opportunities for recreation and education. In terms of meeting the primary project goals, Tres Rios Project receives over 7 million liters of advanced treated municipal wastewater daily. A number of water quality parameters are monitored, including total nitrogen and dissolved oxygen. Denitrifying conditions were established within one year of construction and total nitrogen exiting the system is in the range of 1.5- 2.5 mg/L (http://ufdp.dri.edu/projects/tresrios.htm). A comparison of inlet and outlet water indicates that Tres Rios Project reduces concentrations of hyrdophobic organic compounds (HOC), herbicides, pesticides, and other organic wastewater contaminants by 40-99% (Barber et al, 2006). Concurrently, accumulation of HOC and trace metals has been observed in fish collected from Tres Rios Project. With respect to secondary goals, Tres Rios Project is home to a variety of wildlife, including mammals, a variety of migratory and non-migratory birds, fish, amphibians, and insects; however, in some case the populations of certain species have reached undesirable levels (www.phoenix.gov/TRESRIOS/research.html). The growing beaver population is being managed with trapping and relocation while the mosquito population is being addressed by reconfiguring vegetation, introducing larvivorous fish, and applying pesticides. Lastly, the Tres Rios Project is available for public use, hosting an annual nature festival and offering bird and nature walks, group tours, and other educational programs.

Projects that are not resilient on their own and do not restore natural hydrological regimes are not representative of true restoration projects (Middleton 2002). In arid cities "true" restoration may be difficult, if not impossible, but there is value in designing new environments in these areas in order to provide ecosystem services to society. Here we presented three case studies from the Phoenix metropolitan area, each representing a project that was labeled as restoration but was actually an example of design. In each case, true habitat restoration was not possible due to major water diversions that significantly altered the structure and function of the former system. Instead, these systems were designed to replicate selected ecosystem services that natural desert streams can provide, including riparian habitat and high biodiversity. In addition, these designed streams provide ecosystem services that the pre-settlement streams did not provide. These services include retention and removal of stormwater and treatment of effluent generated by the residents of Phoenix and recreational amenities. Indeed, the secondary goal of recreation is perhaps the service that is most enjoyed by most residents of the Phoenix metropolitan area for each case study.

1.6 Summary and Future Considerations

Although the “restoration projects” described here fail to be successful restorations in the sense that they *do not* recreate the natural hydrology of the river/riparian system, they are successful designed ecosystems that differ in fundamental ways from the natural desert streams they mimic. In the two cases where the design intent is to retain water, they do not restore the natural system’s hydrology and have more stable hydrographs. The ‘success’ of these projects is often weighted toward the more diffuse aesthetic improvements they provide, and for many people, that equates to the presence of water – a condition many native desert aquatic systems cannot consistently provide. Moreover, ecosystem properties of native desert streams—their biota, nutrient cycling characteristics, and highly dynamic processes—are just as much a product of the flashy desert stream hydrograph as they are of the ample sunlight and warm temperatures. Thus, a stabilized hydrograph is unlikely to support the same structures and processes. In those urban streams where flooding is allowed to occur, such as IBW, the changes in upland catchment size and land cover reduce sediment inputs, and channel modifications prevent channel migration, such that the impact of these events differs dramatically from that of native streams (Roach et al. 2008). Thus, the case studies suggest that while recreating natural hydrology may be the pinnacle of restoration it may be neither necessary, nor possible, nor sufficient for successful ecosystem design in arid-land cities, where water storage and recreational usage are highly desirable ecosystem services.

All cities, regardless of climate, must establish systems for water delivery, and storm and wastewater removal. Most cities also develop aesthetic and recreational features, often associated with water. These municipal systems each provide a suite of ecosystem and engineering services (Table 1.1, Figure 1.1) to the citizenry. Some of these services are universal (i.e., safe, reliable water supply; wastewater treatment), while others may be more important for arid-land cities like Phoenix (i.e., saving water; providing 'oases'). Our evaluation of aquatic restoration projects and management practices in the Phoenix metropolitan area reveals a range of designed aquatic systems that provide several types of ecosystem services. Many of these do not map directly onto the ecosystem services that could be provided by pristine streams in the native desert. Natural desert aquatic systems accommodate some of the requisite services (nutrient retention, habitat provision) that cities need; however, they may not be able to do so at high population levels. And, due the typical intermittent flow in many desert streams, these systems in fact, may not provide some ecosystem services needed by urban residents.

The designed ecosystem projects described here represent uncharted territory for researchers to understand their structure and function as ecosystems. The innovative, early project at IBW (constructed in the late 1960's) was conceived, designed, constructed, and implemented without ecological monitoring that could reveal the efficacy of the design. The projects on the Salt River are still quite new and no research describes how they function either mechanistically or systemically. Based on new biogeochemical research showing that IBW floodplains and lakes are hot spots for nitrogen processing (Roach 2005), it appears that the projects actually may have exceeded their ‘design specifications’ with respect to providing ecosystem services.

For future hydrologic modifications, in Phoenix or elsewhere, we suggest that an informed ecological design approach would be the most appropriate. In this scenario, planners would consult with various stakeholders to determine their needs and desires. Systems could then be designed to accommodate as many of these as possible, while recognizing that trade-offs will occur. Visualizations, such as the conceptual framework from Foley et al. (2005), can be used both to assess existing systems and as a planning tool for proposed projects (Figure 1.1). The ecosystem services included are by no means exhaustive, and different communities may develop their own sets of services that are necessary and desirable. This first step in the planning process differs from traditional approaches in its focus on ecosystem services.

Ecologically informed designs next should incorporate a monitoring or research element, enabling managers to assess whether aspects of design are effective or not in terms of the agreed-upon goals and services (e.g., Table 1.1). In turn, research results can provide information for designers and engineers that will create the next generation of urban aquatic systems. Accordingly, the final step in creating effective designed aquatic systems is to change what has proved ineffective and retain those features that deliver the desired services. Accomplishing this will require flexibility in design and in the institutions that govern water delivery and removal systems. The potential to maximize both engineering services and ecosystem services through interdisciplinary dialog is high if the appropriate conversations can be started. Water in arid-land cities is a limiting resource at many levels; coordinated ecosystem design (as opposed to "restoration") may actually allow the creation of aquatic systems that are multi-purpose and sustainable from the perspective of water managers, city planners and ecologists alike.

1.7 Introduction to this Dissertation

The current chapter was the product of a collaborative effort stemming from a conference held at the Cary Institute of Ecosystem Studies on urban design and ecology. Co-authors include Stevan Earl, Elizabeth Hagen, Rebecca Hale, Hilairy Hartnett, Michelle McCrackin, Melissa McHale, and Nancy Grimm. In the research presented in Chapters 2 – 5, I explore several dimensions of existing designed aquatic ecosystems in the Phoenix metropolitan area. This dissertation builds upon the ideas presented in this introductory chapter, which is in review for a symposium volume on resilience in urban design (Larson et al. *In review*; see also Table 6.1). I wanted to (1) evaluate the extent and distribution of these types of ecosystems, (2) assess their functioning with respect to nitrogen biogeochemistry, (3) investigate the potential economic value of the ecosystem services they provide, and (4) examine synergies and tradeoffs between the ecological functions that underlie ecosystem services. Below, I present brief descriptions of the research undertaken to address these objectives, which correspond to chapters 2 – 5 of this dissertation. In Chapter 6, I present a synthesis of the dissertation in the form of brief conclusions

Chapter 2: Small-scale and extensive hydrogeomorphic modification and water redistribution in a desert city and implications for regional nitrogen removal

This chapter describes the dramatic changes to local hydrology in the Phoenix metropolitan area as demonstrated by the construction of artificial lakes and stormwater retention basins. By collating GIS data from several sources, I was able to estimate that there are approximately 1000 human-made lakes (excluding filled in gravel pits along the Salt River channel) and thousands of stormwater retention basins. I was further able to delineate lakes by source water and retention basins by surrounding land use. Basic data from field surveys on N stocks and potential denitrification in these ecosystems are discussed within the context of the regional N budget.

Chapter 3: The Value of Water-Related Amenities in an Arid City: The Case of the Phoenix Metropolitan Area

This chapter was a collaborative effort between myself and Charles Perrings, and fulfills the Integrative Graduate Education and Research Training (IGERT) in Urban Ecology requirement of a collaborative dissertation chapter. We used an economic model to explore the value of water-related amenities in the region. Using residential sales data from the Maricopa County Assessor, environmental and locational data collected by the Central Arizona – Phoenix Long Term Ecological Research project (CAP LTER) and others, we evaluate the value of non-marketed ecosystem services and amenities by supposing that they derive from value of the ‘product’ they help create. We applied a hedonic model, using the sales data as the marketed asset that is composed of numerous attributes, including non-marketed services or disservices. Using these data together with data on residential property sales we estimated the implicit marginal willingness to pay for a range of environmental and locational attributes associated with differing forms of water use. We performed this analysis for the whole CAP LTER region, and then segmented by city to assess potential differences in sub-markets. Finally, we conducted a “second stage” analysis to estimate demand curves for selected ecosystem services, making it possible to predict the implications of changes in ecosystem service ‘prices’ for the quantity demanded of those services.

Chapter 4: Landscape Design and the Fate of Nitrate in Stormwater Retention Basins in the Phoenix, AZ Metropolitan Area

This chapter is an in-depth investigation of the N biogeochemistry of stormwater retention basins. A common feature in the metropolitan area (see Chapter 2), there are two basic landscape designs: grassy and xeriscaped. I conducted a survey of 32 basins (16 of each kind) to characterize bulk soil properties and denitrification enzyme activity (DEA, or potential denitrification). The presence of grassy and therefore the requisite irrigation create significant differences between basins by landscape design. I then conducted a more in-depth study of 10 retention basins (5 of each kind) to investigate potential N storage in soils at depth or transmission to groundwater. It appears that water and potentially N are transported below depths of 50 – 60 cm, but in the grassy basins there is rapid removal of N inputs. Xeriscaped basins have high but variable inorganic N concentrations in the top layer of soil, while the grassy basins are lower and more constrained. To further explore the differences between basin landscape design with respect to N retention and export, I conducted an experimental manipulation, flooding two basins (one of each kind) to simulate a 5-year storm. The experimental flood included the addition of nitrate (NO_3^-), a portion of which was labeled with ^{15}N isotope to allow us to trace the transformation on NO_3^- through the ecosystem. The data suggest that, while both basins export iN through infiltration and gaseous losses, the relative proportion differs by landscape design, with the grassy basin having a greater proportion loss via gas flux than infiltration. Grassy basins may be providing the additional ecosystem service of N removal, but they do so only with the associated costs of irrigation and maintenance.

Chapter 5: Locally Provided Hydrologic Ecosystem Services in the Phoenix, Metropolitan Area: Potential Bundles and Tradeoffs

A common theme throughout this research has been to identify types of designed aquatic ecosystems, characterize the distribution and ecological function, and estimate the potential economic value of water-related ecosystem services. In this final chapter, we expand our spatial analysis to include other proxies for hydrologic ecosystem services, such as golf courses, agricultural irrigation, groundwater pumping, etc. We evaluated the spatial clustering of individual proxies as well as the level of coincidence and interaction between proxies. Our results showed both significant positive and negative interactions between services, and these were in part dependent on the scale of analysis. We found that, for some services, provisioning was constrained by the hydrogeomorphology of the area, while for others human engineering and the creation of designed ecosystems has enabled the delivery of hydrologic ecosystem services independent of natural constraints. Application of this type of framework for the analysis of bundles of and tradeoffs between ecosystem services can help identify key areas of the landscape that require consideration in future planning and management efforts.

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Table 1: Types of urban aquatic systems and the services they provide.

System	**Ecosystem/engineering services needed**
Water delivery systems	• Deliver water reliably to places where it is used • Ensure that water is clean and safe to drink
Stormwater removal systems	• Ensure safety of property and human life • Reduce sediment transport/erosion • Reduce or eliminate loading to downstream systems • Save water (i.e., reduce/eliminate downstream losses)
Wastewater removal	• Protect public health • Reduce downstream loading • Enable wastewater treatment (collect, treat, release) • Save water
Aesthetic/cultural features	• Replicate natural aquatic ecosystems of region • Provide habitat for native species • Preserve biodiversity • Provide a sense of place • Provide opportunity for recreation • Provide opportunity for education

Table 1.2: Ecosystem services provided by treated wastewater effluent

Ecosystem Service	Example
Food production	Agricultural irrigation
Water quality	Constructed wetlands at Tres Rios
Water quantity	Recharge basins at Gilbert Riparian Preserve
Recreation	Gilbert Riparian Preserve trails, fishing; Golf course irrigation
Wildlife Habitat	Constructed wetlands at Tres Rios

Table 1.3. Restoration Projects in Arizona. (Adapted from National Biological Information Infrastructure, National River Restoration Science Synthesis 2006)

Project Intent	Projects	% of Total	Total Costs	Median Cost
Aesthetics/Rec/Education	9	4	$30,030,740	$28,807
Bank Stabilization	11	4	$26,921,671	$87,800
Channel Reconfiguration	16	6	$5,598,645.77	$256,923
Dam Removal/Retrofit	0	0	$0	$0
Fish Passage	4	2	$17,564,672	$8,782,336
Floodplain Reconnection	1	0	$61,951	$61,951
Flow Modification	25	10	$126,023,488	$253,584
In-stream Habitat	11	4	$3,974,720	$54,895
In-stream Species	11	4	$486,510	$30,650
Land Acquisition	6	2	$1,198,488	$510,284
Riparian Management	81	33	$12,587,814	$82,561
Stormwater Management	0	0	$0	$0
Water Quality Management	61	24	$26,822,968	$88,450
Other	13	5	$4,167,019	$83,432
Total for all projects	249	100	$255,438,689	$10,321,674

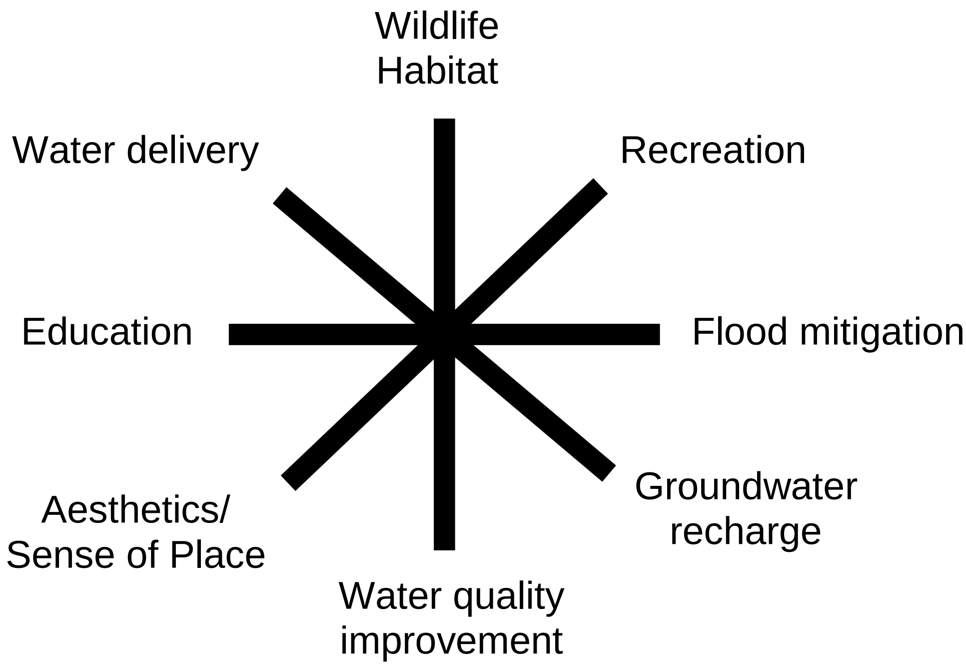

Figure 1.1: Conceptual framework, after Foley et al. 2005, showing the “flower” of potential ecosystem services in aquatic habitats.

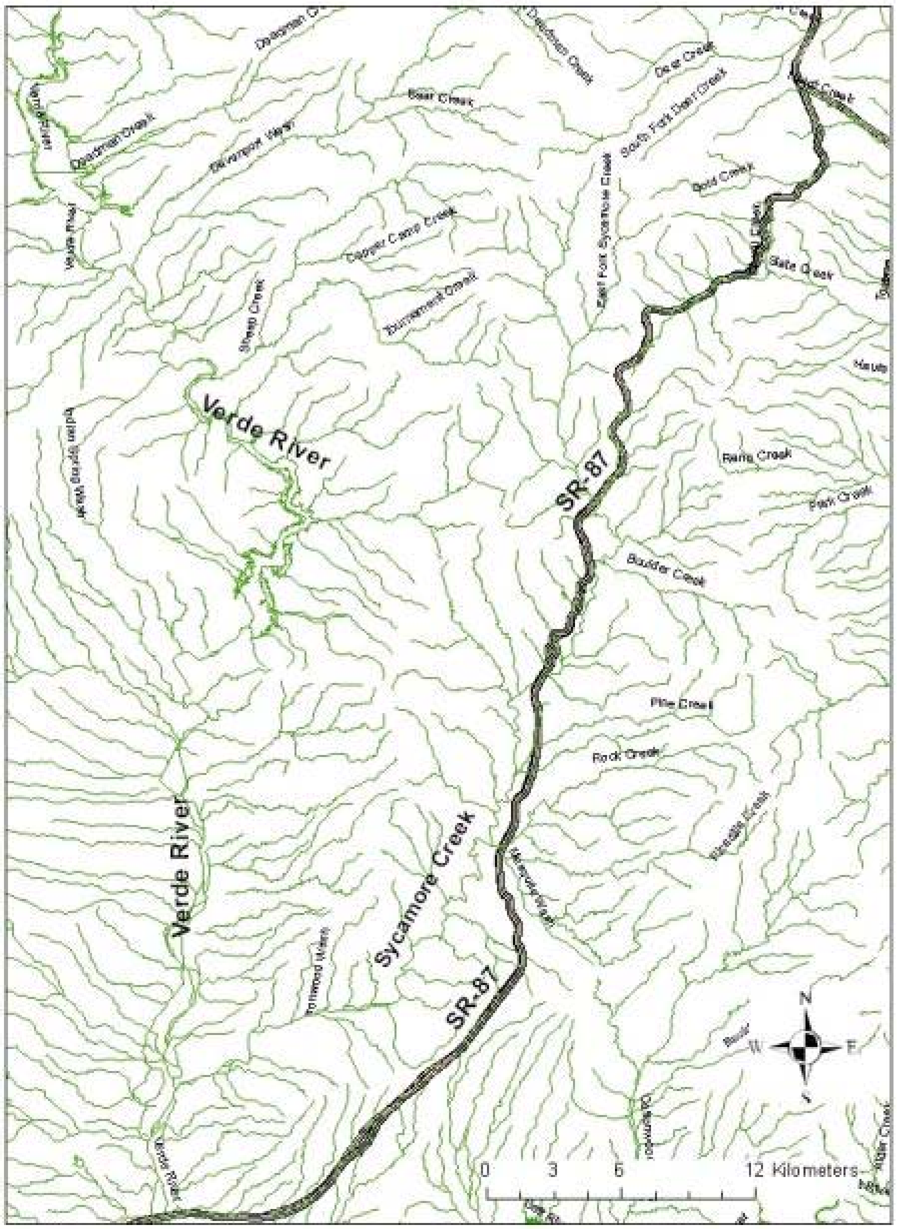

Figure 1.2: Map of the Sycamore Creek catchment, northeast of Phoenix, AZ, exhibiting the high drainage density common in desert watersheds.

Figure 1.3: A) The San Pedro River and B) Sycamore Creek, both in Arizona.

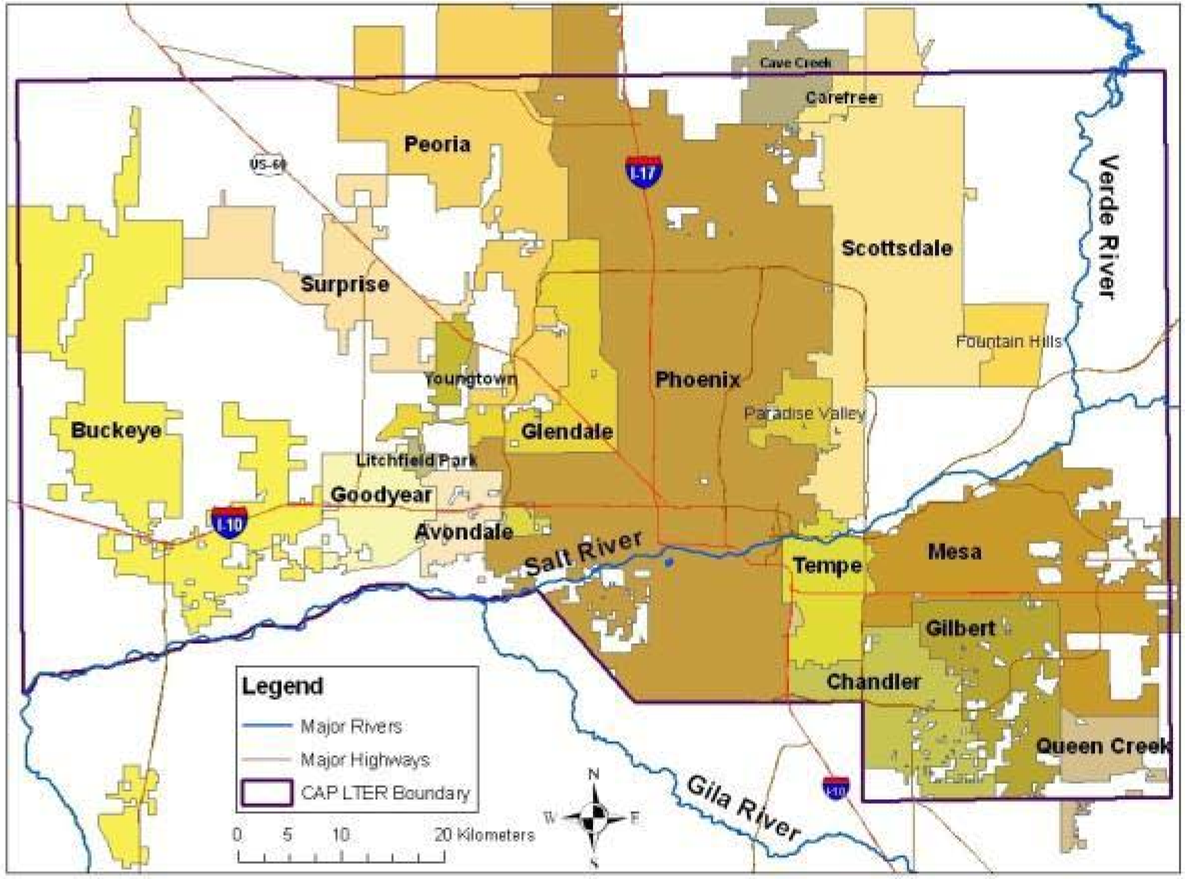

Figure 1.4. Map of the Phoenix metropolitan area

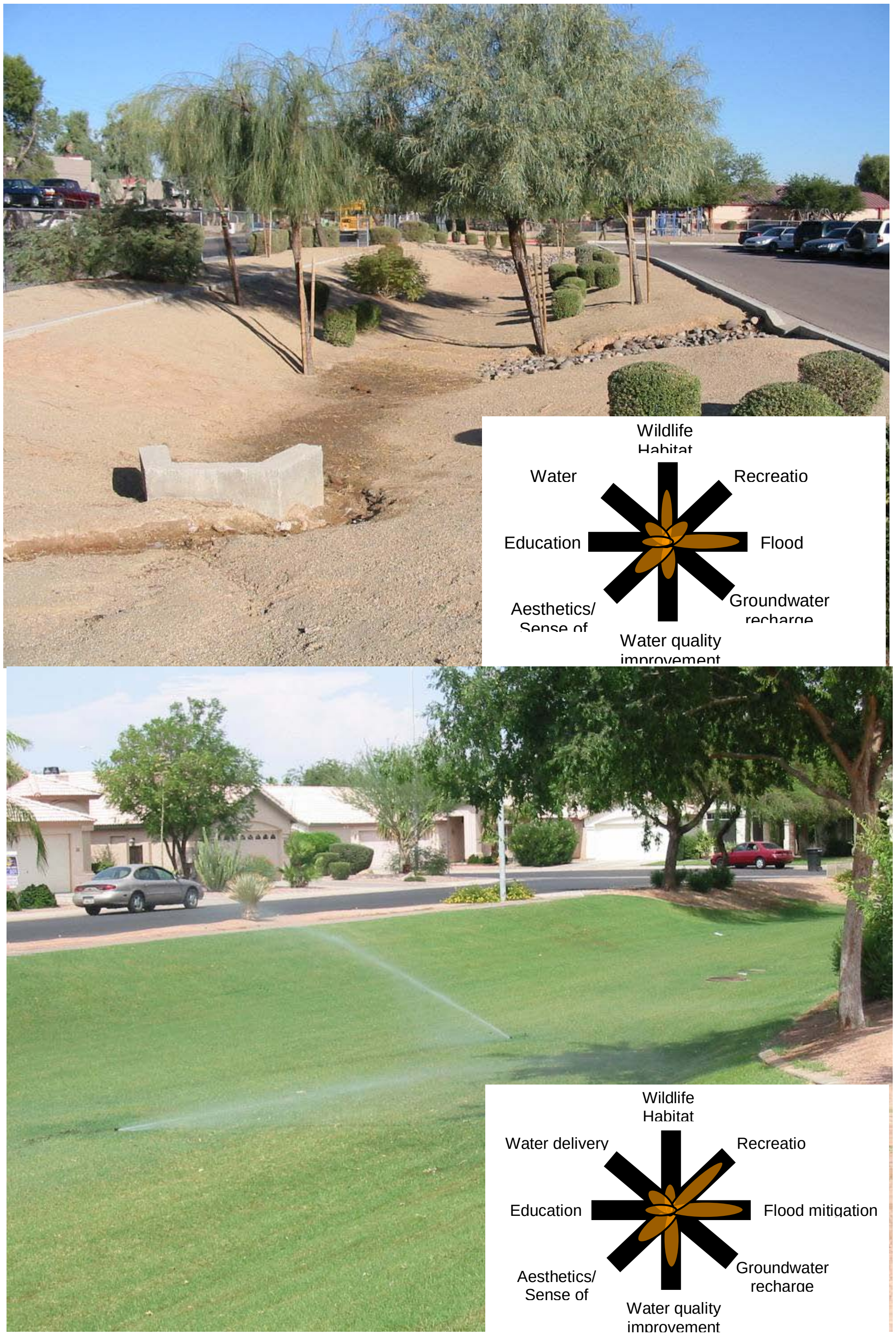

Figure 1.5: Examples of xeriscaped and grassy stormwater retention basins.

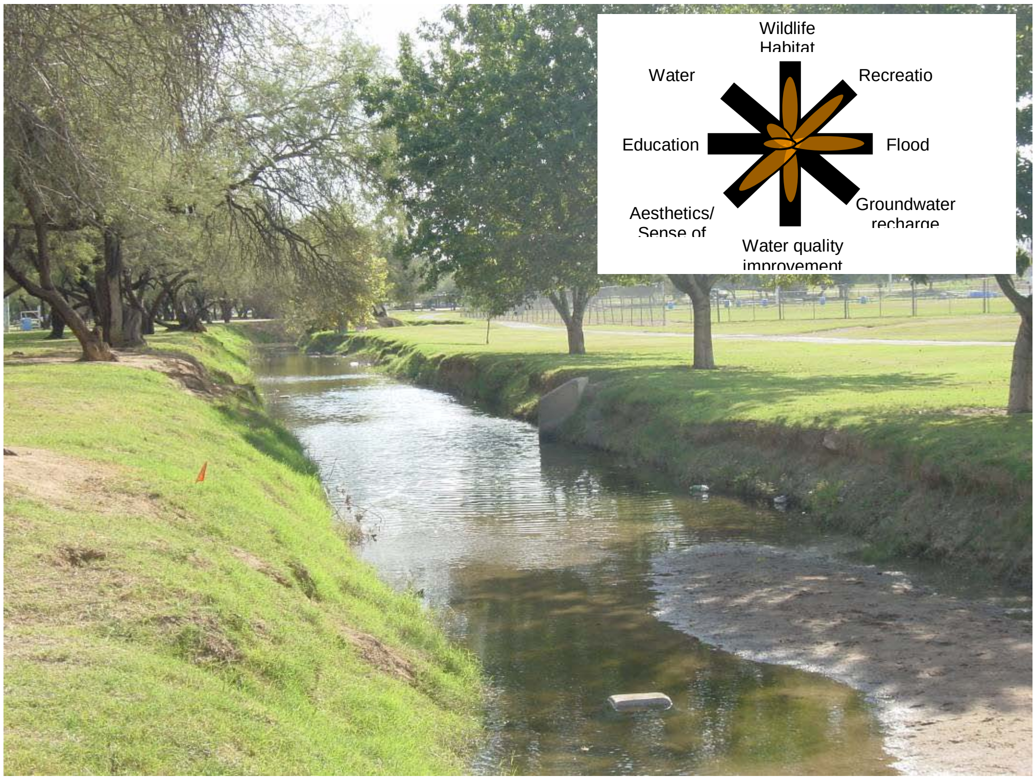

Figure 1.6: Indian Bend Wash

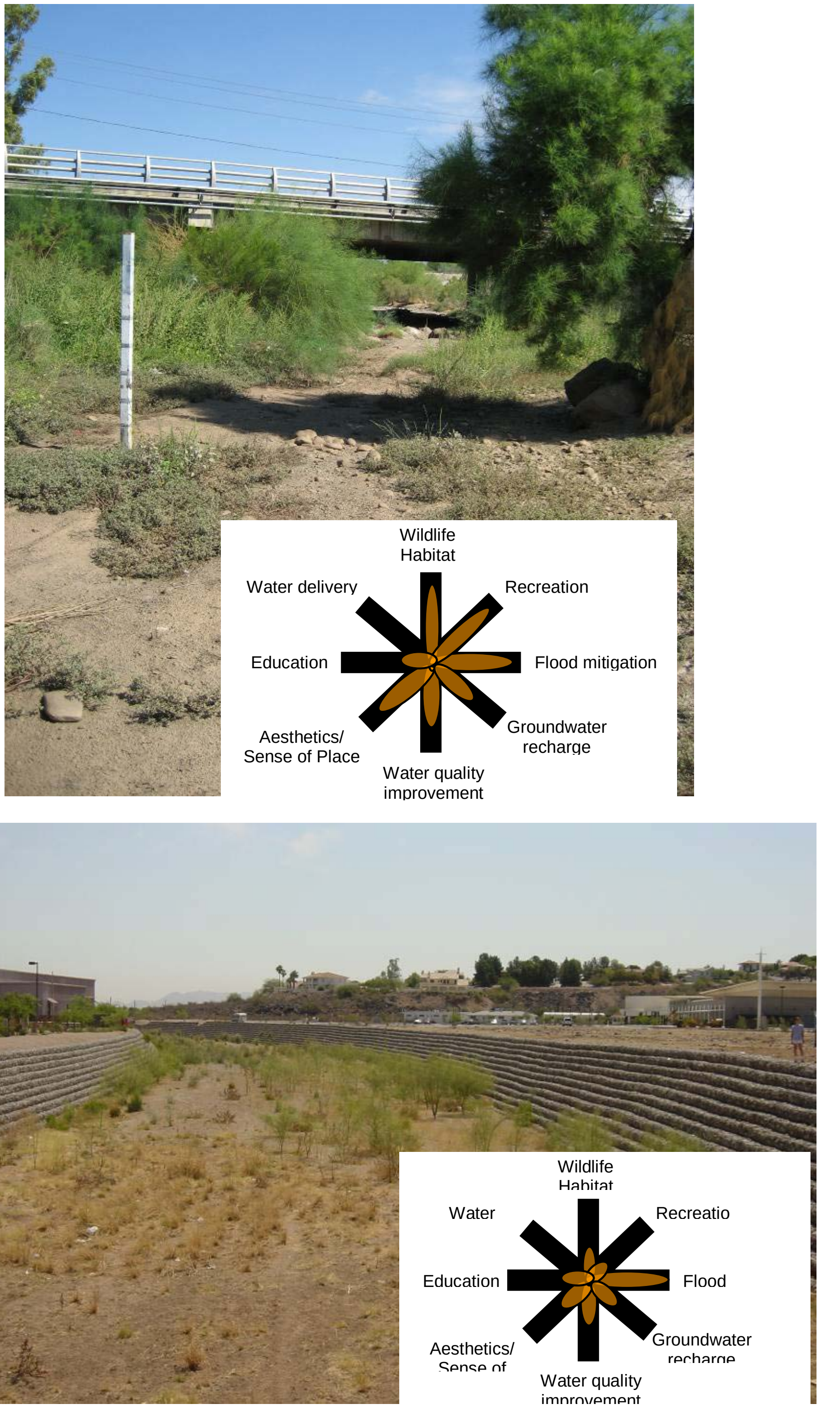

Figure 1.7: Examples of urban desert washes.

Figure 1.8. An example of an artificial lake.

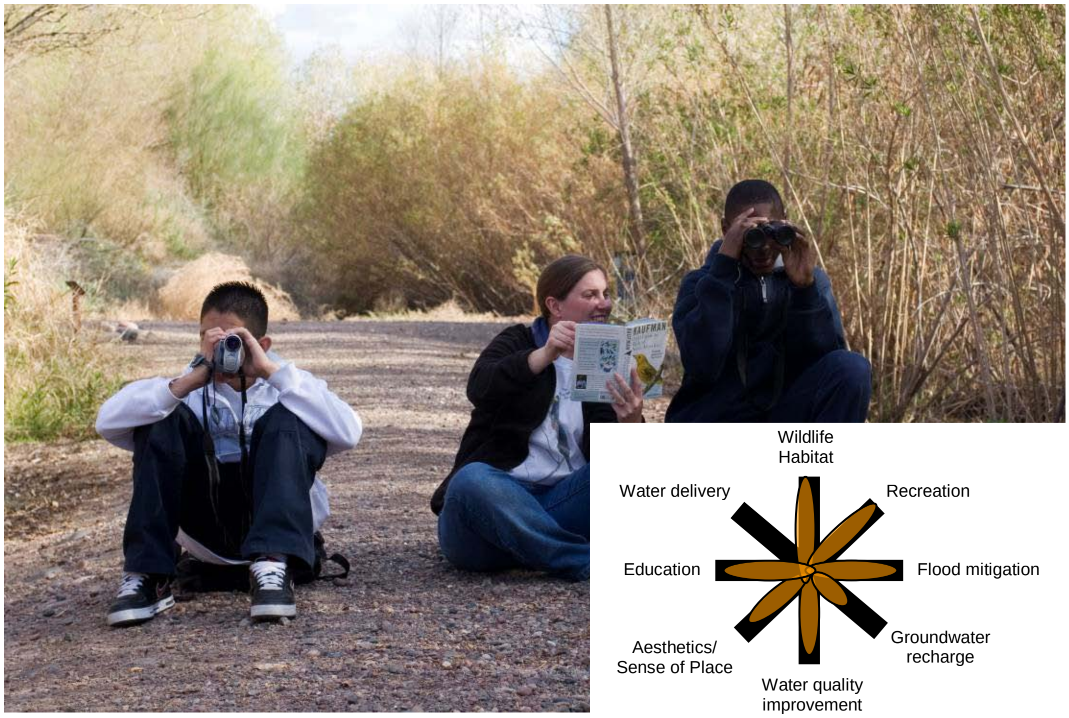

Figure 1.9: Students look for birds in a riparian woodland in the Rio Salado Project.

Chapter 2

SMALL-SCALE AND EXTENSIVE HYDROGEOMORPHIC MODIFICATION AND WATER REDISTRIBUTION IN A DESERT CITY AND IMPLICATIONS FOR REGIONAL NITROGEN REMOVAL

2.1 Abstract

There are numerous examples of small-scale hydrogeomorphic manipulations within urban ecosystems. These modifications are motivated both by a need to handle storm drainage as impervious cover increases and by a human desire for aquatic ecosystems as places for recreation and aesthetics. In the Phoenix Arizona metropolitan area, two examples of these local modifications are artificial lakes and stormwater retention basins. Although lakes are not a natural feature of Sonoran Desert ecosystems, numerous artificial lakes are evident in the region. Retention basins are a common Best Management Practice for preventing damage from rare but potentially large storm events. Here we attempt to quantify the heretofore unknown number and extent of these designed aquatic ecosystems and consider their potential impact on regional nitrogen (N) removal via denitrification. For lakes, we found that official GIS layers from local and state agencies had significant misclassifications and omissions. We used two published GIS datasets and impoundment-permit information to determine the number, areal extent, and water source for artificial lakes. We discovered that there are 908–1,390 lakes in the Phoenix area, with the number varying according to level of aggregation. There are no existing GIS data on retention basins, so we employed

drywell-permit data to estimate that there may be 10,000 retention basins in the region. Basic data on N stocks in these ecosystems are discussed within the context of the regional N budget. Accurate data on the extent and distribution of these designed ecosystems will be vital for water-resources planning and stormwater management.

2.2 Introduction

The world is rapidly becoming more and more urbanized – in 2005 it was estimated that 48.7% of the world's population lived in cities, and that this number will continue to rise over the next few decades (UNPD 2008). As urban areas grow and human impacts on the land, water and atmosphere increase, there is a pressing need for ecological understanding of the structure and function of city ecosystems. Both direct and indirect impacts of urbanization affect urban hydrology and geomorphology. Humans purposefully alter the existing regional hydrology and geomorphic structures, for example via dam and reservoir construction, in order to provide a regular supply of water for municipalities. Smaller-scale modifications include manipulation or creation of aquatic ecosystems such as wetlands and lakes. Indirectly, the expansion of impervious-surface area changes storm hydrology (Arnold and Gibbons 1996, Leopold et al. 2005, Walsh et al. 2005), requiring management of stormwater to prevent flooding.

In the Phoenix, Arizona, USA metropolitan area, the augmentation of nearby surface-water resources with groundwater and water from the Colorado

River supports a population of 4 million people (US Census 2008). The history of water rights and “reclamation” activities (large-scale dam building) in the USA’s western states has resulted in dampened variability of supply and relatively abundant and inexpensive water (Larson et al. 2005), allowing for many uses beyond industrial and municipal needs, such as golf courses and artificial lakes. The Indian Bend Wash watershed, for example, now has approximately 160 lakes, where as at the turn of the 20th Century there were none (Roach et al. 2008). These lakes are typically lined with clay or high-density polyethylene to prevent seepage to groundwater, and must be frequently replenished due to high evaporation rates in this desert locale. Water sources to each lake vary depending on available resources at the time of lake inception, and include groundwater, treated wastewater effluent, and canal water (a variable mix of Salt, Verde, and Colorado River waters, sometimes augmented with groundwater).

The natural flashiness of aridland hydrology, combined with increased impervious surface area in the city, requires careful planning for drainage and stormwater management. Drainage regulations of Maricopa County (which encompasses most of the Phoenix metropolitan area plus additional desert and agricultural lands to the southwest) stipulate that all developers of commercial, industrial, and multi-family residential subdivisions must develop drainage plans to minimize stormwater runoff effects and limit increases in peak discharge, volume, or velocity of runoff. Drainage systems must be able to retain the volume from a 2-hour, 100-year storm, and have the water dissipate within 36 h via percolation, drywells, or conveyance to an approved drainage way (Maricopa Planning and Development Department 2004). The regulations allow only 25% of public parking in the development to be used for stormwater storage, and then only to a maximum depth of 12 inches (30.48 cm), and roof storage of rainwater is not permitted. Stormwater retention basins are therefore common Best Management Practices (BMP) for local runoff retention. These basins are depressions excavated within the development, into which runoff from parking lots, roofs, and streets (if part of a subdivision development) is directed. There are two main types of landscaping within these basins: xeriscaped (drought-tolerant trees and shrubs that are drip-irrigated, with sand/gravel fill), and grassy (sod that is irrigated, mowed, and sometimes fertilized). By concentrating surface flows into these relatively small local areas, retention basins prevent flows into existing washes and rills, altering hydrologic landscape connectivity.

Since water is a premium commodity in the desert, we expected governmental and regional planning agencies to have relatively accurate GIS layers depicting where it is actually located. While the Arizona Department of Environmental Quality (ADEQ), the Flood Control District of Maricopa County (FCDMC), and the Maricopa Association of Governments (MAG) were able to supply GIS layers for lakes in the Phoenix metro area, even with a cursory examination the data sets for lakes collectively evinced numerous instances of omission, misidentification, and misclassification (Figure 2.1). The Arizona Department of Water Resources (ADWR) also provided data on permits for impoundments, including the source of the water (surface water, groundwater, or effluent) and surface area of the lake, spatially identified only the Public Land Survey System (PLSS) township, range, and section, and without information on the number of the lakes making up the surface area covered by the permit. With respect to stormwater retention basins, we are aware of no spatially explicit data for their location, size, and landscaping. The ADEQ does require permits for all drywells, a common but not required feature of stormwater retention basins, but these are only catalogued by street address.

Accurate, spatially explicit data about the location of aquatic ecosystems will be vital for future planning, as the Phoenix area is expected to experience continued population growth (MAG 2007), and climate models predict the region will become hotter and drier in the future, but with increased frequency and altered timing of flooding (Parry et al. 2007, Karl et al. 2009). Additionally, Arizona may have to reduce its allotment under new rules for the Colorado River Compact signed in 2007 (Archibold 2007). These factors may strain existing water resources, requiring Phoenix area decision makers to reassess current use allotment, potentially reassigning luxury water use to meet municipal needs. The changes in flooding regime may make existing design specifications for stormwater retention and drainage inadequate.

Ecological implications of lake and retention-basin creation may be substantial, especially in this arid environment. These novel ecosystems may result in asynchronous timing and differences in spatial distribution of nutrient and water inputs in this desert urban ecosystem. These changes may affect the capacity of the system to retain (via biotic uptake and storage in soils) or remove nutrients, resulting in altered nutrient processing within the larger city ecosystem (Wollheim et al. 2005, Kaye et al. 2006). As locations that potentially combine water with nutrients, artificial lakes and retention basins may be "hot spots" of nutrient transformation, i.e., locations with especially high biogeochemical processing rates (McClain et al. 2003). In the Phoenix area, these systems may help fill in some of the missing pieces of the Phoenix area nitrogen budget (Baker et al. 2001). A more thorough assessment of the contribution of component ecosystems to the greater city-wide ecosystem functioning will be necessary for making sustainable planning and management decisions.

The objectives of this research were 1) to determine the extent of hydrological modification to create lakes and retention basins, and 2) to explore the ecological implications of those modifications. We first identified the extent and number of artificial lakes and retention basins in the Phoenix metropolitan area. For artificial lakes, we refined and corrected existing GIS layers, and augmented this spatially explicit data with information on water source gathered from impoundment permits filed with the ADWR. For retention basins, we utilized drywell permit information collected by ADEQ as a proxy for the number of stormwater retention basins. To investigate the possibility that these novel ecosystems are hot spots for nitrogen removal, we conducted field surveys of both types of systems; collecting sediments from 15 lakes and soils from 32 retention basins (half grassy, half xeriscaped). In addition to evaluating bulk sediment and soil characteristics, we assessed denitrifying enzyme activity (DEA, or potential denitrification) via the acetylene-block method (Groffman et al. 1999).

2.3 Methods

Study Area

Phoenix is home to the Central Arizona–Phoenix Long-Term Ecological Research site (CAP LTER), which encompasses the city and surrounding suburbs and undeveloped desert, for a total area of 6400 km^2. The metropolitan area is located in the northern Sonoran Desert, receiving less than 200 mm of precipitation each year, with an average yearly temperature of 22º C. Despite its extreme climatic conditions, the CAP LTER area is one of the most rapidly growing metropolitan areas in the US, increasing from approximately 300,000 in 1950 to greater than 4 million in 2007 (US Census 2008). With increasing population comes increasing city expansion and land development, estimated at almost one half-mile per year (Gober and Burns 2002). The city sits at the confluence of the Salt and Gila Rivers, once perennial rivers that are now dammed or diverted upstream of the metropolitan area. Tributaries to the Salt are largely ephemeral and dammed as well, resulting in little instream flow throughout the washes, streams, and rivers of the city. Water resources for the metropolitan area include the Salt and Verde Rivers (ten year range from 1997 – 2006 = 21 – 41%), the Colorado River (14 – 27%), groundwater (28 – 40%), and treated effluent (5 – 7%) (ADWR 2009).

GIS Analysis -- Lakes

Stefanov et al. (2004) used spectral data from ASTER satellite imagery to classify land use/cover for the CAP LTER area in the year 2000. They originally created 16 categories, which included water, and were able to achieve a range of 81% to 98% user accuracy. The user accuracy at this stage for the water classification was 97.8% (Stefanov 2004). However, in their final analysis they aggregated several categories into five more general groupings, and eliminated water because of its small areal coverage. Thus, we began creating our lakes layer by selecting the areas identified as water by Stefanov et al. in their initial, underlying classifications.

The ASTER satellite makes diagonal passes over the region, and Stefanov et al. analyzed spectral data from only one pass for their categorization. Thus, areas to the northwest and southeast of the city of Phoenix are not included. These fringe areas have been regions of rapid development over the past few decades (Gober and Burns 2002), and may be sites of lake creation. Therefore, we augmented the ASTER imagery with LANDSAT ETM analysis conducted by Moeller (2007) that created a land use/cover map with similar classifications as Stefanov et al. (2004), including one for water. While accuracies for individual classes were not reported by Moeller, the overall accuracy for that analysis was 83%. The LANDSAT imagery analyzed encompassed the entire CAP LTER region, but at a coarser resolution (30 m) than the analyzed ASTER bands (15 m). Since many artificial lakes may be small golf-course ponds, we wanted to include the finer resolution analysis where available. Thus, we concatenated the LANDSAT and ASTER analyses to create a water layer that included both larger and smaller water bodies.

Upon manual truthing, we discovered that both Stefanov et al.'s ASTER analysis and Moeller's LANDSAT analysis had several instances when mountain areas were mistakenly categorized as water due to the darkness of the mountain shadows. To correct for this, we eliminated all areas with a slope > 10%. Additionally, some areas of dark pavement or skyscraper shadows were classified incorrectly as lakes, so we eliminated areas classified as water in the Phoenix business districts and airport. Because the layers are based on satellite imagery, bridges sometimes divide single water bodies into multiple objects. Therefore, to estimate the number of unique waterbodies, we merged individual lakes over a range of buffers from 10 to 50 m. Because we were interested in artificially created water bodies, we removed water identified within the Salt and Verde River floodplains. This process eliminated some water bodies that are in the numerous gravel-pit mines along the river, but GIS data were not available to accurately identify these pits as different than other surface water along the river. We then added Tempe Town Lake, an artificial lake of 0.89 km^2 that was created within the Salt River channel in 1999.

To determine the primary use for the identified lakes, we overlaid the layer with land-use data for the year 2000 provided by MAG. If a lake fell within more than one category, only the first category encountered in the GIS join was used. There were several categories that had a small number of lakes, so these were aggregated into an "Other" category, which includes cemeteries, public facilities, warehouses, and industrial and institutional land uses.

The ADWR provided information on permits for water impoundments in the Phoenix metropolitan area. Non-agricultural users who create impoundments larger than 1,145 m^2 must report the water source, and surface area of water used to the ADWR yearly. The permittees are not required to report the number of impoundments. As mentioned above, the only geographical reference for the lake location is the PLSS township, range, and section. A section is one square mile (2.59 km^2). The ADWR data had unique associations between sections and permits, so for our analysis, we assigned all spectrally identified water within a section to the water source provided by ADWR. It is possible that owners of small lakes below the permitting size limit, or agricultural lakes not subject to reporting requirements may share a section with an ADWR permitted lake, and thus may be incorrectly assigned to that water source.

GIS Analysis – Retention Basins

There are no known GIS data of retention basins for the entire Phoenix metropolitan area. Some individual cities have maps of the basins that are maintained by the city, but not all cities have digitized information, and there are many more private basins throughout the region. However, drywells are a common feature in retention basins, as they aid in post-storm dissipation of water (Maricopa Planning and Development Department 2004). Although the association of drywells with retention basins is unknown as is the potential number of retention basins without drywells, the number and location of drywells may potentially serve as a rough-cut estimate for the number of basins in the area. Drywells are point locations that promote movement of water to groundwater or to the vadose zone. They are likely associated with at least some sort of depression or locations where water accumulates, so even if only a fraction of them are in retention basins, information about drywells reflects the extent to which water is retained rather than directed to pipes, streams, and rivers. We obtained a list of drywell permits recorded from the the ADEQ's inception in 1986 through January 2009 for Phoenix Metropolitan area; about 38,500 drywells. The dataset included the facility name, address, permit date, and sometimes the depth of the drywell. Each drywell for the facility was a unique record. We used the StreetmapUSA address locator in ArcGIS to geocode these records. Adequate address information was unfortunately lacking for ~ 14% of the permits, and some permits only recorded the cross-streets as the location. We were able to map approximately 78% of the records with a match score of at least 50 and 86% with a score of at least 30. This resulted in 10,369 unique locations, including the records with only cross-streets as addresses (Figure 1.2). However, even in instances where the permit address was relatively accurate, we had no information as to the actual location on the property of the drywell(s). We overlaid the locations with a CAP LTER GIS layer for land cover (Moeller 2007) and 2004 MAG land-use data to characterize the potential landscape context for these wells based on address, and a random selection of drywell addresses was ground-truthed.

Lake water, sediment and soil biogeochemistry

In 2005, surface-water samples from 15 lakes were collected from a small boat anchored away from the littoral zone of each lake. Dissolved oxygen, temperature, specific conductivity, and salinity were measured using a YSI 85 meter. Sediment cores were collected with a custom-built lake corer of approximately 5-cm diameter for measurements of porewater chemistry, sediment chemistry, and denitrification enzyme activity (DEA). The top 8 cm were extruded and deposited into a plastic bag, taking care to exclude as much overlying surface water as possible. All samples were placed on ice and then refrigerated in the laboratory until analysis, within 24 – 48 h for time-sensitive analyses. To extract sediment porewater for analysis, half of the sediment cores for each site were centrifuged, and the supernatant decanted.

All water samples, both surface and porewater, were filtered first with pre-ashed, glass-fiber filters (Whatman GF/F; 0.7µm pore size). Samples for dissolved organic carbon (DOC) were then acidified and stored for analysis on a Shimadzu TOC-VC. Remaining water was further filtered with 0.2-µm syringe filters for analysis on Dionex DX 600 Dual Ion Chromatograph for bromide, chloride, nitrate, nitrite, sulfate, and phosphate. Because many of the lakes have either blue dye or algaecides such as copper sulfate in the water, colorimetric analysis of ammonium (NH_4^+) content was not possible for these samples.

In July 2006 we collected samples from 32 stormwater retention basins: 16 xersicaped and 16 grassy. At each basin, 4 samples were collected with a hammer corer in the lowest part(s) of the basin to approximately 8 cm depth and composited in a plastic bag. All samples were placed on ice and then refrigerated in the laboratory until analysis within 24 – 48 h for time sensitive analyses. Because we were interested in the amount of N immediately available after a storm event, we used deionized water, rather than a strong salt solution, to conduct soil nutrient extractions. Briefly, soils were sieved at field-moisture conditions and the < 2mm size fraction was homogenized manually. Then, 10 mg (± 0.05 mg) soil and 50 ml deionized water were agitated in sealed 250-ml polyethylene bottles for one h. Samples were then filtered through pre-leached Whatman #42 filters, and frozen until analysis. Filtrate was analyzed on a Lachat Quick Chem 8000 Flow Injection Analyzer for NO_3^- and NH_4^+.

Subsamples of sediment and soil were used in a laboratory experiment to determine DEA, a measure of denitrification potential, which is the maximum rate of denitrification possible under ideal conditions. We used the acetylene block method to measure DEA, with amendments as described by Groffman et al. (1999). Briefly, 50 mg sediment were added to Wheaton® bottles and then amended with 50 ml of media providing NO_3^- (100 mg N kg^{-1} sediment), dextrose (40 mg kg^{-1} sediment), and chloramphenicol (10 mg kg^{-1} sediment), the last added to prevent enzyme synthesis. Samples were sealed and flushed with N_2 gas to create anoxic conditions, leaving an N_2 headspace, to which 10 ml acetylene was injected through the septum. Pressure was equalized by briefly puncturing the septum with an open needle, and the samples were shaken. Initial gas samples were taken immediately after acetylene injection, and then 4 h later. Gas samples were analyzed on Shimadzu GC-14A with Poropak Q 50/80. DEA rates are reported as µg N_2O-N g soil or sediment $^{-1}$ h^{-1}.

Differences between grassy and xeriscaped soil characteristics were tested using a two-sample t-test. We used a General Linear Model (GLM) to explore the relationship between DEA and water and soil/sediment characteristics for both lakes and retention basins. Variables were log-transformed to reduce heteroskedasticity, when appropriate. For all tests $\alpha = 0.05$.

2.4 Results

Landscape Characteristics of Artificial Lakes

The ASTER + LANDSAT water layers created in our analysis revealed approximately 900–1,400 artificial lakes in the Phoenix Metropolitan Area, with a total area ranging from 7.9–8.2 km^2, depending on the level of aggregation. This represents approximately 0.4% of the urbanized area (Table 2.1, Figure 2.3). When compared with the existing data layers from ADEQ, FCDX, MAG, and ADWR, our estimate is greater in number but not areal extent (Table 2.2). There are several possible reasons for this discrepancy. For example, the MAG data categorizes a significant portion of the Salt River floodplain as a lake. The other layers could also potentially be missing newer or smaller lakes, or agricultural impoundments not officially considered lakes under state legislation.

A substantial portion of lakes were associated with golf courses, residences, and general open space (parks). Figure 2.4 shows the breakdown of land use and water source for the lakes at the 50-m level of aggregation. The relative proportions of land use and source do not change much for the 10- and 30-m aggregations (data not shown). One of the most striking results is that approximately 30% of the lakes identified spectrally have an unknown source of water according to ADWR. This is partially due to the fact that the Arizona Revised Statutes do not require permits for impoundments smaller than 1,145 m^2 (ARS 1987). At the highest level of aggregation, approximately 40% of the lakes are smaller than the legal reporting limit. In addition, many of the unknown water bodies occur on agricultural land and thus are not subject to reporting requirements. Of the remaining lakes, a small percentage has commingled water sources, otherwise the sources are roughly equally divided for groundwater, surface water, and treated effluent.

Landscape Characterization of Drywell Locations

The number of drywell permits increased rapidly in the 1990s, and seemed to slow in the latter part this decade (Figure 2.5). This period of time coincides with rapid population growth in the Phoenix metropolitan area, which from 1990 – 2000 experienced a 45.3% increase in population (US Census 2003), and a 24.2% from 2000 – 2006 (US Census 2006). The majority of the drywells occurred at addresses with xeric land cover and commercial, industrial, or office land use (Figure 2.6). The large number of drywells in asphalt locations is due in part to the 2,186 addresses that were identified only as cross-streets, and also due to the relatively high levels of asphalt associated with commercial/ industrial/ office land use. However, a survey of 30 asphalt and commercial/industrial addresses showed that most of these sites did have xeric retention basins associated with them. Even in instances where drywells were located within the parking lot, the approximately 75% of sites had retention basins as well. The low number of residential drywell locations, and especially of mesic (grassy) basins, is surprising, given the emphasis in the Maricopa drainage regulations of the benefits of using retention basins as multi-use sites (e.g., parks and soccer fields). It may be that basins that are large enough to support other uses do not require drywells to aid in water dissipation after storm events, either due to increased surface area (allowing for greater evaporation or potentially infiltration rates) or increased transpiration by grass and other mesic vegetation.

Lake and Retention Basin Biogeochemistry

For the fifteen lakes sampled in 2005, surface water chemistry was highly variable among lakes for all analytes. Surface water nitrate values ranged from 0.009 to 6.906 mg-N L^{-1}, surface sulfate concentrations ranged from 4.46 to 78.62 mg-S L^{-1}, and surface bromide concentrations ranged from 0.02 to 0.97 mg L^{-1}. Sediment porewater chemistry was variable as well, although all sediment nitrate concentrations were below 0.05 mg-N L^{-1} (data not shown). DEA by mass ranged from 0.1 to 2.8 mg N_2O-N kg^{-1} h^{-1} (Figure 2.7). We selected surface nitrate concentration and sediment sand percentage as the two dependent variable for the

GLM of DEA rates because NO_3^- is the terminal electron acceptor for denitrification, and denitrification is negatively correlated with sediment particle size (Garcia-Ruiz et al. 1998). The statistical analysis yielded and adjusted $R^2_{(2,12)}$ of 0.484 for DEA as a function of $\ln(NO_3^-)$ and percent sand, with both independent variables being significant. As expected, DEA showed a positive relationship with NO_3^- concentration and negative relationship with percent sand. Attempting to estimate the DEA by NO_3^- concentration alone yielded insignificant results.

For retention basins, clear differences in bulk soil properties were found between xeriscaped and grassy basins (Figure 2.8). Grassy basins had significantly higher soil water and organic matter content, which is to be expected since they are regularly irrigated. Grassy basins also had significantly lower NO_3^- concentrations than xeriscaped basins ($T_{(30)} = 2.03$, $p = 0.028$). This could be because grassy basins had significantly higher DEA than xeriscaped ones ($T_{(30)} = -6.07$, $p < 0.00001$). Indeed, DEA in grassy basins had a strong significant positive relationship with soil NO_3^- concentration (Figure 2.9), while no soil characteristics correlated with DEA in xeriscaped basins.

2.5 Discussion

Artificial lakes and stormwater retention basins in a management context

The lack of accurate data among the state agencies and regional organizations regarding the presence, location, size, and water sources of artificial lakes poses serious impediments to future planning and management. As a luxury water use, these lakes may need to be the first to be sacrificed in a future situation in which municipal and industrial demands increase, such as in the case of extreme prolonged drought or dramatic population growth. Both of these scenarios have been predicted for the Phoenix metropolitan area. Currently — aside from the very general information collected by ADWR on a subset of all lakes — there is no complete database and associated regulatory agency to enable assessment and prioritization.

Under the Clean Water Act (CWA) of 1972 (33 U.S.C. §1251 et seq.), all states are required to identify surface water bodies within their boundaries and classify their designated uses. For polluted waters, each state must report to the US Environmental Protection Agency a list of impaired waterbodies, or “303d” list – so named after the section of the CWA. However, according to ADEQ, urban, “man-made” lakes are not considered surface waters of the state of Arizona (ADEQ 2004). ADEQ notes that it is “interested in tracking water quality data” for these lakes, but it is difficult to imagine what impetus would drive this costly pursuit other than a regulatory requirement. And if the ADEQ is unaware of the existence of hundreds of these water bodies, the success of such a monitoring endeavor would be limited.

The large number of artificial lakes built in this desert city is a testament to the aesthetic, cultural, and recreational values these lakes provide. Although the state of Arizona has taken steps to limit creation of new lakes greater than 1,145 m^2 and reduce the dependency of existing lakes on groundwater sources (ARS

1987), only recently has ADEQ begun to explore establishing water quality standards for just 3 of these lakes (ADEQ 2008). As the climate and population change, planners and managers may have to reassess the luxury use of water in artificial lakes, potentially allocating the water to more pressing needs. Adequate understanding of the number, extent, and water quality of these lakes will be vital for making sound, sustainable decisions for water distribution and consumption.

The available data (dry well permits) allow for only the roughest approximation of the number and general location. Further investigation of the correlation between the presence of drywells and retention basins will be crucial for evaluating the proportion of stormwater that interacts with an ecosystem at the surface vs. being directed directly into the subsoil layers. Currently, spatially explicit data on the location, type, and contributing drainage area of retention basins are unknown. The existing mosaic of grassy and xeriscaped basins were created in a piecemeal fashion, as developers need only consider their specific property and use standardized equations and runoff coefficients. Without an understanding of these basins within a more regional context, adaptation to future growth and climate change may be difficult. Because retention basins cannot be identified using available remote sensing data, the only way to accurately map them (save manually going through tens of thousands of site drainage plans) would be to use Light Detecting and Ranging (LIDAR) technology, which can measure relatively small-scale differences in topography.

Ecological Implications

The creation of hundreds of permanent waterbodies in an arid city is certain to have numerous ecological ramifications. Shochat et al. (2006) identify three mechanisms that alter ecological patterns in cities: "(i) elevation of habitat productivity and interspecific competition; (ii) buffering of temporal variability; and (iii) alterations of trophic dynamics." Artificial lakes in the Phoenix area could potentially do all three of these. The exposure of often nutrient-rich water to over 300 days of sun a year creates ideal conditions for abundant algae growth. Many lakes are managed aggressively to prevent eutrophication by application of copper sulfate (an algaecide) or blue dye to shade growth, which may result in the accumulation of heavy metals in lake sediments. By providing a stable water supply, lakes could influence inter and intra-specific competition between birds, insects, synanthropic generalist mammals. In addition to buffering the variability of water supply, lakes may also modulate temperature variability, and mitigate the urban heat island effect. Artificial lakes are also frequently stocked with non-native fish for algae management and sport.

The Phoenix ecosystem is known to have elevated inputs of nitrogen (N) when compared to the surrounding desert (Baker et al. 2001). Hydrologic and geomorphic alterations in the city have eliminated many of the typical desert locations for N transformation. The creation of lakes and stormwater retention basins could provide new settings for biogeochemical processing. We have found that lakes receiving treated effluent or groundwater have significantly higher nitrate concentrations than those filled with surface water sources (unpublished data). While lake sediments tend to have high rates of potential denitrification (microbial transformation of N from nitrate to N_2 gas), the actual rates are probably limited by the rate of diffusion from the surface water into the sediments (Seitzinger et al. 2006). Stormwater retention basins, with intermittent wet periods, may remove more N from the system. However, questions remain about the magnitude and timing of actual denitrification rates. Zhu et al. (2004) measured denitrification rates from intact cores, ranging from 3.3 to 57.6 mg N m^{-2} d^{-1} (0–7.5 cm soil depth), but these were from grassy retention basins only. Our data suggest that grassy basins, while better at removing N from the urban ecosystem, are relatively rare. Additionally, it may be that the wetting frequency and duration in xeric basins from storms does not sustain large enough periods of denitrification to have a measurable impact on the Phoenix N budget. Further research is required to determine if lakes and retention basins are sinks for N in the greater Phoenix ecosystem. If they contribute significantly to important ecosystem services in addition to their intended built function, spatially explicit ecological data will be vital in evaluating potential tradeoffs among ecosystem services if future resources are strained for these water-dependent designed ecosystems.

2.6 Acknowledgments

This work would not have been possible without field and laboratory help from John Schampel, Kris Gade, Kayla Graham, Bony Ahmed, Sarah Moratto, and Katelyn Parady. Thanks to Kymberly Luttermoser from AZ Department of Water Resources for data and information on AZ regulations. Thanks to Carrolette Winstead from AZ Department of Environmental Quality for drywell data and assistance. Many thanks to Jessica Block for assistance with GIS analyses. This material is based upon work supported by the National Science Foundation under Grant No. 987612, IGERT in Urban Ecology, and Grant No. DEB-0423704, Central Arizona – Phoenix Long-Term Ecological Research (CAP LTER).

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Table 2.1: Artificial lakes in the Phoenix Metropolitan area by level of aggregation

Aggregation Buffer (m)	Count	Mean Area (m^2)	Total Area (km^2)
10	1350	5820	7.86
30	1039	7771	8.07
50	908	9002	8.17

Table 2.2. Calculated lake number and area by data source

Data Source	Count	Total Area (km^2)
ADEQ	83	4.2
FCDMC	211	17.9
MAG	66	37.2
ADWR	247*	10.8
This study	908 – 1390	7.6 – 8.2

*number of permittees. ADRW only requires surface area, not count.

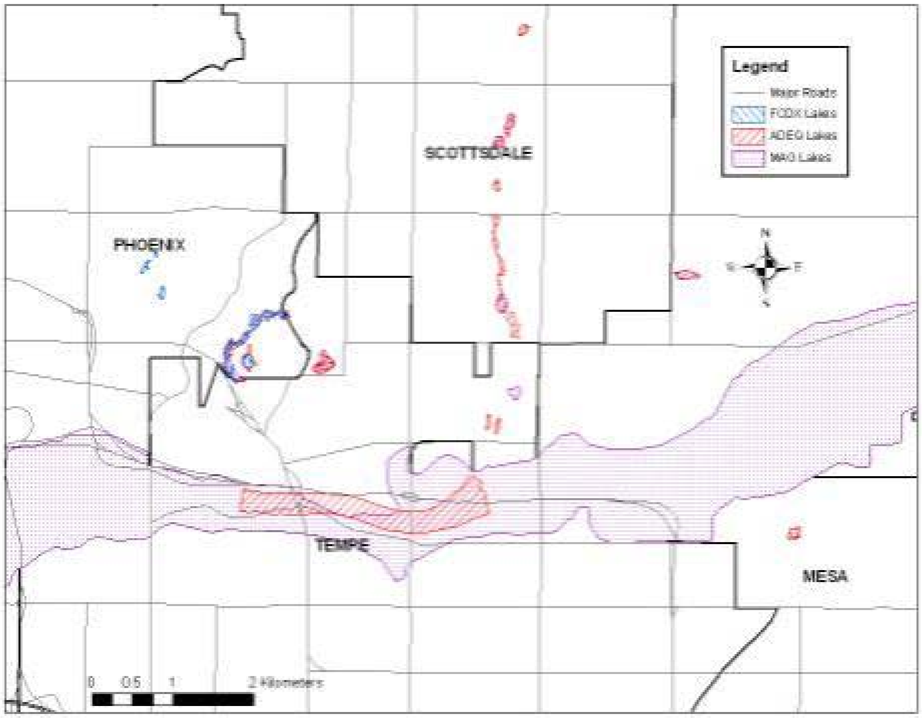

Figure 2.1: Example of discrepancies in GIS information regarding lakes in the Phoenix

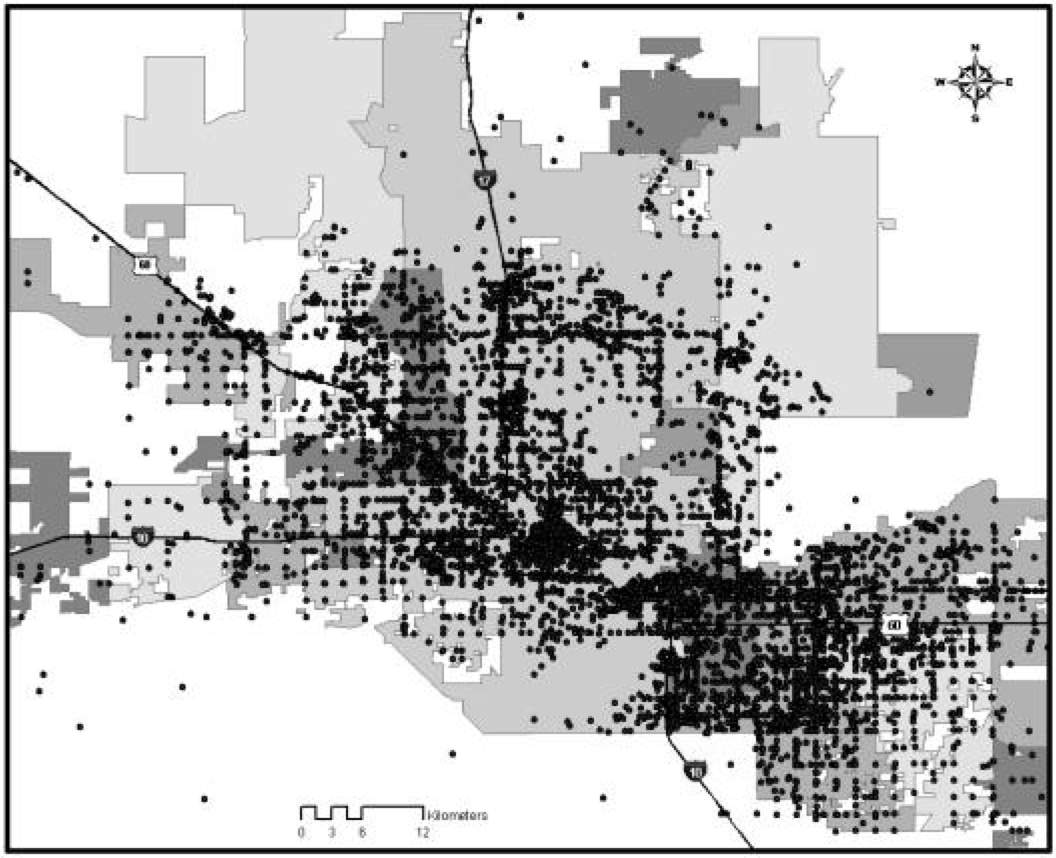

Figure 2.2: Map of unique addresses with drywells in the Phoenix metropolitan area.

Figure 2.3: Map of the Phoenix metropolitan area with lakes identified by this study. The extent of ASTER imagery analyzed is shown by the rhomboid with dashed lines. The extent of LANDSAT ETM imagery analyzed is shown by the solid-line rectangle.

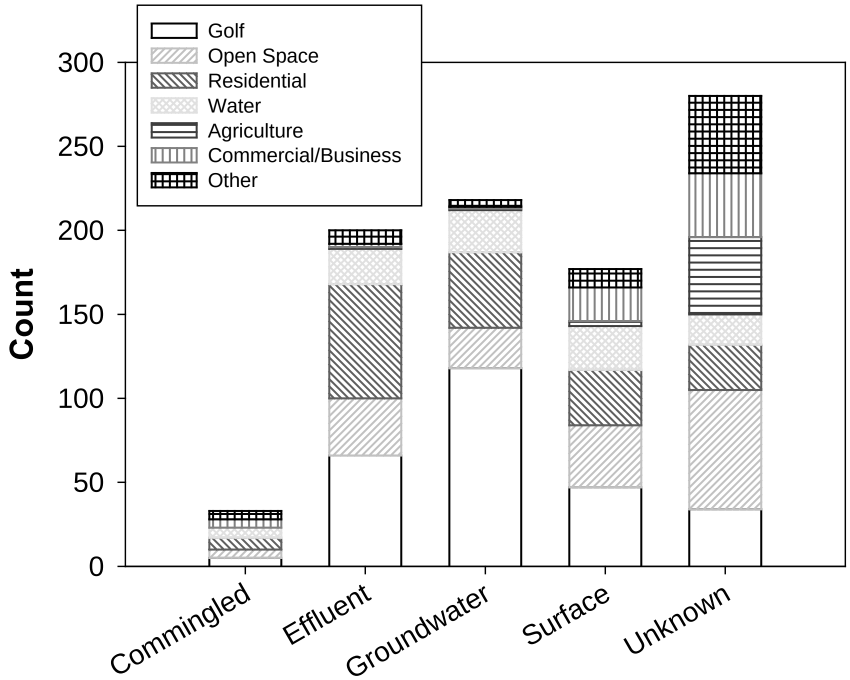

Figure 2.4: Frequency of artificial lakes by water source and land use.

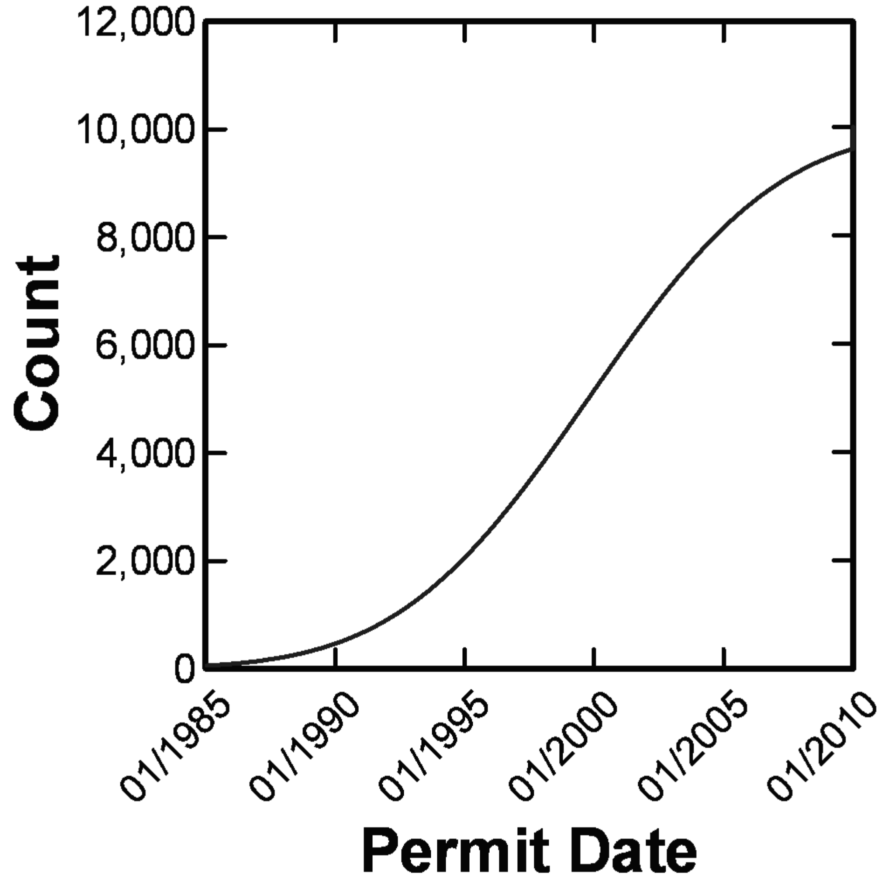

Figure 2.5: Cumulative number of unique addresses with drywells in the Phoenix metropolitan area by permit date.

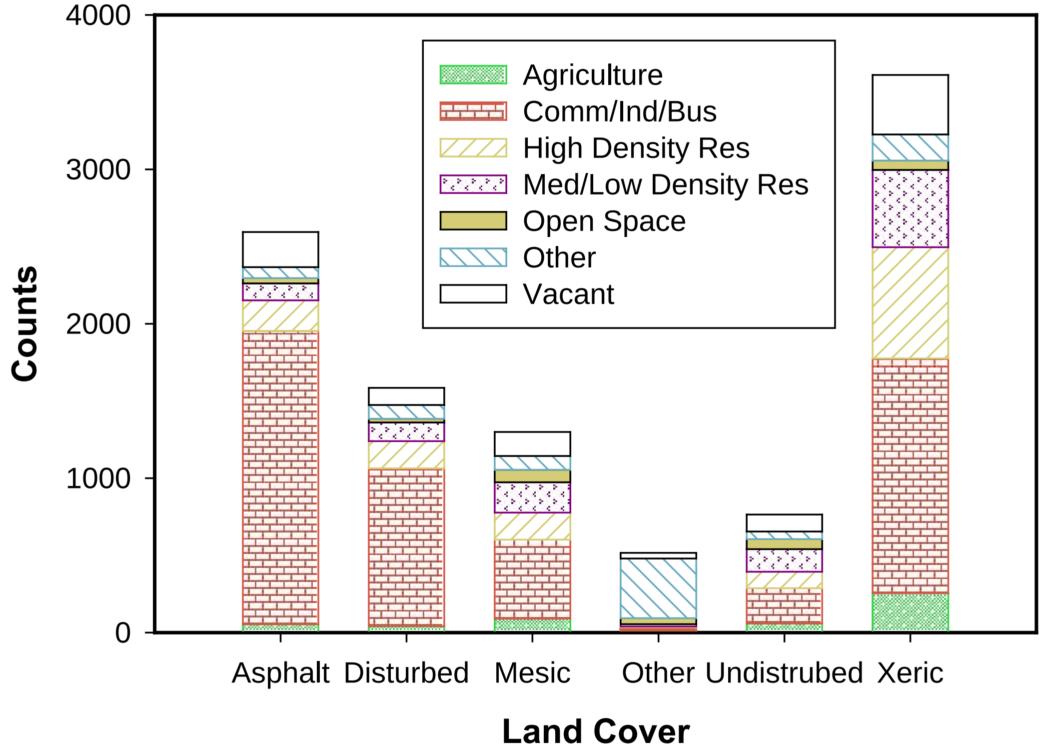

Figure 2.6: Unique addresses with drywells by land cover and land use.

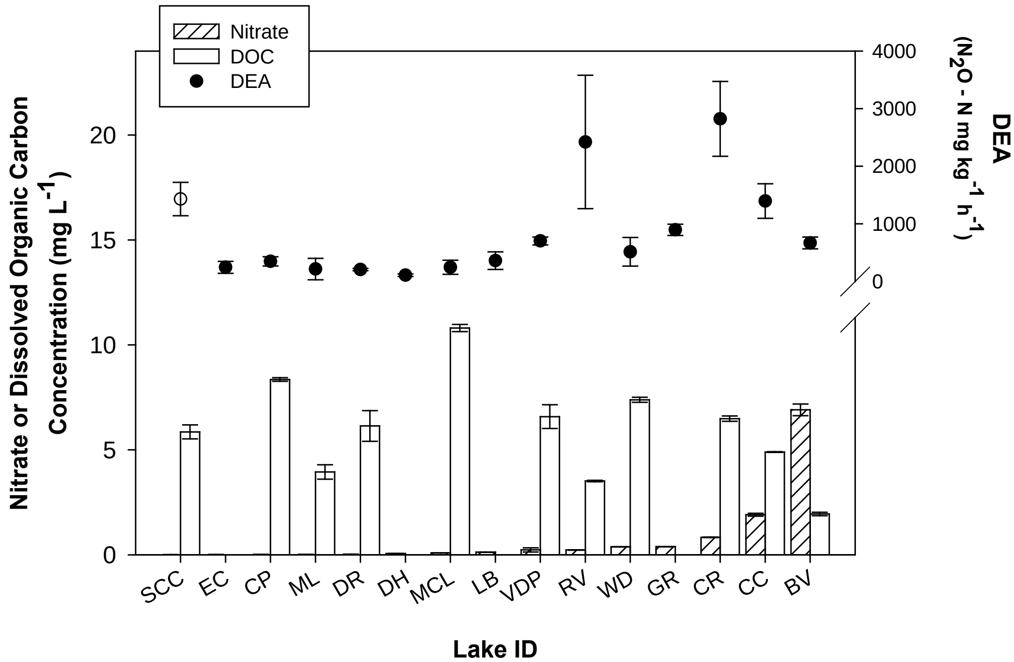

Figure 2.7: Surface chemistry and denitrification enzyme activity (DEA) for artificial lakes.

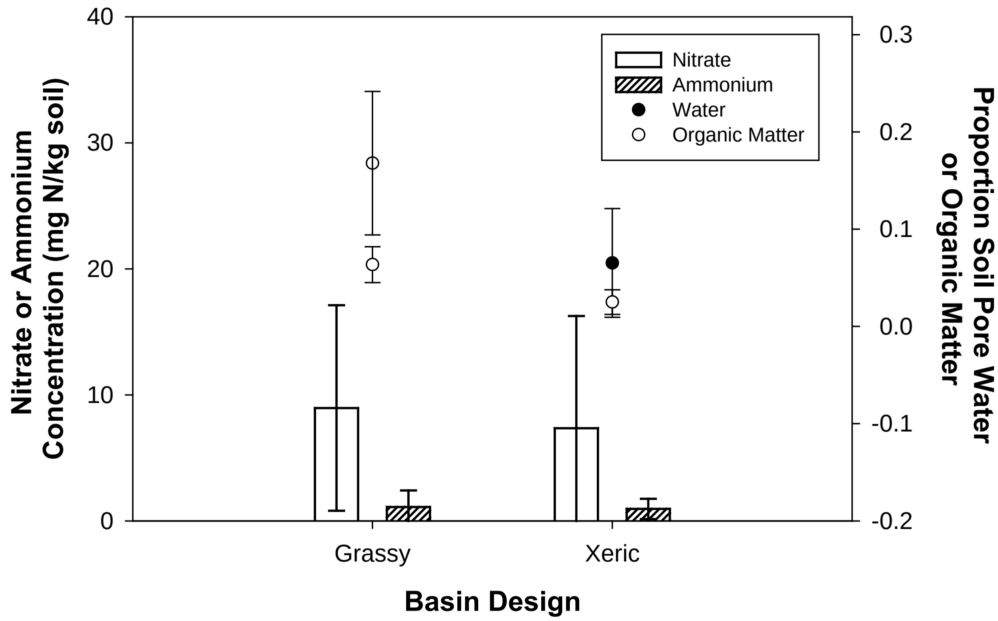

Figure 2.8: Retention basin bulk soil characteristics by landscape design.

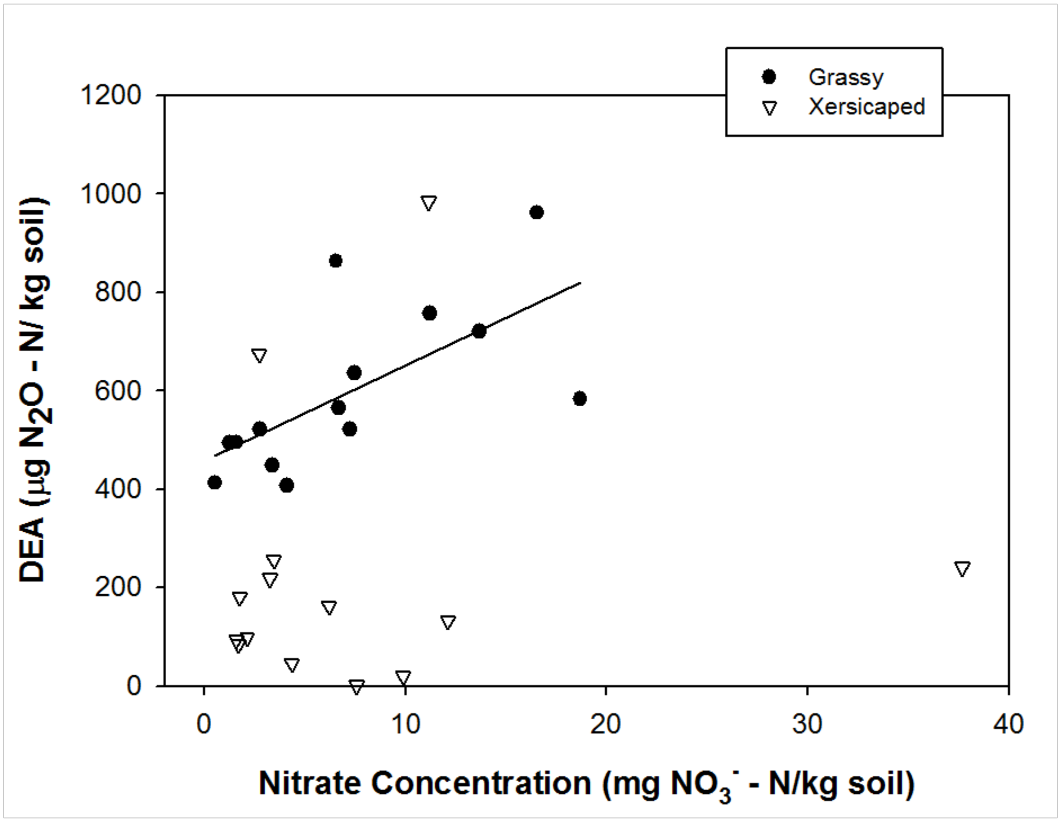

Figure 2.9: DEA by water-extractable soil nitrate concentrations for grassy and xeriscaped basins.

Chapter 3

THE VALUE OF WATER-RELATED AMENITIES IN AN ARID CITY:

THE CASE OF THE PHOENIX METROPOLITAN AREA

3.1 Abstract

In the arid metropolitan area of Phoenix, AZ, water resources play a vital role in maintaining and enhancing the urban ecosystem. There are several examples of "luxury" uses of water to create amenities not common to desert ecosystems: reduced temperatures, artificial lakes, golf courses, and abundant vegetation. In this study our goal was to appraise the relative value of these water-related amenities for urban residents. We correlated spatially explicit housing sales data from the Maricopa County Assessor's Office with environmental and locational data provided by the Central Arizona – Phoenix Long Term Ecological Research project to construct hedonic models at the regional and city scales to estimate the marginal willingness to pay for amenities associated with intensive water use. Our results revealed the preferences of homeowners for irrigation, lowered temperatures, and vegetation abundance, however we found proximity to small parks to be generally considered a disamenity despite their frequent landscape design of grass, trees, and artificial lakes. At the city level of analysis, our reveal instances where one attribute (e.g., plant richness) is considered an amenity in one place, but a disamenity in another, suggesting that there may be several markets in the metropolitan region. Because climate change models

predict the US Southwest to become hotter and drier, evaluation of the importance of these water-dependent luxury amenities will be vital for future planning.

3.2 Introduction

While the Millennium Ecosystem Assessment (MA) popularized the notion that ecosystems are a source of multiple services to people, it paid relatively little attention to constructed urban ecosystems. Since publication of the report of the Assessment (MA, 2005), there has been considerable interest in the identification and evaluation of the services delivered by urban ecosystems (Andersson 2006, Andersson et al. 2007, Tong et al. 2007, Tratalos et al. 2007, Phaneuf et al. 2008). However, these assessments have generally not addressed how changes in the built environment affect the value of the urban ecosystem to the inhabitants of cities. Phoenix, Arizona, is the location of one of two urban Long Term Ecological Research sites in the United States, and thus has a wealth of ecological data on the metropolitan area. Although there was little population growth in metropolitan Phoenix during the recession of 2008-9, the Phoenix Metropolitan Statistical Area (MSA) has been one of the most rapidly growing areas in the United States over the last sixty years. The population increased from around 300,000 in 1950 to more than 4 million inhabitants in 2008, with an estimated growth rate of 31.7% for the years 2000 -2008 (US Census 2008). This growth has been associated with extensive land use change (Jenerett and Wu 2001, Gober and Burns 2002). The development and change in the structure of natural and built environment over time has altered the set of services delivered by the local environment. Indeed, in many cases the changes were specifically wrought to enhance or reduce specific environmental characteristics, as in the case of landscaping and flood irrigation. Other changes, such as the urban heat island (UHI) effect, have been unintentional side effects. Our goal was to evaluate the relative value to urban residents of several environmental variables that are associated with this growth and structural change and that vary widely within the Phoenix area.

Urbanization typically creates landscapes that are heterogeneous and fragmented (Wu and Loucks 1995, Pickett and Rodgers 1997, Alberti 2005). In the case of Phoenix, a desert city of relatively low density, land use/cover patches range from built structures of widely varying height and mass, through impervious paved areas, residential lawns to relatively ‘natural looking’ parks and undeveloped areas. The conversion of peri-urban land from agriculture to domestic and industrial uses has fundamentally changed the patch structure within the Phoenix MSA, and in so doing has altered the array of ecosystem services it supports. A large part of the change involves change in the way that water is used. Some converted land maintains heavy use of water (e.g., artificial lakes, golf courses, flood-irrigated lawns) (Larson et al. 2005, Gober 2006). However, given concerns about the impact of climate change on available water resources (Hirschboeck and Meko 2005), and the future sustainability of water use in the area (Phoenix Water Services Dept. 1995, ADWR 1999, Morehouse et al. 2002, Gammage 2003, Jacobs and Holway 2004, Gober 2006), newly converted land is increasingly parsimonious in its use of water, hence it no longer provides many of the ecosystem services associated with heavy water use, such as attenuation of the urban heat island effect, public and private spaces characterized by lush vegetation, and water-intensive recreational amenity. In this paper we pay special attention to the effect of changes on water-related environmental attributes, such as vegetation abundance, proximitiy to water, and reduced UHI. We hypothesize that the ecosystem services associated with these variables – aesthetic value, cooling, recreation, and health – will be of great relative value to residents of this desert city. However, these attributes are not evenly distributed throughout the region, varying with development age, demographics, and location. Therefore, these variables will be of importance in determining the relative magnitude of value for water-related environmental variables.

What all urban patches have in common is that they are created to provide specific services to urban dwellers. At the same time, these services are seldom delivered in isolation. Just as agricultural monocultures have a range of impacts aside from the production of foods, fuels or fibers, so urban patches deliver a range of services/disservices aside from those they are designed to yield. Moreover, since different patches deliver distinct bundles of services, city dwellers typically rely on a range of patch types for the services they need (Karieva et al 2007). Although direct markets exist for some of the services patches are designed to yield, this is not the case for many designed services and for most ancillary services/disservices. Recreational areas are frequently provided on an open-access basis, and there are very few markets for incidental disservices such as exposure to noise, pollution or disease (Smith and Huang 1995, Leggett and Bockstael 2000, Zabel and Kiel 2000, Deaton and Hoehn 2004, Clark 2006, Greenstone and Gallagher 2008, Mendelsohn and Olmstead 2009). Yet those services and disservices do have value, and in many cases some or all of this value gets capitalized into the value of urban property.

Our approach supposes that the value of non-marketed ecosystem services derives from the value of the ‘product’ they help create. More particularly, we suppose that the marginal value of some ecosystem service is just the value of the marginal physical product of that service, and that it can therefore be derived from information on the value of the final product and on the role of the service in generating the product (Barbier 1994, Hueting et al. 1998, Barbier 2000, 2007). Examples of ecosystems services that have been valued in this way include the support of biological productivity in agro-ecosystems, climate regulation, maintenance of soil fertility, control of water runoff, and cleansing of water and air (Polasky and Solow 1995, Simpson et al. 1996, Mahan et al. 2000, Rausser and Small 2000, Goeschl and Swanson 2003, Heal et al. 2005).

The most widely used approach to deriving the demand for ecosystem services in built environments is the hedonic method. Since this method decomposes assets into the individual attributes that confer value, it is well suited to the analysis of constructed systems that yield an array of services or disservices. The value of the hedonic method in urban areas lies in the fact that cities are frequently characterized by well-developed property-markets. These markets provide a direct measure of peoples' willingness to pay for the attributes associated with the property, whether those attributes are priced or not. So for example, the price of a residential home encapsulates not only the specific attributes of the house (e.g., living space, type of roof, etc.), but a set of environmental and locational attributes as well (e.g., the amount and type of vegetation in the neighborhood, exposure to air, soil or water pollution, proximity to amenities such as parks, schools and hospitals or disamenities such as waste-disposal sites, etc.). By analyzing the relationship between house prices and environmental conditions, it is possible to estimate people's willingness to pay for a range of both amenities and disamenities. One caution is that the resulting value estimates capture only part of the value of the identified characteristics. The hedonic pricing method is not able to ascertain the specific services that influence people's economic behavior – e.g. whether the homeowner likes living near the park because of the recreation provided, or the vegetation and plant diversity, or privacy, or some combination of these.

We applied the hedonic pricing method to data on residential property sales in Phoenix in order to understand the value of the shifting array of ecosystem services provided by changes in the built environment. The study benefits from the fact that Phoenix is one of the most intensively studied urban ecosystems in the US, as it is home to the Central Arizona – Phoenix Long Term Ecological Research project (CAP LTER), one of two urban sites funded by the

National Science Foundation to study urban ecosystems over long time periods. CAP LTER seeks to expand the field of urban ecology by incorporating human dynamics into the understanding of cities as complex systems. Over the past 10 years, CAP LTER has compiled data on many environmental factors for the Phoenix metropolitan area and outlying desert (Figure 3.1). Using these data together with data on residential property sales we estimated the implicit marginal willingness to pay for a range of environmental and locational attributes associated with differing forms of water use. We performed this analysis for the whole CAP LTER region, and then segmented by city to assess potential differences in sub-markets. Finally, we conduct a "second stage" analysis to estimate demand curves for selected ecosystem services, making it possible to predict the implications of changes in ecosystem service 'prices' for the quantity demanded of those services. We anticipate that the results may be useful to urban planners interested in exploring different options for the provision of particular ecosystem services.

Urban ecosystem services and their valuation

The most studied environmental consequences of urban life are a set of environmental hazards or disamenities: increased levels of noise, pollution, pathogens, heat (McMichael 2000), and flooding (Paul and Meyer 2001, Walsh et al. 2005). However, urban dwellers also benefit daily from environmental amenities, some of which offset the impact of the hazards mentioned above. Ecosystem services provided by various types of urban ecosystems include microclimatic regulation, noise reduction, stormwater drainage, sewage treatment, air filtering, recreation and aesthetics. These services are provided by distinct terrestrial and aquatic ecosystems found within cities (Bolund and Hunhammer 1999, Tratalos et al. 2007). Most such systems provide multiple services. Indeed, urban hedgerows, parks, lakes, lawns, green roofs, and vacant lots (to name a few of these ecosystems), can simultaneously provide an array of services to urban inhabitants, some of which will be easily apparent to its recipients, but many will not be. One task of economic valuation in these circumstances is to tease out the value of individual services from the full array, taking into account the composite and sometimes invisible nature of the ecosystems providing these benefits. A second is to assess the value of particular urban ecosystems in terms of the full array of services.

Researchers have used a variety of valuation methods to estimate the value of both disamenities and amenities in urban areas. Some studies have focused on single services, such as water quality improvement (Leggett and Bockstael 2000, Bateman et al. 2006), pest control (Jetter et al. 2004), seed dispersal (Hougner et al. 2006), air quality improvement (McPherson 1992, McPherson et al. 1998, Zabel and Kiel 2000, Escobedo et al. 2008, Clark et al. 2008), climate regulation (Clark et al. 2008), recreation and aesthetics (Kulshreshtha and Gillies 1993, Jim and Chen 2006) and stormwater abatement (McPherson 1992, Clark et al. 2008). Others have focused on the value of specific types of ecosystems, such as wetlands (Doss and Taff 1996, Mahan et al. 2000, Lupi et al. 2002, Boyer and

Polasky 2004, Tong et al. 2007), parks (More et al. 1988, Lockwood and Tracy 1995, Salazar and Menendez 2007), urban forests (Tyrvainen 2001, Popoola and Ajewole 2002, Kwak et al. 2003, Mansfield et al. 2005, Treiman and Gartner 2006) and open space (Breffle et al. 1998, Lindsey and Knaap 1999, Smith et al. 2002, Morancho 2003).

The valuation methods applied have been quite diverse including replacement cost (how much it would cost for a technological solution), travel cost (how much people spend to visit particular ecosystems), contingent valuation and contingent choice modeling, and hedonic modeling. The dominant methods used have been contingent valuation and hedonic pricing. Contingent valuation is a set of survey-based methods which elicit respondents' willingness to pay for specific ecosystems and/or the services they provide (or, conversely, their willingness to accept compensation for the presence of disservices or undesirable land uses). While these methods have the advantage that they are able to target specific ecosystem services they suffer from two main drawbacks. One is that the method itself is resource intensive. A second is that the contingent nature of the investigation makes it vulnerable to a range of biases. While these can be reduced through efficient instrument design, they cannot be eliminated. Hedonic valuation methods, on the other hand, fall into the category of "revealed preference," as they are based on actual market purchases. They do not therefore suffer from the same potential for bias. They are, however, necessarily limited in their ability to assess the value of ecosystem services for which there are no direct measures.

For example, research in Portland, Oregon, USA, has shown that proximity to a wetland increases the value of a house, and the marginal implicit price of increasing the size of the nearest wetland by one acre is $24.39 (Mahan et al. 2000). However, this analysis does not reveal why people value the wetland: is it due to aesthetics, bird-watching, flood abatement, or privacy, or some combination of these things? Of course if spatially explicit measures of such services exist and are a part of the analysis, the method will yield an estimate of their value.

The selection of valuation method depends on the question of interest, but also frequently on the data available for analysis as well. In many cities, spatially explicit environmental data are sparse or non-existent. For this paper, we saw a unique opportunity to utilize the environmental data available from the CAP LTER in conjunction with housing sales data for the Phoenix area using hedonic valuation. Although we were constrained by the nature of the available environmental data, and the fact that not all urban ecosystem services are reflected in house purchases, we were able to generate estimates of the value that urban residents place on an important set of environmental amenities. In particular, we were able to estimate the value of environmental attributes that are dependent on a substantial allocation of water resources for their maintenance. As predictions of climate change for the US Southwest point to a hotter, drier climate, this will help decision-makers understand the trade-offs involved in planning decisions that alter the set of services enjoyed by residents.

3.3 Data and Methods

Housing Sales and Environmental Data

Data for the study were obtained from a number of sources. The US Census data were used for demographic information of the tracts encompassing each parcel. The housing sales data came from the Maricopa County Assessor's Office. In order to correlate sales with a time period for which we had the most environmental data, we selected sales for only the year 2000 from the total dataset, which was compiled in 2005. Because of the structure of the Assessor's database, the resale of homes overwrites prior sale data, so we were not able to capture data for homes sold in 2000 and again in a later year. However, given this restriction, there were still greater than 42,000 records of single-family residential sales. To eliminate the unduly large influence of very expensive and inexpensive homes, we limited our analysis to sales where the price was greater than $60,000 and less than $750,000. This reduces our sample size to approximately 40,000. The Assessor's data include vital information about the properties that were included in our analysis: size of the house and lot, type of roof, presence of a garage and pool, and the construction year.

The environmental and locational data came primarily from the CAP LTER, although additional data were gathered from the City of Phoenix, the Maricopa Association of Governments (MAG) and GIS Services at the Institute for Social Science Research (ISSR) at Arizona State University, plus individual researchers (Table 3.1). For our analysis, we selected attributes representing ecosystem services that would be apparent to homeowners and might be expected to influence their purchase decision.

We were primarily interested in ecosystem services associated with the vegetation type or land-water balance on urban patches. These services include provision of habitat for bird and other species, association with the desert environment (a sense of place), recreation and amenity, noise abatement, microclimatic regulation and air quality enhancement. Vegetation cover varies between private and public spaces. The vegetation on residential properties tends to be one of two main types. Some homes have green lawns with trees and shrubs (mesic vegetation), while others are landscaped with desert plants and an inorganic mulch (xeric vegetation). The ecosystem services provided by the two vegetation types differ: xeriscaped areas can enhance a sense of ‘place’, being more desert-like, but mesiscaped landscaping provides cooling, recreation, privacy, and/or noise reduction. Vegetation data were derived from the Soil Adjusted Vegetation Index (SAVI), created from the remotely sensed Landsat Thematic Mapper (ETM) image. CAP LTER also has interpolated data from CAP LTER surveys which estimate plant richness (diversity) and bird abundance, and the ISSR provided information about which parcels were flood-irrigated.

Air quality enhancement (particulate matter reduction) and microclimate regulation (heat reduction) were assessed by ambient (neighborhood) attributes. Air quality was measured by the mean concentration ($\mu g\ m^{-3}$) of particulate matter $\leq 10\ \mu m$ (PM_{10}). This pollutant is one of the few to be visible to the naked eye.

Microclimate regulation was measured by the mean minimum temperature for August. In the Phoenix metropolitan area, the urban heat island effect is observed in the elevation of night-time temperatures (Baker et al. 2002), thus locations with high mean August minima are hot-spots. The recreational service of public open space and water amenities were measured by the log of distance to small recreational parks, larger open space and native desert areas, canals, and streams. We also included a distance measure to the Phoenix Sky Harbor International Airport, which is near the central business district. The habitat value of vegetation was measured by bird abundance data drawn from the CAP LTER.

ArcGIS was used to calculate both ambient (e.g. vegetation amount, air pollution concentration) and distance characteristics for each parcel sold. Distances were measured as the Euclidian distance in ft from the centroid of the parcel to the centroid of the feature of interest. Ambient conditions were calculated as the average of conditions within the parcel. To reduce heteroskedasticity of the data, all distance measurements, as well as the house price, were log-transformed. A wide range of values for each variable were found across the metro region (Table 3.2).

The hedonic property model

Hedonic property models are widely used for assessing environmental impacts on property values. The basic approach has already been described. Property values are decomposed to reflect the attributes of those properties, some of which may be environmental. These values may be more or less specific to the location in which the attributes occur. Hedonic property models have become a popular candidate for meta-analysis where values are not necessarily location-specific. Meta-regression models use a range of estimates of the value of some environmental attribute at different sites with a view to estimating the value of the same attribute at some policy site (Rosenberger and Loomis 2000, Shrestha and Loomis 2001, Johnston et al. 2005, Johnston et al. 2006), and there exist a number of meta-analyses based on hedonic models (e.g., Smith and Huang 1995, Mrozek and Taylor 2002, Smith and Pattanayak 2002). However, wherever the value of particular ecosystem services is context-dependent (is specific to the location where the services occur), meta-analyses are of limited value, and a hedonic model should be estimated for that location.

We estimated a hedonic price function for the Phoenix area of the following general form:

$$p_i = f\left(\mathbf{h}_i, \mathbf{a}_i, \mathbf{s}_i\right) \tag{1}$$

where p_i is the price of the i^{th} property sold during the reference period, $\mathbf{h}_i$ is a vector of house characteristics, $\mathbf{a}_i$ is a vector of environmental or ambient conditions, and $\mathbf{s}_i$ is a vector of location-specific ecosystem services. This reflects an underlying household utility function of the form:

$$U_j = U\left(\mathbf{x}_j, \mathbf{h}_j, \mathbf{a}_j, \mathbf{s}_j\right) \tag{2}$$

in which $\mathbf{x}_j$ is a vector of all other commodities consumed by the j^{th} household. The household preferences described in (2) are assumed to be weakly separable in the set of housing and housing-related services and all other commodities. The assumption allowed us to estimate demand for housing services independent of the prices of other commodities, since the marginal rates of substitution between housing services does not depend on the quantities consumed of all other commodities. This implies that demand for housing services depends only on the ‘prices’ of those services and total expenditure.

There are a number of options for the specification of (1) depending on the assumptions made about interactions in the ecosystem services associated with distinct residential properties. If the characteristics of individual properties impose significant external effects on neighboring properties a spatial autoregressive specification would be appropriate (Can 1992). However, if location effects are restricted to the impact of shared ambient conditions, it is more appropriate to estimate a single regression equation relating property prices to ambient conditions. While it is possible that interactions due to vegetation characteristics may involve neighbor-externalities in parts of Phoenix, we were primarily concerned with the impact of shared air quality and air temperatures, and so elected to use a single regression equation of the form:

$$\mathbf{p} = f\left(\mathbf{h}, \mathbf{a}, \mathbf{s}, \mathbf{y}, \boldsymbol{\beta}, \boldsymbol{\mu}, \boldsymbol{\lambda}, \boldsymbol{\gamma}\right) + \boldsymbol{\varepsilon} \tag{3}$$

where **p** is a vector of observed market prices of housing, **h**, **a** and **s** are vectors of housing, ambient (neighborhood) and ecosystem service attributes, **y** is a vector of household characteristic, $\boldsymbol{\beta}$, μ, λ and $\boldsymbol{\gamma}$ are the associated parameter vectors, and $\boldsymbol{\varepsilon}$ is a vector of random error terms.

We expected that ambient effects would be associated with differences in attribute ‘prices’, implying the existence of distinct sub-markets. This implies spatial heterogeneity in the parameters of the hedonic price function, and so heterogeneity in the marginal willingness to pay for common attributes across the city. To address this we identified N geographically distinct sub-markets characterized by distinct neighborhood conditions, and estimated the hedonic price function initially for the whole area and then separately for each submarket. The estimated hedonic price function for the N submarkets has the form:

$$\mathbf{p}_n = f_n(\mathbf{h}, \mathbf{s}, \mathbf{y}, \boldsymbol{\beta}, \boldsymbol{\lambda}, \boldsymbol{\gamma}) + \boldsymbol{\varepsilon}_n, n = 1, \ldots, N \qquad (4)$$

In addition, we used ArgGIS to create maps of the spatial configuration of the economic value of selected (dis)amenities: vegetation abundance, plant richness, and a summation of all ambient environmental attributes (vegetation abundance, plant richness, bird abundance, PM_{10}, and August mean minimum temperature). For each spatial location, the level of the variable of interest was multiplied by the corresponding regression coefficient and the mean house price for the metropolitan area.

3.4 Results

For the total metropolitan area dataset (N = 38,333), the hedonic property model explained 87.1% of the variance in housing price (Table 3.3). As one would expect, the dominant characteristics were those associated with the structural features of the house, and in particular size, construction materials, garaging and pools. However, certain ambient environmental characteristics were also found to be important, including heat island effects and air quality, along with a number of proxies for ecosystem types including on site irrigation, plant richness and vegetation, and access to off-site water bodies and open spaces. In general, the signs of the coefficients were as expected. We found that both particulate pollution (PM_{10}) and heat-island effects (August minimum temperature) are disamenities in that they lower the price people are willing to pay for housing. This implies that the corresponding ecosystem services, air quality enhancement and heat attenuation, are positively valued. We also found that vegetation associated with ecosystem types delivering positively valued services (measured by the Soil Adjusted Vegetation Index and the index of plant richness) enhanced property prices. One caveat, here, is that because mesic systems deliver multiple ecosystem services – including habitat provision and recreation as well as air quality enhancement and heat attenuation – we were unable to infer willingness to pay for particular services from the preference for increased vegetation abundance.

Not all coefficients had the expected sign. For example, bird abundance had been selected as measure for the habitat value of vegetation type, and we had expected it to contribute positively to house prices. Instead we found the coefficient to be negative. Similarly, the relationship between house price and distance to recreational open spaces turned out to vary between types of open spaces in ways that were not expected. The sign on the coefficient on distance to water and small parks, for example, suggests that both are a source of disamenity.

By contrast, proximity to large parks is a clear amenity. The unexpected signs on small parks and water areas may be an artifact of the aggregation of sub-markets, for example neighborhoods in the older parts (close to small parks) of the city being lower priced than elsewhere. The amenity of proximity to waterbodies may only be actualized on very small local scales. As we see in the submarket analysis, at the city level proximity to parks is most often revealed to be an amenity (Table 3.7).

To explore differences between potential submarkets, we also estimated a hedonic price function for individual cities in the Phoenix metropolitan area. Several cities had too few sales to allow for adequate statistical analysis, but we were able to estimate regression coefficients for eleven cities, as well as approximately 1,900 sales of homes in unincorporated (county) areas (Tables 3.4 and 3.5). For the vegetation and bird abundance variables, the coefficients have the same sign across all cities. That is, increasing vegetation is clearly an amenity, and increasing bird abundance is clearly not. But for other variables, the signs change depending on city; so some environmental attributes are amenities in some areas and disamenities in others. For PM_{10}, the sign of the coefficient is positive in half of the cities, and negative in the other half (two cities have insignificant coefficients). The sign of the coefficient also varies for the plant richness variable. While in some cities greater plant diversity may be an amenity, in other cities homeowners may prefer the more homogeneous landscaping of lawns with one or two trees.

The values of locational attributes are more variable, although proximity to small parks is a disamenity in all cities (Table 3.5). House values increase with distance from streams for most cities; this may be due to potential flood risk in this arid environment. But it may also reflect the desire to live at higher elevations. The mostly negative signs for the desert distance coefficients could be indicative of the desire to live at the edge of urban development. Proximity to water and large parks are almost uniformly considered amenities.

For environmental attributes (Table 3.6), MWTP ranged from -$26,453 for a 10% increase in mean bird abundance (Tempe) to $12,315 for a 10% increase in mean vegetation abundance (Scottsdale). For the entire metropolitan area, a 10% increase in mean vegetation yielded a MTWP of $3,944, which is greater than the MTWP for increased plant diversity ($1,123). For locational variables (Table 3.7), MTWP ranged from -$2,257 for an increase of 1,000 ft proximity to small parks (Scottsdale), to $3,032 for an increase of 1,000 ft proximity to large parks (Goodyear). For the total dataset, MWTP values were more modest in both directions, ranging from -$437 for increased proximity to small parks, to $139 for increased proximity to steams.

The average decade of development is mapped as well. In general, it appears that the value of vegetation abundance is greater in the older tracts as revealed by Figures 3.2 and 3.3, respectively. We tested this quantitatively by looking at the interaction between vegetation abundance and average year of construction for the census tract, which was significant ($p < 0.0001$) and negative

(-1.2×10^{-8}). There are potentially two factors at work here. One is that older areas are more likely to be flood irrigated. The other is that older areas are more likely to have more mature vegetation. Both factors promote vegetation abundance. Conversely, the value of plant richness is clearly higher in the newer areas of the north and northeast metropolitan area. The coefficient in the interaction between plant richness and average census tract construction year was significant ($p < 0.0001$) and positive (0.002). These areas are more likely to be xeriscaped, and to have previously been a desert rather than agriculture land cover. The urban areas in the southeast and western parts of the region generally were previously likely to be agriculture, and then converted to mesic residential land use (Buyantuyev et al. 2010), which tends to have less plant diversity. People in the Phoenix metropolitan area prefer an "oasis" style of xeriscaping, which has a greater variety of plants than desert landscapes (Martin et al. 2003). This apparent inverse relationship further supports the ideas that in the older sections homeowners value the more homogeneous, water-intensive landscaping of water in trees, and in the newer northern areas they prefer a landscape of diverse mixture of xeric plants. The interaction coefficient between vegetation and abundance and plant richness is significantly negative ($\beta = -7.2 \times 10^{-6}$, $p < 0.0001$), indicating a tradeoff between the value of vegetation abundance and plant diversity.

In general, most of the metropolitan area received positive net value from ambient environmental (dis)amenities (Figure 3.4). The places with the highest net value (dark green), are either in older neighborhoods near central Phoenix where there is abundant, mature vegetation that is flood irrigated, or rich communities to the north and northeast, such as Fountain Hills (2008 median household income: $81,377), Scottsdale ($72,033), Paradise Valley (1999 median household income: $150,228) (US Census 2010a, b), which are largely xeriscaped but have lower UHI and PM_{10}, and a large number of artificial lakes (often associated with golf courses). The areas to the southeast that exhibit medium net value are recently converted agricultural lands with somewhat lower UHI, PM_{10}, but few artificial lakes. For the most part, places with negative net value are near highways and/or industrial/business and airport land uses.

3.5 Discussion

The results of the overall hedonic model reveal homeowner preferences for several important environmental and locational characteristics for the Phoenix metropolitan area. The coefficients on several variables had the expected sign, with irrigation, vegetation abundance, plant richness, low air pollution and summer time temperatures, and proximity to large parks and the desert all being statistically significant amenities. However, the signs on the coefficients on some variables were unexpected. This might reflect omitted variables or confounding factors. The unexpected signs on these variables may also, however, reflect the dominance of variables that, in some areas, are correlated with them. For example, research has shown that bird abundance increases with development density and housing age in urban areas (Loss et al. 2009, Ortega-Alvarez and

MacGregor-Fors 2009). Thus, the bird abundance may be acting a proxy for other undesirable conditions associated with crowding and older homes. It may also be that the birds themselves are unwanted. Studies have found that birds in urbanized areas with higher abundances tend to be non-native generalists and are sometimes viewed as pests (Palomino and Carrascal 2006, Loss et al. 2009, Ortega-Alvarez and MacGregor-Fors 2009, van Rensburg et al. 2009).

With respect to air quality, the positive correlation in some instances between PM_{10} and house price initially appear counterintuitive. However, two major sources of PM_{10} pollution in the Phoenix area are agriculture and construction. It may be that higher levels of PM_{10} are indicative of new construction, especially near agriculture. Most of the agriculture in the metropolitan area is located near the urban fringe, thus PM_{10} may actually be a proxy for new homes in a desirable location. Estimates of MWTP from previous studies have been mixed. While Smith and Huang (1995) report that most studies found a positive MWTP for improvement in air quality in their meta-analysis, in several cases the MWTP was negative. In many economic analyses, these negative results are described as “perverse” and excluded from further analysis (e.g. Zabel and Kiel 2000) or set to zero (e.g. Palmquist 1982) because they do not conform to theoretical expectations. In our analysis of the city submarkets, we found both positive and negative MWTP for air quality improvement. But, as noted above, the air pollution may be correlated with some other characteristic that homeowners value.

One would expect that proximity to parks would be a benefit, as they provide many ecosystem services such as recreation, greenery, access to biodiversity, and aesthetics. But, while living close to parks may provide easier access to these services, it may also increase the exposure to potential disamenities associated with parks, such as crime and noise. In their 2005 review, McConnell and Walls found examples of both negative and positive relationships between house price and park proximity. Troy and Grove (2008) demonstrated that consideration of neighborhood crime rates altered homeowners' willingness to live close to parks. In our study, we separated into two size categories with the expectation that people might derive different benefits from large vs. small parks. Small parks are more likely to have playgrounds and fields, while large parks are less congestible and may offer opportunities for hiking and access to desert flora and fauna. Thus, while we found a negative MWTP for proximity to small parks, large parks are considered an amenity for the metropolitan Phoenix area.

The disaggregation of the greater metro area into component cities gives further insight into the relative importance of the environmental and locational variables. The consistency of coefficient sign for vegetation abundance, bird abundance, and proximity to small parks highlight their importance across the entire metropolitan area. The existence of cases where the sign changes depending on city, such as PM_{10} or plant richness reinforces the notion that there are separate markets at play. Tiebout (1956) suggested that municipalities may offer unique packages of public goods. The homeowners in our study may be sorting into cities that best provide their desired environmental and locational amenities, depending on their needs.

Our findings confirm the importance of water-related environmental amenities in a desert environment. Vegetation abundance, lower August temperatures, and proximity to water are all amenities, reflecting the influence of the hot desert climate on homeowner choice. Vegetation and water provide a cooling effect, mitigating the urban heat island effect, and providing opportunities for recreation. Since climate models indicate that the region may become hotter and drier due to global climate change (Parry et al. 2007, Karl et al. 2009), and since Arizona may have to reduce its allotment under new rules for the Colorado River Compact signed in 2007 (Archibold 2007), these factors are likely to increase in importance. There may not be enough water to sustain lush vegetation and recreational lakes, and urban planners and managers may have to restrict 'luxury' water uses in favor of more 'basic' uses. Our findings suggest that if this is the case, the amenity value of water-related ecosystem services in areas that continue to enjoy the benefits of historic water access rights may increase significantly.

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Table 3.1: List of variables and their sources for the hedonic model with predicted relationship to the dependent variable, house price. MCA = Maricopa County Assessor, CAP LTER = Central Arizona – Phoenix Long Term Ecological Research Project, ISSR = Institute for Social Science Research at Arizona State University, MAG = Maricopa Association of Governments.

Variable	Description	Year	Source	Expected Relationship to House Price
House Characteristics				
ASPHALT	Presence/Absence of Asphalt Roof	2005*	MCA	
TILE	Presence/Absence of Tile Roof	2005*	MCA	
CONST_YR	Construction Year	2005*	MCA	Positive
GARAGE	Presence/Absence of Garage	2005*	MCA	Positive
POOL	Presence/Absence of pool	2005*	MCA	Positive
AREA	Size of lot, in sq. ft.	2005*	MCA	Positive
SQ_FT	Size of house, in sq. ft.	2005*	MCA	Positive
LIV_RATIO	Ratio of house size to lot size	2005*	MCA	Negative
Non-Structural Characteristics				
PM10	Particulate Matter ≤ 10 µm, concentration (µg m^{-3})	2000	CAP LTER	Negative
AUGMIN	Average Minimum August Temp, °C	2000	CAP LTER	Negative
VEG_ABUND	Vegetation abundance based on Soil-Adjusted Vegetation Index	2000	CAP LTER	Positive
BIRD_ABUND	Bird Abundance	2000	CAP LTER	Positive
PLANT_RICH	Plant Richness (Diversity)	2000	CAP LTER	Positive
IRRIGATION	Designated flood-irrigated area	n.d.	ISSR	
Distance Variables, all in ft				
STREAM	Distance to nearest stream		CAP LTER	Negative
CANAL	Distance to nearest canal		CAP LTER	Negative
PHX	Distance to Phoenix Sky Harbor airport (Central Phoenix)	1999	City of Phoenix	Negative

Variable	Description	Year	Source	Expected Relationship to House Price
WATER†	Distance to nearest water, excluding Salt River	2000	CAP LTER	Negative
SMPK‡	Distance to nearest small park (<250 acres)	2000	MAG	Negative
LGPK	Distance to nearest large park (>250 acres)	2000	MAG	Negative
DESERT	Distance to nearest desert area	2000	CAP LTER	Negative
Census Data, all by tract				
MED_AGE	Median age of residents	2000	US Census	Positive
HS_PER	% of population > 25yo who have only completed high school	2000	US Census	Negative
INCOME	Average household income	2000	US Census	Positive

* Year 2000 sales were selected from the 2005 Assessor's data

† Water in the Salt River is excluded from this measure due to industry associated with this area

‡ Large and small park areas derived from MAG active open space category.

Table 3.2: Descriptive Statistics of the Variables

Variable	Mean	Std. Dev.	Min	Max
House Characteristics				
HOUSE PRICE	167,334	96,157	60,040	749,900
CONST_YR	1984	16.11	1906	2000
AREA	9,111	7,275	1,612	231,453
SQ_FT	1,860	691	344	6,718
LIV_RATIO	0.23	0.085	0.01	0.97
Non-Structural Characteristics				
PM10	65.51	17.8	30	130
AUGMIN	22.23	0.56	19	23
VEG_ABUND	32,066	2,771	19,239	56,243
BIRD_ABUND	151.32	24.42	64.98	194.19
PLANT_RICH	0.426	0.159	0.030	0.900
Distance Variables, all in ft				
STREAM	2,049	1,889	0	12,950
CANAL	8,601	7,656	0	51,843
PHX	57,713	29,508	0	149,518
WATER	9,556	5,957	0	34,306
SMPK	3,675	4,180	0	41,980
LGPK	28,600	16,790	11	72,146
DESERT	14,930	10,343	94	45,097
Census Data, all by tract				
MED_AGE	34.37	8.72	21	76
HS_PER	0.223	0.067	0.065	0.396
INCOME	61,054	20,957	15,000	156,153

Table 3.3: Results of the Overall Regression for the Phoenix Metropolitan Area ($N = 38,333$, Adj. $R^2 = 0.871$)

Effect	Coeff.	Standard Error	Std. Coeff.	Tolerance	T	p-Value
CONSTANT	6.8559	0.2161	0.000	.	31.72	<0.0001
ASPHALT	-0.0538	0.0035	-0.057	0.25	-15.45	<0.0001
TILE	0.0798	0.0040	0.088	0.17	19.86	<0.0001
GARAGE	0.0379	0.0030	0.033	0.49	12.71	<0.0001
POOL	0.0544	0.0019	0.057	0.81	27.94	<0.0001
CONST_YR	0.0025	0.0001	0.088	0.21	22.19	<0.0001
AREA	0.0000	0.0000	0.062	0.46	23.02	<0.0001
SQ_FT	0.0004	0.0000	0.567	0.40	195.71	<0.0001
LIV_RATIO	-0.4289	0.0140	-0.084	0.45	-30.57	<0.0001
AUGMIN	-0.0026	0.0005	-0.001	0.93	-5.12	<0.0001
PM10	-0.0013	0.0001	-0.048	0.41	-16.82	<0.0001
VEG_ABUND	7.35×10^{-6}	3.6×10^{-7}	0.047	0.63	20.27	<0.0001
IRRIGATION	0.0493	0.0036	0.028	0.84	13.77	<0.0001
BIRD_ABUND	-0.0003	0.0000	-0.041	0.63	-17.54	<0.0001
PLANT_RICH	0.1576	0.0071	0.063	0.42	22.15	<0.0001
CANAL	0.0032	0.0008	0.009	0.74	4.15	<0.0001
STREAM	-0.0017	0.0006	-0.005	0.86	-2.67	0.0075
DESERT	-0.0023	0.0011	-0.005	0.73	-2.15	0.0317
PHX	-0.0842	0.0024	-0.124	0.27	-34.86	<0.0001

Effect	Coeff.	Standard Error	Std. Coeff.	Tolerance	T	p-Value
WATER	0.0041	0.0011	0.008	0.84	3.77	0.0002
SMPK	0.0096	0.0009	0.021	0.82	10.21	<0.0001
LGPK	-0.0105	0.0014	-0.018	0.55	-7.37	<0.0001
INCOME	3.5×10^{-6}	7.75×10^{-8}	0.161	0.27	45.33	<0.0001
MED_AGE	0.0067	0.0001	0.118	0.75	55.33	<0.0001
HS_PER	-0.7892	0.0210	-0.118	0.34	-37.55	<0.0001

Table 3.4: Regression Coefficients by City for Selected Environmental Variables

City	N	Adjusted R^2	PM_{10}	Vegetation Abundance	Bird Abundance	Plant Richness
Avondale	678	0.863	-0.003	n.s.	n.s.	-0.824
Chandler	3,055	0.878	-0.001	<0.001	n.s.	n.s.
Gilbert	2,626	0.857	0.002	n.s.	n.s.	-0.290
Glendale	2,951	0.894	n.s.	<0.001	-0.0014	-0.423
Goodyear	846	0.836	0.006	<0.001	n.s.	0.697
Mesa	5,522	0.880	0.002	<0.001	n.s.	0.255
Peoria	2,534	0.901	0.003	<0.001	-0.0052	0.220
Phoenix	11,727	0.859	-0.001	<0.001	-0.0003	n.s.
Scottsdale	3,324	0.845	-0.004	<0.001	-0.0008	n.s.
Surprise	1,357	0.704	n.s.	<0.001	-0.0035	n.s.
Tempe	914	0.831	0.002	<0.001	-0.0092	n.s.
Unincorp.	1,915	0.890	-0.002	<0.001	-0.0002	0.280

Table 3.5: Regression Coefficients by City for Selected Locational Variables

City	N	Adjusted R^2	Stream	Desert	Water	Small Park	Large Park
Avondale	678	0.863	0.014	-0.025	-0.052	n.s.	-0.957
Chandler	3,055	0.878	0.006	-0.087	-0.042	0.016	n.s.
Gilbert	2,626	0.857	0.007	n.s.	-0.026	0.009	0.116
Glendale	2,951	0.894	n.s.	-0.012	-0.017	n.s.	-0.027
Goodyear	846	0.836	0.010	n.s.	0.209	n.s.	-1.193
Mesa	5,522	0.880	n.s.	0.016	n.s.	0.008	-0.104
Peoria	2,534	0.901	-0.006	-0.017	-0.075	n.s.	-0.035
Phoenix	11,727	0.859	0.008	-0.010	0.006	0.007	-0.018
Scottsdale	3,324	0.845	n.s.	n.s.	-0.023	0.054	-0.061
Surprise	1,357	0.704	0.009	-0.057	n.s.	n.s.	-0.216
Tempe	914	0.831	n.s.	0.067	-0.045	n.s.	n.s.
Unincorp.	1,915	0.892	-0.009	n.s.	-0.012	0.025	0.012

Table 3.6: Willingness to Pay for Selected Environmental Variables

City	Mean House Price	PM_{10} Decrease of 10% Mean Value		Veg. Abundance Increase of 10% Mean Value		Bird Abundance Increase of 10% Mean Value		Plant Richness Increase of 10% Mean Value	
		Mean	WTP	Mean	WTP	Mean	WTP	Mean	WTP
Metro Area	$167,334	65.5	$1,425	32066	$3,944	151	-$760	0.43	$1,123
Avondale	$138,680	81.5	$3,351	30027	n.s.	165	n.s.	0.30	-$3,483
Chandler	$172,253	64.0	$1,125	31567	$2,440	70	n.s.	0.29	n.s.
Gilbert	$183,658	65.2	-$2,990	31755	n.s.	165	n.s.	0.22	-$1,188
Glendale	$148,965	61.6	n.s.	31777	$1,154	149	-$3,109	0.50	-$3,145
Goodyear	$168,146	67.5	-$6,810	29931	$6,349	152	n.s.	0.27	$3,166
Mesa	$157,366	57.2	-$1,639	31703	$2,477	77	n.s.	0.29	$1,183
Peoria	$163,258	54.9	-$2,473	30292	$5,793	145	-$12,301	0.48	$1,719
Phoenix	$144,642	72.7	$953	32532	$5,407	140	-$649	0.42	n.s.
Scottsdale	$311,808	53.3	$6,410	32257	$12,315	118	-$2,832	0.64	n.s.
Surprise	$156,984	37.1	n.s.	29594	$3,350	148	-$9,291	0.50	n.s.
Tempe	$167,368	88.4	-$3,345	33691	$6,436	176	-$26,453	0.34	n.s.
Unincorp.	$198,887	68.2	$2,441	32042	$3,575	120	-$457	0.49	$2,736

Table 3.7: Willingness to Pay for Moving 1,000 Feet Closer to Selected Locational Variables

		Streams		Desert		Water		Small Parks		Large Parks	
City	Mean House Price	Mean	WTP	Mean	WTP	Mean	WTP	Mean	WTP	Mean	WTP
Metro Area	$167,334	2049	$139	14930	$26	9556	-$72	3675	-$437	28600	$61
Avondale	$138,680	1355	-$1,453	4957	$706	5333	$1,394	3583	n.s.	52566	$2,525
Chandler	$172,253	1251	-$846	27616	$540	3982	$1,824	4246	-$630	47318	n.s.
Gilbert	$183,658	1238	-$1,026	32591	n.s.	5786	$822	1673	-$1,018	59575	-$359
Glendale	$148,965	2256	n.s.	11749	$147	10584	$242	3835	n.s.	18044	$223
Goodyear	$168,146	1793	-$968	7223	n.s.	21139	-$1,659	2149	n.s.	66173	$3,032
Mesa	$157,366	1180	n.s.	17224	-$145	13546	n.s.	2206	-$573	33310	$493
Peoria	$163,258	2177	$469	11362	$242	9862	$1,243	4552	n.s.	24342	$233
Phoenix	$144,642	2870	-$387	8260	$175	10457	-$82	3741	-$278	19463	$132
Scottsdale	$311,808	1273	n.s.	16461	n.s.	9149	$791	7398	-$2,257	25930	$734
Surprise	$156,984	1543	-$898	15717	$568	9193	n.s.	6651	n.s.	39167	$864
Tempe	$167,368	1445	n.s.	19228	-$581	5803	$1,298	2039	n.s.	16339	n.s.
Unincorp.	$198,887	1411	$1,261	14724	n.s.	9017	$258	4028	-$1,242	24894	-$98

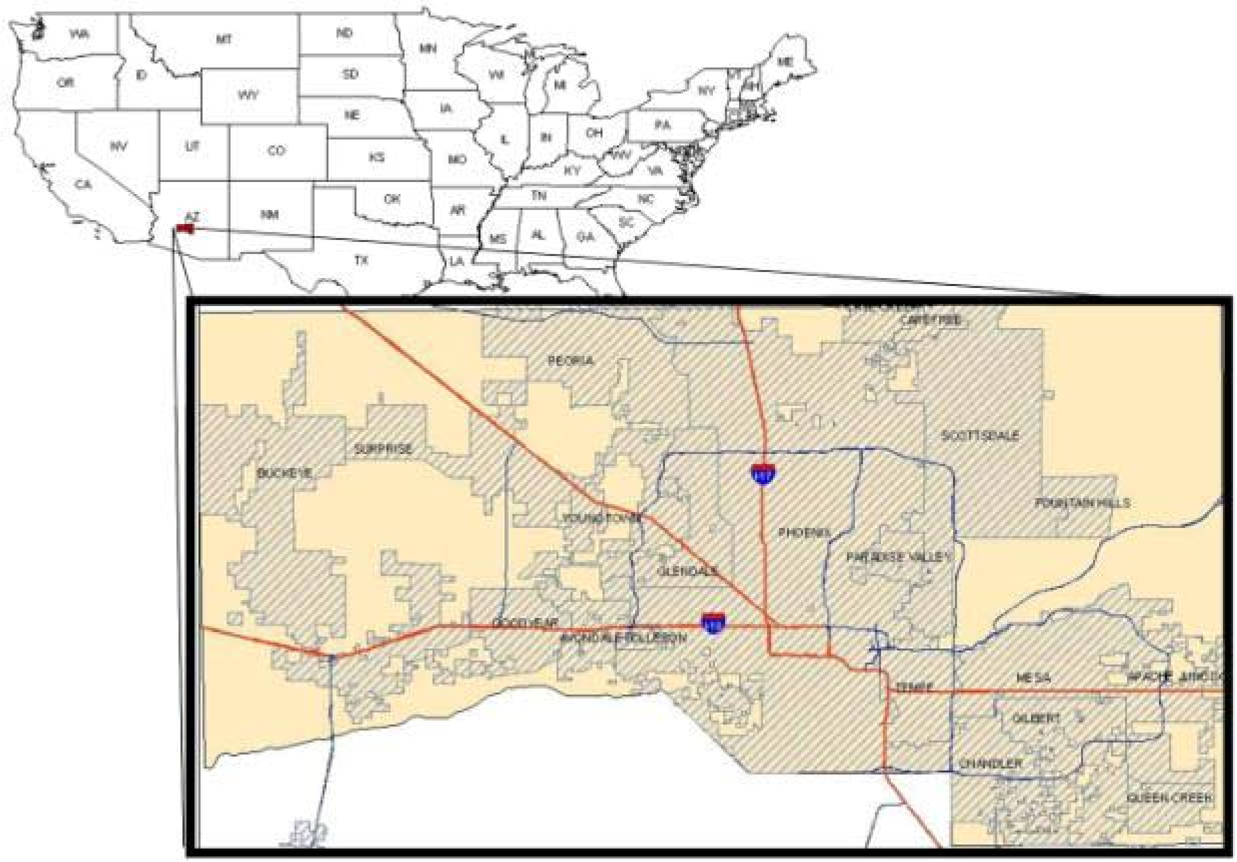

Figure 3.1: Map of the Central Arizona -- Phoenix Long Term Ecological Research Project area.

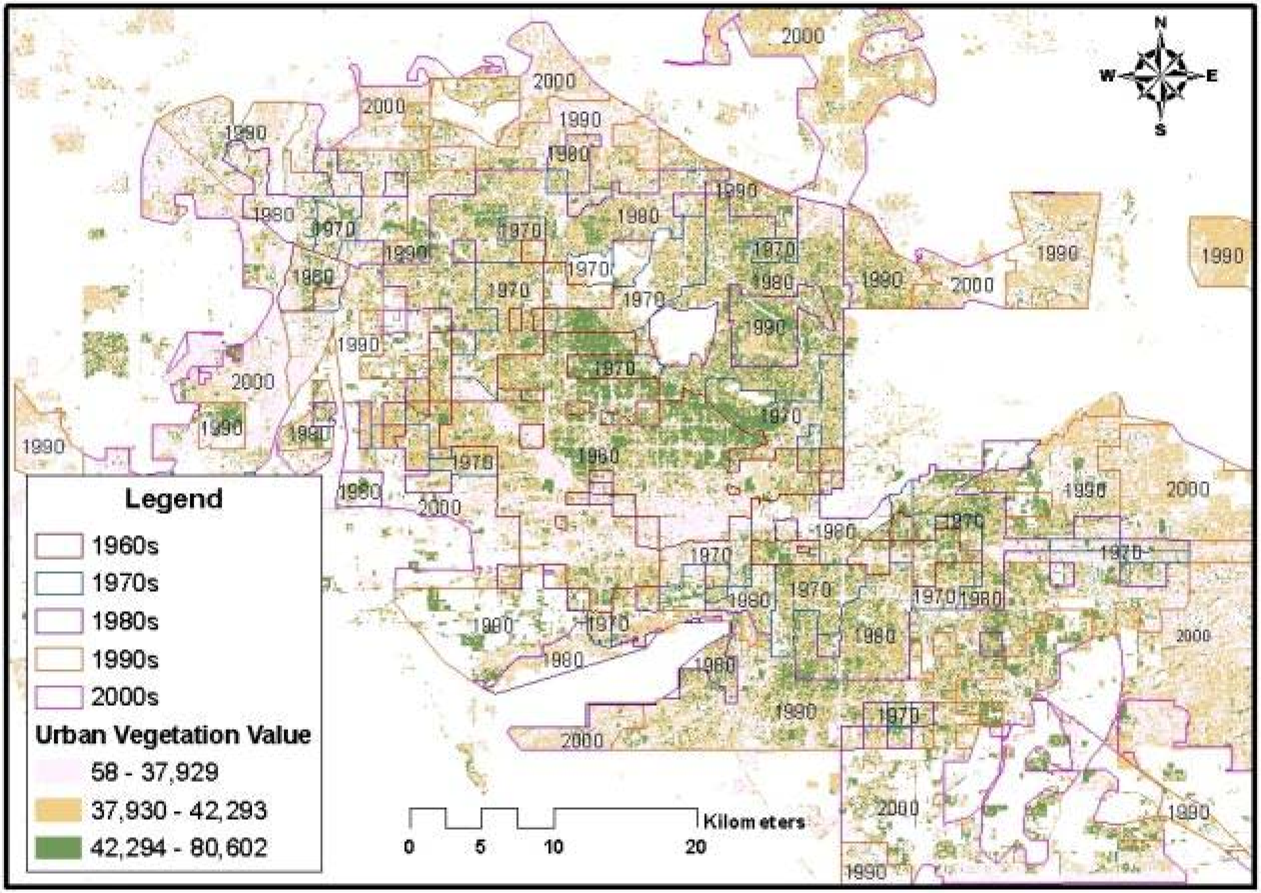

Figure 3.2: Map of the value of urban vegetation abundance and era of development.

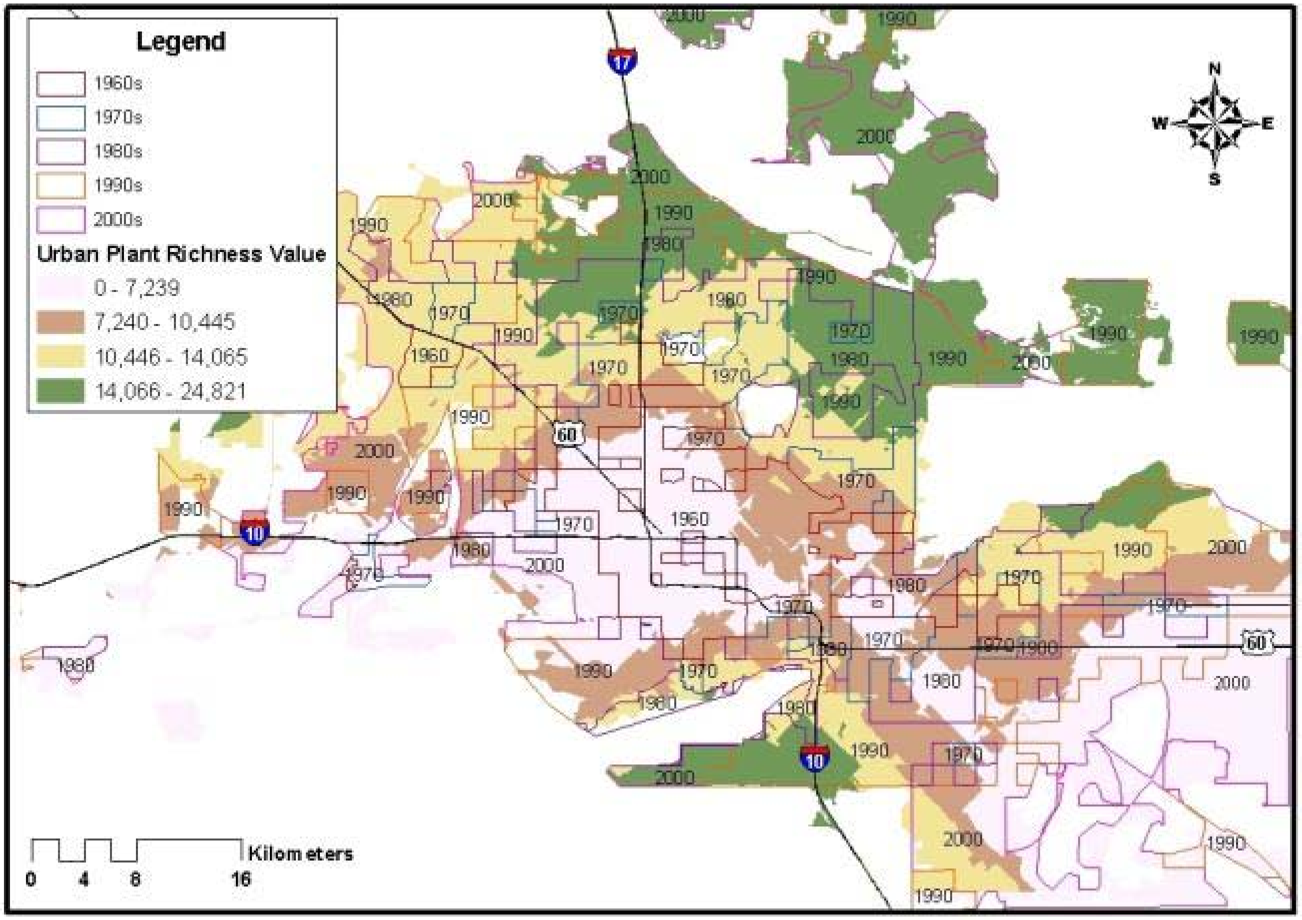

Figure 3.3: Map of the value of urban plant richness and era of development

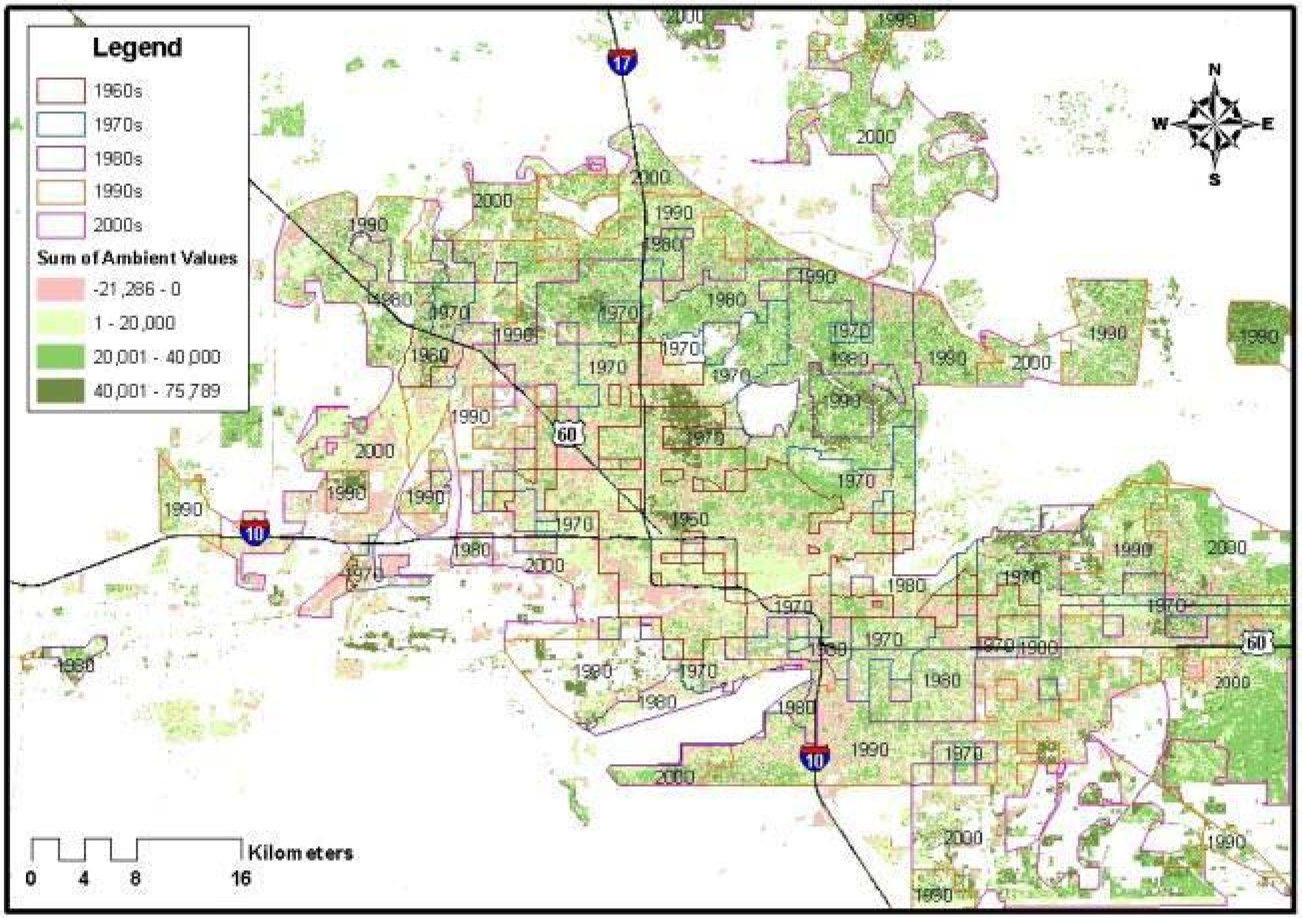

Figure 3.4: Map of the summed value of all ambient variables and era of development.

Chapter 4

LANDSCAPE DESIGN AND THE FATE OF NITRATE IN STORMWATER RETENTION BASINS IN THE PHOENIX, AZ METROPOLITAN AREA

4.1 Abstract

The process of urbanization has extensive and dramatic impacts on local ecosystems, altering hydrology and biogeochemistry. Within cities, novel ecosystems (such as parks, lakes, and stormwater retention basins) are created to provide specific ecosystem services locally to residents. However, often these ecosystems are designed for a single or very limited number of purposes, without consideration of the potential tradeoffs and synergies of ecosystem functions. Here, we investigate the potential impact of stormwater retention basins, designed mainly for flood abatement, on groundwater quality and regional nitrogen dynamics. We hypothesized that the two dominant landscape designs, grass and xeriscaping, would have significant impact on the fate of nitrate entering into these systems. We conducted a survey of 32 basins (16 grassy, 16 xeriscaped) which revealed that grassy basins have significantly higher soil water and organic matter content, correlated with higher denitrification enzyme activity (DEA, or potential denitrification) rates. We then focused on evaluating differences between basins with respect to water infiltrating through the soil profile towards groundwater. Our results indicate that grassy basins have rapid removal of nitrogen (N) in the top layer of soil, facilitated by regular irrigation and fueled by organic inputs from the grass. In an experimental manipulation, we flooded two

basins, one of each kind, and evaluated the changes in pools and fluxes of inorganic N (iN) via an addition of ^{15}N nitrate. The data suggest that, while both basin types export iN through infiltration and gaseous losses, the relative proportion differs by landscape design, with the grassy basin having a greater proportion loss via gas flux than infiltration. Grassy basins may be providing the additional ecosystem service of N removal, but they do so only with the associated costs of irrigation and maintenance. Planners and managers will need to consider these benefits and tradeoffs when designing multi-function, sustainable basins for the future.

4.2 Introduction

Water and nitrogen (N) are often limiting factors for biogeochemical processes in ecosystems. Over the past century, humans have assumed increased control over both of these essential materials. Vitousek et al. (1997) documented the dramatic impact of human activities on both the N and water cycles at a global scale. In urban ecosystems, N loading from anthropogenic sources (combustion, fertilizer use) is high, water delivery and stormflows are managed heavily, and runoff is significantly increased by impervious surfaces (Arnold and Gibbons 1996, Kaye et al. 2006). The combination of these modifications can result in altered N processing in cities and their surroundings (Wollheim et al. 2005, Kaye et al. 2006). To assess the fate of elevated N inputs to urban ecosystems, several researchers have constructed mass balances for nutrients. For the watershed that includes Phoenix, AZ, Baker et al. (2001) estimated that, during the mid 1990's, approximately 80% of the N inputs exited the ecosystem via harvest, downstream flow, and denitrification, but 20% of inputs were stored in vegetation and soils, or lost via underestimated gas flux. Results like these and others (Faerge et al. 2001, Boyer et al. 2002, Groffman et al. 2004, Wollheim et al. 2005) have been enigmatic, as unexpectedly high rates of retention and removal have been found despite reduction or elimination of sites and conditions where storage or denitrification is typically thought to occur (Paul and Meyer 2001). Therefore, many researchers are deconstructing the “black-box” city ecosystem, turning their attention to capacity of N retention and removal in component ecosystems of cities (Groffman et al. 2002, Zhu et al. 2004), and attempting to identify “hot spots,” locations with especially high processing rates (McClain et al. 2003). The process of urbanization not only alters existing ecosystems within and surrounding the city, but also creates novel ecosystems (Grimm et al. 2004, Palmer et al. 2004). Such designed ecosystems include aquatic features such as artificial lakes and wetlands, as well as stormwater retention basins or detention ponds. In Phoenix, the augmentation and redistribution of water has resulted in numerous component ecosystems that are atypical for a desert environment (Larson et al. 2005). Stormwater retention basins are a common designed ecosystem in the Phoenix metropolitan region. These ecosystems, depending on their design and maintenance, may provide the important ecosystem services of flood abatement and water quality improvement.

Even the simplest of ecosystems has multiple processes acting at several spatial and temporal scales—that is, there is not one static function of an ecosystem, thus specific ecosystem functions may be enhanced to provide more than one relevant ecosystem service. In cities, where management and design define ecosystem structure, some ecosystems are specifically designed for multiple uses (e.g., stormwater retention basins that are also playing fields), but maximizing one service often entails trade-offs with other services (Grimm et al. 2004, Tallis et al. 2008, Bennett et al. 2009, Nelson et al. 2009, Pejchar and Mooney 2009), which are rarely considered in urban ecosystem design and management. While stormwater retention basins in the Phoenix area were created to provide the ecosystem service of flood mitigation, they have potential to mitigate the high nutrient concentrations of urban stormwater. For example, Hope et al. (2004) found average nitrate (NO_3^-) concentration of runoff from parking lots in the Phoenix area to be 15.4 mg N/L, compared to 0.4 mg N/L in undeveloped desert sites. Therefore, as recipients of pollutant-laden stormwater, basin soils may also be purifying the infiltrating water; or removing nitrogen (N) permanently from the terrestrial ecosystem. While other communities have utilized retention basins for potential removal of pollutants in stormwater (Fischer et al. 2003, Dechesne et al. 2004, Birch et al. 2005), the current Arizona standards focus exclusively on water volume and do not mention water quality. This is true despite the fact that the US Environmental Protection Agency (US EPA) requires

Arizona to implement the National Pollution Discharge Elimination System (NPDES) stormwater-permitting program.

Researchers have found that retention-basin soils in mesic climates effectively remove pollutants within the top few centimeters (Datry et al. 2003, Birch et al. 2005, Dechesne et al. 2005), but the filtration capacity of basins in arid areas has not been fully investigated. In the Phoenix area, some developers choose to design multi-use basins, often sodding the area to produce a park or playing field requiring fertilization and irrigation. Other basins are xeriscaped with minimal vegetation (although sometimes drip-irrigated) on the upslope areas, and often are covered with decomposing granite (Figure 4.1). In addition, some retention basins have dry wells, which rapidly transfer water to deeper soil layers, while others rely on infiltration of soils and evaporation for water removal. While the frequency and distribution of these landscape designs is currently unknown, the selection and maintenance of these landscape design elements is likely to have significant ecological ramifications, which in turn will affect the ecosystem services provided by these basins.

In this research, we assess the influence of basin landscape design on N biogeochemistry and the ability of stormwater retention basins in the Phoenix metropolitan area to improve water quality by removing or retaining N. The presence of grass, with its requisite irrigation and probable fertilization, are *a priori* reasons we expect bulk soil characteristics to vary depending on landscape design. Because water, N, and availability of organic carbon (oC) are vital to biogeochemical processes, these soil properties are likely to affect ecosystem functions, such as denitrification (conversion of NO_3^- into the gaseous N forms of N_2 and N_2O, thereby removing N from the ecosystem) and other biogeochemical transformations. This, in turn, will influence the amount and form of N infiltrating out of the retention basin ecosystem (and thus may have an adverse effect on groundwater quality). We used a combination of synoptic surveys and an experimental flooding event to evaluate the following hypotheses:

> H_1: Grassy basins have higher soil water and organic matter content than xeriscaped basins, because of irrigation regimes and abundant vegetation.

These differences in bulk soil characteristics imply the following:

> H_2: Denitrification rates are higher in grassy basins than xeriscaped basins because of the availability of oC and prevalence of anoxic conditions during irrigation.

> H_3: Export of inorganic N via infiltration is higher in xeriscaped basins because of the lower rates of denitrification and uptake.

These differences in biogeochemical process rates influence the ratio of export to removal in grassy versus xeriscaped retention basins. Denitrification, by converting NO_3^- into biologically unavailable gaseous N forms, effectively removes N from not only the local retention basin ecosystem, but also from the greater urban ecosystem in which the basin is embedded. If retention basins support high rates of denitrification, and are a common occurrence in the urban landscape, they may be hot spots for N removal in the Phoenix metropolitan area and, by extension, in similar young, aridland cities.

4.3 Methods

Study Site Description

Located in the northern Sonoran Desert of the southwestern USA, the Phoenix metropolitan area receives approximately 180 mm of precipitation a year, with an average January temperature of 12º C and an average July temperature of 34º C (Baker et al. 2004). Most rain is concentrated in two seasons: a summer “monsoon” season with short, intense, localized thunderstorms and a winter rainy season characterized by frontal storms of longer duration and lower intensity. Given its hot, dry climate, the area experiences an average potential evapotranspiration of two m annually.

The city is situated in an alluvial valley surrounded by rugged mountain ranges typical of Basin and Range topography (Jacobs and Holway 2004). Phoenix is one of the most rapidly growing cities in the USA, increasing in population size from approximately 300,000 in 1950 to greater than 3.5 million inhabitants of more than 20 municipalities in 2004. Models predict that in 2025, the population will exceed 6 million inhabitants, representing a 280% increase since 1980 (Jacobs and Holway 2004), and nearly all of the undisturbed and agricultural lands will be developed into urban land uses (Jenerette and Wu 2001). With few geographical barriers to expansion, growth has been largely in an outward direction, estimated at almost one half-mile per year (Gober and Burns

2002). Most new construction has been the result of conversion of agricultural to residential use, but increasingly, new areas of desert are being transformed into housing developments. The region is the focus of the Central Arizona – Phoenix Long Term Ecological Research project (CAP LTER).

Stormwater retention basins are a common feature in the Phoenix metropolitan area (see Chapter 2). The Arizona Department of Water Resources (ADWR) in 1999 issued standards and procedures for stormwater detention and retention for all new developments except those that are *i*) single residential lots not associated with a subdivision, *ii*) residential subdivisions with lots ≥ 1 acre in area, or *iii*) projects smaller than 160 acres which drain *directly* into a watercourse intercepting a drainage area of > 100 square miles (ADWR 1999). Counties and individual municipalities may create their own drainage standards as well. The Maricopa County drainage regulations require that onsite retention (in natural depressions, man-made basins, or depressed parking areas) must be able to handle a 2-h, 100-y runoff event without ponding for longer than 36 h (MPDD 2004). Some retention basins are part of a neighborhood drainage network, while others serve a single lot. For this study, all of the basins were either in parks or on school property in the cities of Phoenix and Tempe. This minimized the number of necessary permissions and helped control for potential differences in maintenance (e.g. frequency of mowing, fertilization).

Synoptic Field Sampling

We conducted an extensive biogeochemical survey of stormwater retention basins in July 2006, after at least one month of no precipitation. We collected soil samples from 32 basins (factorial combination of grassy/xeriscaped and drywell/no well, 8 replicates). At each basin, four 5-cm diameter by 8-cm depth cores were taken and composited for analysis of extractable NO_3^-, ammonium (NH_4^+), chloride (Cl^-), soil organic matter, texture, bulk density, and denitrification potential.

We then selected 10 of these 32 basins (5 grassy and 5 xeriscaped), all located on school properties in Phoenix, for more intensive investigation of possible N storage in soil or transmission to groundwater. For this assessment we used a method similar to one employed by several authors (e.g., Walvoord et al. 2003, Deans et al. 2005, Seyfried et al. 2005), in which the Cl^- serves as an inert tracer indicating net water movement through the soil profile. If there are no peaks in concentration for either Cl^- or NO_3^- in the profile, then over time both Cl^- and NO_3^- must be infiltrating to deeper storage or recharging groundwater (Fig. 4.2A). Both Cl^- and NO_3^- peaking at approximately the same depth indicates soil storage at that depth, due to insufficient infiltration further downwards (Fig. 4.2B). If Cl^- peaks and NO_3^- does not, then some process (denitrification, assimilation, or abiotic adsorption) may be removing nitrate at that location (Fig. 4.2C). In April and May of 2007, after at least two weeks of no precipitation, we dug a pit in each basin to 45 – 60 cm depth and delineated the horizons. We then collected samples at the following depths: 0 – 5, 5 – 10, 10 – 15, 15 – 20, and then every 10 cm to the bottom of the pit. Sample extracts were analyzed for NH_4^+ in addition to Cl^- and NO_3^-, and soil moisture and organic matter content were measured.

Experimental Flooding with ^{15}N Enrichment

While the general survey and soil pits provided a snapshot of soil conditions in xeriscaped and grassy retention basins, we were especially interested in describing actual basin response to a storm event. Only direct measurements of infiltrating water quality and N gas production would sufficiently probe the relative differences in ecosystem functioning between the two types of basins and thus their potential capacity to provide the ecosystem services of water quality improvement and N. Therefore, we decided to experimentally flood two basins (one grassy, one xeriscaped, Table 4.1) by simulating a 1-in (2.54-cm) storm in the drainage area, which is approximately the size of a typical 5-y storm for this region. The basins were selected primarily for their ease of access and well defined drainage areas. To create the storm, we first set up a metered tap at a nearby fire hydrant, and then directed this high-pressure water in to a small pool to reduce the velocity. Then two sump pumps in the pool pumped the water back out via fire hoses into the two main inlets to the basin (as determined by curb cutouts and catch-basin location). The water from the hydrant meter was turned off after the appropriate volume of water had been transferred to the pool.

Nitrogen was also added to the system, in the form of NO_3^-, to approximate the load of 1 mg NO_3^-- N/L. Although higher concentrations of nitrate have been reported for runoff samples in the area, frequently these samples are taken during the "first flush," when N on the land surface is first mobilized by the runoff. However, concentrations tend to drop significantly afterwards, resulting in a lower mean storm event load (Bertrand-Krajewski et al. 1998). Since we were also interested in measuring N gas fluxes, a portion of the applied NO_3^--N was labeled with ^{15}N isotope. Because the atmospheric N_2 content is so high, small fluxes of N_2 would be impossible to measure accurately. Enrichment of the applied NO_3^--N with the relatively rare ^{15}N isotope would allow us to trace the transformation of NO_3^- through the ecosystem, especially via denitrification, whose end product is N_2 gas. After calculating the mass of NO_3^--N in the 1-in storm load, we added an enrichment of ^{15}N at 5000‰. To ensure uniform distribution throughout the approximate wetted area of the retention basin, we added the NO_3^--N (and $^{15}NO_3^-$-N) via a backpack sprayer before adding water to the system.

Prior to the beginning of the experiment, lysimeters were installed in the middle of the basin at a depth of 20 cm. A shallow plastic tray was filled with acid-washed gravel and covered with a screen. At one bottom corner of the tray, a hole led to tubing which fed into a 1-L bottle buried at a lower depth. Another tube extended above the soil to allow sampling from the bottle via peristaltic pump. Background soil core samples (5-cm diameter x 8-cm depth) were taken at

3 locations just prior to flooding to establish baseline N and ^{15}N content. After approximately 1 inch water had ponded in the basin (after ~0.5 h of addition), water samples were taken at 3 locations in the basin periodically at 0.5 h and then hourly (up to 9 h for the xeriscaped basin, and 7 h for the grassy basin).

In the xeriscaped basin, there was an outfall at one end of the basin leading to a drywell. During the time that water was flowing into this pipe, samples were taken at the same intervals as the 3 regular transects, as well as depth, width and velocity measurements needed to calculate discharge. Lysimeter samples were taken periodically when enough water had accumulated in the sample reservoir to pump out. Separate samples were also taken for analysis of concentration of dissolved N gases as well as ^{15}N:^{14}N. In each basin, transects were established and monitored periodically to determine basin water volume by measuring the width, length, and average depth. Air and water temperature were recorded at each sample time step.

By the next morning, all water had infiltrated into the soil. Soil core samples were again collected, as well as lysimeter samples, if possible. We then installed small chambers at 3 locations to measure N_2 and N_2O gas fluxes from the soil. A 25-cm diameter cylinder with a height of 20 cm was pressed in to the soil, and a tight-fitting lid applied for a 0.5-h incubation period (Rochette and Eriksen-Hamel 2008) at two times: 0800 and 1100. A port on the lid allowed for gas sampling into pre-evacuated 12-ml Exetainer® tubes (Labco Ltd, High Wycombe, UK). A sample was taken at the beginning and end of the incubation to calculate the rate of accumulation of $^{15}N_2$ and $^{15}N_2O$ gases. To avoid air contamination, Exetainer® tubes were stored in a centrifuge tube filled with water. We also conducted plot-level water additions, similar to the above flooding experiments, at 10 basins (5 grassy, 5 xeriscaped), but do not present the results here (see Appendix IV).

Soil and Water Chemistry Analyses

Before any analyses, all soil samples were sieved through a 2.0-mm mesh and homogenized. For the initial survey of 32 basins, we used water (rather than a strong salt solution) to extract cations and anions from the soil, because we were interested in the amount of inorganic N immediately available for export or biogeochemical processing following a rain event. Approximately 10 g of field-moist soil were shaken with 50 ml of deionized water for 1 h. The solutions were then filtered through pre-ashed and leached glass fiber filters (Whatman GFF). Deionized water was used to extract Cl^- for the depth profile samples as well. For the inorganic N extraction in the depth-profile analysis and basin flooding experiments, we used a 2M KCl solution for all extractions. Filtrate samples were stored at 4° C until they were analyzed colorimetrically on a Lachat QC8000 (Lachat Instruments, Loveland, CO) autoanalyzer for nitrate+nitrite (hereafter NO_3^-) and NH_4^+ at the Goldwater Environmental Laboratory at Arizona State University (protocols available online at http://www.asu.edu/gel/pdf/qc8000protocol.pdf).

For analysis of soil moisture content, sub-samples were weighed at field conditions and then dried for at least 24 h at 105º C to constant mass. We also used the data on gravimetric water content to calculate bulk density, calculated as the amount dry mass per sample volume. To determine soil organic-matter content, these samples were then ashed in a muffle furnace for 4 hours at 450º C, and soil organic matter was calculated as mass lost on ignition. For texture analysis, we used the LTER method as described in Elliot et al. (1999). Approximately 40 g of sieved, air-dried soil were weighed into a container along with 100 ml of sodium hexamatophosphate solution (50 g L^{-1}). Samples were shaken overnight, and then decanted into 1-L graduated cylinders. Deionized water was added to the 1 L mark, samples were agitated to achieve thorough re-suspension, and then left undisturbed for 7 h. Readings were then taken with a hydrometer (and temperature-corrected) to determine clay content. Sand content was assessed by sieving the sample again through a 0.5-mm screen and drying. Silt content was determined by difference.

Sub-samples of soil were also used to determine denitrifying enzyme activity (DEA), a measure of denitrification potential or the maximum rate of denitrification possible under ideal conditions, using the acetylene block method with amendments as described by Groffman et al. (1999). Briefly, 50 mg of sediment were added to Wheaton® bottles fitted with septa, and then amended with 50 ml of media providing NO_3^- (100 mg N/kg sediment), dextrose (40 mg/kg sediment), and chloramphenicol (10 mg/kg sediment), the last added to prevent enzyme synthesis. Samples were then well mixed and flushed with N_2 gas to create anoxic conditions, with the remaining headspace filled with N_2. Ten ml of acetylene were then injected through the septum. Pressure was equalized by briefly puncturing the septum with an open needle. Gas samples were taken immediately after acetylene injection, and then again 4 h later. Gas samples were analyzed on Shimadzu GC-14A with Poropak Q 50/80.

Water samples from the flooding experiment were put on ice after collection and then filtered later the same day with pre-ashed glass-fiber filters (Whatman GF/F). Filtrate was then stored at 4° C until analysis. Filtered water samples were analyzed for NO_3^-, NH_4^+, and Cl^- colorimetrically on a Lachat QC8000 (Lachat Instruments, Loveland, CO) at the Goldwater Environmental Laboratory at Arizona State University.

^{15}N Isotope Analysis

For soil extracts and water samples, we used a modified version of the diffusion method described by Stark and Hart (1996). Briefly, aqueous NO_3^- and NH_4^+ were converted to ammonia (NH_3^+) in a sealed sample cup containing a filter packet comprised of acidified paper discs enclosed between two polytetrafluoroethylene (PTFE) filters of 1 μm pore size (Millipore, Billerica, MA). The filters prevented contact between the discs and solution (which would have neutralized the acid on the discs), but allowed the NH_3^+ to diffuse through the filters onto the discs. We conducted sequential diffusions, first adding magnesium oxide (MgO) to the solution to convert the NH_4^+ to NH_3^+. Sample cups were sealed and gently agitated regularly for at least 6 days. After these filter packs were removed, new filter packs were inserted, along with Devarda's alloy, which is a reducing agent that converts NO_3^- to NH_4^+. This NH_4^+ was converted NH_3^+ because of the MgO remaining in solution. Sample cups were again sealed and agitated regularly for at least 6 days. Filter packs, once removed, were dried in a dessicator, and then the paper discs were packed into tin capsules. ^{15}N isotope analysis was conducted by the Stable Isotope Analysis Laboratory at Utah State University, using continuous-flow direct combustion coupled to isotope-ratio mass spectrometry on a Europa Scientific SL-2020 system.

Dissolved gas samples were collected via a modified helium headspace-equilibration method as described by Mulholland et al. (2004). Briefly, 120 ml of water was collected in a syringe and all bubbles expelled. To avoid air contamination, syringes were then place in a bucket filled with water. Underwater, 20 ml of helium was transferred into the sample syringe, the sample was shaken vigorously for 5 min to allow for equilibration of the dissolved gases in the syringe headspace, and finally, again underwater, the headspace from the syringe was injected into a pre-evacuated Exetainer® tube (Labco Ltd, High Wycombe, UK), which was stored as described previously. Both dissolved and chamber gas samples were analyzed for N_2 and N_2O concentration and ^{15}N:^{14}N for each gas species at the Stable Isotope Facility at the University of California, Davis. Trace-gas isotope ratios were measured using a SerCon Cryoprep trace-gas concentration system interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (Sercon Ltd., Cheshire, UK).

Calculations and Statistical Analyses

For the depth-profile investigation, in addition to presenting the raw results of N concentration by depth, we wanted compare measured N to expectations indicated by patterns of Cl^- concentration with depth. Therefore we assigned the Cl^- concentration at the lowest depth as the "background" and then calculated the proportion of Cl^- at each shallower depth as a proportion of this base Cl^- concentration:

$$[Cl^-_i]/ [Cl^-_b] \tag{1}$$

where *i* is the depth of interest and *b* is the lowest depth. We then consider the NO_3^- at the lowest depth also to be the "background" NO_3^- concentration, and then estimate expected NO_3^- at each depth by the following equation:

$$[NO_3^-{}_b] * [Cl^-_i]/ [Cl^-_b] \tag{2}$$

This equation predicts what NO_3^- concentrations would be at each depth if no biogeochemical processing had occurred and infiltration depth was the only determinant of NO_3^- concentration.

In the retention basin flooding experiments, we use an Explicit Green-Ampt model to estimate infiltration rates of standing water in the basin (Williams et al. 1998). The components of this model are the following equations:

$$\frac{q}{K_s} = \frac{\sqrt{2}}{2}\tau^{-1/2} + \frac{2}{3} - \frac{\sqrt{2}}{6}\tau^{1/2} + \frac{1-\sqrt{2}}{3}\tau \tag{3}$$

$$\chi = \frac{(h_s - h_f)(\theta_s - \theta_0)}{K_s} \tag{4}$$

$$\tau = \frac{t}{t + \chi} \tag{5}$$

where q is the infiltration rate (cm/h), K_s is the saturated hydraulic conductivity (cm/h), t is time (h), h_s is ponding depth or capillary pressure head at the surface (cm), h_f is capillary pressure head at the wetting front (cm), θ_s is saturated volumetric water content (cm^3/cm^3), and θ_0 is the initial volumetric water content (cm^3/cm^3). We did not measure K_s, h_f, or θ_s in the field; rather, we used typical values as reported in Williams et al. (1998). We had used a tension infiltrometer at eight basins to calculate Ks, but we unable to get convergent calculations at most locations (see Appendix III).

Insufficient $^{15}N_2$ or $^{15}N_2O$ evolved into the standing water during the flooding experiment to estimate denitrification rates by isotope pairing (Appendix II, Table 1). Instead, we used an iterative model developed by Laursen (2008) for calculating denitrification rates in standing water based on changes in N_2 and N_2O concentrations over time. This model accounts for gas exchange with the atmosphere and is adjusted for windspeed, humidity, pressure, and temperature. Beginning with the initial N_2 concentration, the model calculates subsequent concentrations at 1-minute intervals. The user adjusts the denitrification rate until the predicted N_2 concentration at the end of the time period matches the measured N_2 concentration for the same time. For the xeriscaped basin, we also adjusted the calculations to account for water and N loss from the system in the water flowing to the pipe. We measured air and water temperature in the field, and used historical data for mean windspeed, humidity and pressure. Gas samples taken from chambers after water infiltrated the soil (Appendix II, Table 2) did have enough ^{15}N signature to allow for calculation of denitrification via the “non-equilibrium” technique as described by Bergsma et al. (2001).

To evaluate the fate of nitrogen in response to the experimental flooding, we calculated an event budget for inorganic N for each basin. Initial pool size was determined by multiplying the mean concentration by the volume of soil calculated from the bulk density, maximum wetted area and to a depth of 8 cm. The N load to each basin was calculated by multiplying the background water NO_3^- concentration by the volume of water, plus the total g N added via the sprayer addition. The infiltration volumes, calculated via the Green-Ampt equation, were combined with N concentration data from the lysimeters to estimate N losses via infiltration. For the xeriscaped basin, we also accounted for loss of N via water flowing into the pipe.

To estimate the gas losses from the standing water and wetted soil, we made the assumption that the rates for standing water, calculated via the iterative denitrification model, are an approximation the first 12 h and wetted soil rates, calculated by the ^{15}N non-equilibrium equations, for another 12 h, allowing for the construction of an event budget of 24 h duration. For each basin, we evaluated the sensitivity of the budget for the assumptions made both in the budget itself and in the models contributing to the budget calculations.

Differences between grassy and xeriscaped soil characteristics were tested using ANOVA. We used a General Linear Model (GLM) to explore the relationship between DEA and soil characteristics by landscape design. Variables were log or square-root transformed to reduce heteroskedasticity, when appropriate. When assumptions about normality and/or homogeneity of variances were violated, we used the non-parametric Mann-Whitney U test. For all tests α = 0.05.

4.3 Results

General Soil Characteristics and DEA

As expected, there were marked differences in between xeriscaped and grassy retention basins soils (Table 4.1). The presence/absence of a drywell had no significant impact on any measured soil characteristics (data not shown). The bulk density of soils in both types of basins averaged 1.2 g cm^{-3}, but was highly variable. In grassy basins, gravimetric water content ranged from 4.6 to 29.4 percent, while soil organic matter content ranged from 3.3 to 10.3 percent. The means for these characteristics were significantly higher than those in xeriscaped basins (Fig. 4.3), which had ranges of 1 – 17 % for soil moisture and 1.2 – 4.7%, soil organic matter, respectively. However, mean NO_3^- and NH_4^+ concentrations did not differ significantly by landscape design (Fig. 4.4). When compared to the 2005 CAP LTER soil characteristics by most common land use (as referenced in Figure 4.10), NO_3^-concentration in retention basins was similar, but SOM differed ($F_{(4,94)} = 18.18$, $p < 0.0001$). Post-hoc tests revealed that grassy retention basins had higher SOM than xeriscaped basins and all CAP LTER common land uses.

Grassy retention basins had significantly higher DEA rate than xeriscaped basins ($F_{(1,29)} = 14.9$, $p = 0.0006$). Denitrification rate ranged from 407 to 1251 μg N_2O-N kg soil^{-1} h^{-1} in grassy basins, and from below the detection limit to 1090 μg N_2O-N kg soil^{-1} h^{-1} in xeriscaped basins. Considering all basins together, in the multiple regression percent gravimetric water was the only significant predictor for DEA (Adjusted $R^2 = 0.439$, $p = 0.0004$, Fig. 4.5), while the effects of SOM content and NO_3^- concentration were not significant. When considering grassy and xeriscaped basins separately, both gravimetric water content and NO_3^- concentration were significant factors for DEA (Adjusted $R^2 = 0.6374$ $p = 0.0015$) in grassy basins, while none of the variables were significant predictors for DEA in xeriscaped basins.

Depth Profiles

Grassy and xeriscaped basins had distinct soil profiles, especially in the top layers (Appendix I, Fig. 4.6). In the uppermost layer, xeriscaped basins had a wider range of NO_3^-, from 4.1 to 17.5 mg N/kg soil with a mean of 8.92, while grassy basins were more constrained: 0.46 to 2.47 mg N/kg soil with a mean of

0.49. Two-way ANOVA indicated that both depth ($F_{(7,62)} = 14.33$, $p < 0.01$) and design ($F_{(1,62)} = 56.34$, $p < 0.01$) had an effect on NO_3^- concentration, and the interaction term was insignificant ($F_{(7,62)} = 0.6$, $p = 0.75$). For Cl^-, both depth ($F_{(7,62)} = 7.49$, $p < 0.01$) and design ($F_{(1,62)} = 76.54$, $p < 0.01$) were significant, along with the interaction term, which was positive($F_{(7,62)} = 3.6$, $p = 0.003$). Grassy basins had much higher mean Cl^- concentrations, 131 mg/kg soil (ranging from 51 to 235), than xeriscaped basins, with 14 mg/kg soil (ranging from 9 to 45). Below the top layer, differences between by landscape design were not significant for NO_3^-, while the higher Cl^- concentrations persisted through the top 15 cm for grassy basins. When comparing the measured NO_3^- concentration with the concentration predicted by Cl^- (water movement alone), in grassy basins predicted and measured NO_3^- were generally the same throughout the profile (Fig. 4.7). The same was true for xeriscaped basins except that the measured NO_3^- in the top layer was well above the predicted estimate.

Experimental Flooding

Prior to flooding, the grassy basin had a mean NO_3^- concentration of 11.05 mg N/kg soil, while the xeriscaped basin had 3.47 mg N/kg soil (Tables 4.2, 4.3). This difference was not significant (Mann-Whitney $U = 6$, p = 0.08), however, due to high variance in the grassy basin. For NH_4^+, the two basins had statistically similar concentration (6.51 and 4.41 mg N/kg soil, respectively, $F_{(1,4)} = 1.5915$, p = 0.2757). After the addition, the grassy basin had a lower NO_3^- concentration (3.24 mg N/kg soil), and the xeriscaped basin had a higher NO_3^-

concentration (4.03 mg N/kg soil), but neither of these changes was significant (grassy $F_{(1,3)} = 8.43$, $p = 0.062$; xeriscaped $F_{(1,4)} = 0.22$, $p = 0.66$).

Based on the calculations of 5000‰ ^{15}N, the amount of ^{15}N applied in both flooding experiments was very small when compared to background values. For the grassy basin, we applied only 0.8 g ^{15}N, when the total background pool was 2880 g ^{15}N. In the xeriscaped basin, we applied 0.83 g ^{15}N, and the background pool was 321 g ^{15}N. For this reason, it was not possible to calculate recovery in these experiments, and we can assume that any isotope captured in soil or water was probably from the background abundance.

We were able to collect lysimeter samples from both the grassy and xeriscaped basins during the addition. In the grassy basin, lysimeter water samples were taken at 4 h and 7 h post addition, and in the xeriscaped basin samples were taken at 4.5, 5.5, and 9.25 h post addition. Samples from the lysimeter in the grassy basin had mean NO_3^- and NH_4^+ concentrations of 0.46 (standard deviation, SD = 0.27) and 0.06 (SD = 0.07) mg N/L respectively, while the xeriscaped basin's mean NO_3^- and NH_4^+ concentrations were 2.46 (SD = 0.73) and 0.16 (SD = 0.31) mg N/L, respectively. For the xeriscaped basin, gas samples were also taken from the lysimeter water at 4.5 and 7.5 h, with mean concentrations of 3.01 (SD = 0.36) mg N/L for N_2, and 41.37 (SD = 16.73) ng N/L for N_2O (Table 4.4).

The basins also lost gaseous N from the standing water during and from the wetted soils after the flood event (Table 4.4). The xeriscaped basin had no detectable N_2 production during ponding, and afterwards had an estimated denitrification rate of 4.8 mg N m^{-2} d^{-1} for the wetted soil. Based on the iterative denitrification model, the grassy basin had a denitrification rate of 415.3 mg N m^{-2} d^{-1} while there was standing water. Using the ^{15}N non-equilibrium equations, the mean denitrification rate was 9.6 mg N m^{-2} d^{-1} after the water had infiltrated. The denitrification model for the xeriscaped basin yielded a negative rate of N_2 production (-4.1 mg N m^{-2} d^{-1}) in the standing water. The mean chamber denitrification rate calculated via the ^{15}N non-equilibrium equations was 4.9 mg N m^{-2} d^{-1}. Both basins had a relatively low N_2O production rate, always less than 0.1 mg N m^{-2} d^{-1}during ponding, and less than 0.02 mg N m^{-2} d^{-1} after the water had infiltrated.

With respect to the event budgets, the grassy basin was larger and had higher initial soil concentrations of nitrate and ammonium, resulting in much larger iN pools than the xeriscaped basin. The xeriscaped basin had a larger contributing area, so that the total N load in the simulated storm was higher than in the grassy basin, and proportionately much larger. In the grassy basin, N inputs were only 5% of the pre-addition soil iN. N inputs to the xeriscaped basin, on the other hand, were 132% of the pre-addition iN.

With respect to exports from the system, both basins lost NO_3^- and NH_4^+ via infiltration. If the lysimeter concentration measurements are representative of all water exported via vertical infiltration, the grassy basin lost approximately 31 g N, or 3.7% of the available inorganic N (background NO_3^- and NH_4^+, plus N added via the sprayer and water additions). The xeriscaped basin, on the other hand, lost approximately 79 g N via infiltration, or 42% of available iN (Table 4.5). The xeriscaped basin lost approximately 29 g N as N_2 and 0.001 g N as N_2O dissolved in the infiltrating water, or 21% of the available iN. Although dissolved gas samples were only analyzed from lysimeters in the xeriscaped basin, lysimeter N_2 concentrations were comparable to surface N_2 concentrations at the same time period. Lysimeter N_2O concentrations were roughly double that of the surface N_2O, but still very low (< 1 μg/L). If we assume that the pattern for lysimeter dissolved gases is similar to that of xeriscaped basin, then the grassy basin lost 181 g N as N_2 and 0.001 g N as N_2O, or 23% of the available iN. The total gas losses (via infiltration and upwards toward the atmosphere) are 264 g N for the grassy basin (or approximately 32% of the available inorganic N), and 29 g N for the xeriscaped basin (15 % of the available inorganic N).

At the end of the experiment (24 h post addition), both basins had lower total iN mass in the soil; these differences were not significant for the xeriscaped basin ($t = 0.1224$, $p = 0.91$), nor the grassy basin ($t = 1.5581$, $p = 0.22$). When comparing the proportions that NO_3^- and NH_4^+ represent of total iN, it is apparent that, after the flood, this proportion changed very little in the xeriscaped basin, while in the grassy basin NH_4^+ made up a larger proportion (Fig. 4.8), although this change was not significant ($t = -0.9884$, $p = 0.38$).

4.5 Discussion

Through an extensive survey and intensive investigations and manipulations, we have demonstrated significant differences in the ecological properties and functioning between grassy and xeriscaped stormwater retention basins. Our synoptic field sampling supported our hypotheses that grassy basins have higher soil water and organic matter contents, which sustain higher potential denitrification rates (DEA). Our reported DEA rates for grassy basins are on the high end of reported literature values, and are similar to results of Zhu et al. (2004) for other grassy retention basins and of Roach and Grimm (in review) for irrigated grassy floodplains, both in the same area as our study. Grassy basins are, of necessity, irrigated, and thus are much more likely to have periodic anoxic conditions in the soil, a required condition for denitrification to occur. The lack of significant differences between basin designs for soil inorganic N, despite the probable higher rates of N removal via denitrification, may be due to the heavy fertilization these basins typically receive. One facilities manager for school district properties reported that the landscaping maintenance teams apply approximately 7 g N m^{-2} y^{-1} to the grassy basins, and that clippings are not removed after mowing (D. Koontz, Paradise Valley Unified School District, personal communication). The frequent watering and high availability of organic matter support denitrification rates that, along with uptake and storage in the grass, potentially remove substantial amounts of N from the soil. The xeriscaped basins, on the other hand, typically do not have soil organic matter to sustain high levels of denitrification, even during wet conditions.

The depth profiles give further insight into the likely rapid N transformations and removal occurring in the top few cm of soil in grassy retention basins. The high levels of chloride in the top layers of grassy basins are a clear signature of the frequent watering they receive. We interpret the high average nitrate concentration in the top layer of soil in the xeriscaped basins as input to the ecosystem via N deposition. The concordance of predicted and measured nitrate throughout most layers indicates that nitrate is infiltrating the soil, and, in the case of the xeriscaped basins, the pulse of N input from the top has been flushed below the sampling depth. The grassy and xeriscaped basins are geographically interspersed and generally located fairly close to each other, and so are likely receiving similar inputs via N deposition. If this is the case, then the correspondence between predicted and measured nitrate in the top layer of the grassy basin soils suggests rapid removal of this excess N (as well as any inputs from fertilizer, as mentioned above).

By examining the pools and fluxes of iN from the experimental flooding manipulation of the basins, we gain insight into the similarities and differences of ecological functioning of the basins as influenced by basin landscape design. The key apparent difference between basins is the relative proportion of losses to different vectors. The relative proportions of N loss by infiltration of nitrate and ammonium vs. via gas evasion for each basin can be seen in Figure 4.9. The grassy basin attenuated most of the NO_3^- input by denitrification, but also lost some soil inorganic N via infiltration. The xeriscaped basin had much lower gas production, instead having higher proportional losses via infiltration. As concentrators of water and nutrients, retentions basins have the potential to “turn on” soil biogeochemical processes, but also to act as conduits for transport to groundwater. Several lines of evidence from our study that suggest that xeriscaped retention basins are more likely to act as the latter. There are significant observed differences in soil moisture and organic matter content, lower rates of DEA, apparent high rates of accumulation of iN in the top layers of the soil, which then infiltrates with little transformation through the soil towards groundwater.

There are several notes of caution to be considered when interpreting this data, especially since many of the variables for contributing models had to be assumed or based on literature values. For the Green-Ampt infiltration model, a sensitivity analysis revealed that estimates of K_{sat} were highly influential on model results, with impacts up to 61% of calculated infiltration volume. Other tested variables (standing head, saturated volumetric water content, air entry head, initial water content, and bulk density) had much lower effects, ranging from < 1 to 9%. Tracing the influence of K_{sat} through to the constructed budget, we calculated the highest K_{sat} possible that would not result in more water infiltrating through the basin than was added to it. In the case of the grassy basin, this alteration caused the ratio of total fluxes : total change in iN to increase to 1.18. In the xeriscaped basin, making the same alteration to the model increased the ratio to 165.18.

The Laursen model for denitrification relies on historic weather data, as we did not record wind speed and humidity during the experiments. However, increasing the windspeed by an order of magnitude had negligible effects on modeled denitrification rates. Similarly, doubling or halving the humidity had little impact. For this model, the larger influence on the event budget derives from the assumptions made regarding the duration of the modeled denitrification rate. We know that there was standing water in each of the basins for at least 7 h. If we change the calculations so that the denitrification rate for standing water was in effect for only 8 h, and chamber rates in effect the other 16, the total fluxes : total change for the grassy basin decreases to 0.87, but increases to 0.72 for the xeriscaped basin. Conversely, if the split of hours is the inverse, then the ratio increases to 1.07 for the grassy basin, and decreases to 0.63 for the xeriscaped basin.

And finally, another assumption we made was to correct for the lack of observations regarding gas concentrations in the lysimeter samples for the grassy basin. If, instead of being equal to the concentration in the overlying water (our assumption), the concentrations were only half, then the total fluxes : total change ratio decreases to 0.62. On the other hand, of the concentration of gases is double that of the overlying water, then the ratio increases to 1.66.

Generally speaking, it is clear that estimates of the event budget for the grassy basin are better constrained than the xeriscaped one. This may be due, in part, to the fact that the ecosystem has fewer vectors than the xeriscaped one, not having any pipe flow. Additionally, the grassy basin may have greater homogeneity of soil structure due to the stabilizing effect of grass. Anecdotally, we have observed that the xeriscaped basins frequently have problems with erosion of the sides.

It is useful to consider these results within the larger urban ecosystem. The CAP LTER has conducted a general field survey, including measures of soil nutrient status of 204 locations spanning the urban to rural interface of the Phoenix metropolitan area (Zhu et al. 2006). Of the urban locations (108 sites), the three most common land uses are xeriscaped residential (N = 30), mesic residential (N = 28) and commercial (N = 10). Zhu et al. (2004) conducted a study of soil characteristics and nitrogen removal capacity of eight grassy stormwater retention basins in the CAP LTER area. The mean nitrate concentrations for these 4 types of soils, along with the results of this study can be seen in Figure 4.10. When compared to the region’s most common urban soils, the nitrate concentrations of retention basin soils in Zhu’s 2004 study are roughly 5 to 7 times lower. The basins in this study had higher mean concentrations, although still significantly lower than mesic residential soils.

With respect to the larger urban ecosystem N budget, this research indicates that grassy basins are likely to be locations of permanent N loss from the system via denitrification. In the Phoenix-area N budget constructed by Baker et al. (2001), groundwater is included within the ecosystem boundary, so that N transported out of retention basins into groundwater via infiltration would still be considered to be retained within the ecosystem. This is especially appropriate since the region relies on groundwater for approximately 30 – 40% of its water resources. Therefore, grassy ecosystems could be providing an important ecosystem service by attenuating N inputs to groundwater, whereas xeriscaped basins may be localized, concentrated sources of inorganic N to groundwater.

Current research (see Chapter 2) indicates that retention basins are a ubiquitous feature in the Phoenix metropolitan area, but that the majority of them are likely to be xeriscaped rather than grassy. If planners, managers, and regulatory agencies wish to include water quality improvement as an ecosystem service provided by these basins, it follows that requiring new basins to be grassy, and converting existing xeriscaped basins to grass, would be the best approach. However, there are significant costs associated with such a plan. Grassy basins require water and maintenance, and possibly the addition of fertilizer. This research does not assess possible performance of grassy basins without water and fertilizer inputs. On the other hand, grassy basins have the potential to provide other benefits, such as recreation, mitigation of the urban heat island, and improved aesthetics. In cases such as this, Bennett et al. (2009) propose a useful typology for understanding the relationships among multiple ecosystem services. They propose that careful examination of both drivers and interactions will help identify ecological leverage points for increasing the resilience of coupled socio-ecological systems. In the context of this arid, rapidly growing city, investigating the impact of such drivers as increased water demand for municipal uses (whether due to population growth and/or climate change) or reductions in fertilizer use on multiple ecosystem services, will be critical, at the scale of both stormwater retention basins and the larger urban ecosystem. There may be synergistic or antagonistic interactions between ecosystem services that could influence planning and management of urban areas at multiple scales.

4.6 Acknowledgements

This work would not have been possible without field and laboratory help from Bony Ahmed, Stevan Earl, Kayla Graham, Rebecca Hale, Tamara Harms, Kathleen Lohse, Katie Mayer, Patrick Ortiz, Ryan Sponsellor, Jolene Trujillo, Christina Wong, and Lin Ye. Thanks to Doug Koontz from the Paradise Valley Unified School District, Dave Wheeler from the Kyrene School District #28, and Mark Richwine from City of Tempe Parks and Recreation Department for site access. This material is based upon work supported by the National Science Foundation under Grant No. 987612, IGERT in Urban Ecology, Grant No. DEB-0423704, Central Arizona – Phoenix Long-Term Ecological Research (CAP LTER), DDIG Award Grant No. DEB-0808524, and the C. Lisa Dent Memorial Fellowship.

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Table 4.1: Soil characteristics of grassy vs. xeriscaped retention basins in the Phoenix metropolitan area. N = 16 for both designs.

Design	Statistic	Bulk Density g/cm^3	Gravimetric Water %	Soil Organic Matter %	NO_3^- (mg N/kg soil)	NH_4^+ (mg N/kg soil)	DEA (µg N_2O-N/kg soil/h)
Grassy	Mean	1.2	17.0*	6.4*	8.96	1.11	673*
	Range	*(0.8 - 1.6)*	*(4.6 - 29.4)*	*(3.3 - 10.3)*	*(0.49 - 31.83)*	*(0.03 - 5.67)*	*(407 - 1251)*
	Std. Dev.	*0.3*	*0.07*	*0.02*	*8.16*	*1.31*	*257*
Xeriscaped	Mean	1.2	6.50*	2.70*	7.35	0.95	285$_*$
	Range	*(0.8 - 1.8)*	*(1.0 - 17.1)*	*(1.2 - 4.7)*	*(1.56 - 37.70)*	*(0.18 - 3.64)*	*(BDL$^\wedge$ - 1090)*
	Std. Dev.	*0.3*	*0.06*	*0.01*	*8.91*	*0.81*	*345*

* Significant at $\alpha = 0.05$

$^\wedge$ BDL= below detection limit

Table 4.2: Soil characteristics of grassy and xeriscaped retention basins selected for experimental flooding.

Design	Total Area m^2	Contributing Area m^2	Bulk Density g/cm^3	Gravimetric Water %	Soil Organic Matter %	NO_3^- (mg N/kg soil)
Grassy	858	2222	0.87	20.6	11.8	11.1
Xeriscaped	458	4341	1.6	2.5	1.9	3.4

Table 4.3: Mean nitrate and ammonium concentrations (mg N/kg soil) and $\delta^{15}N$ by basin design prior to and after flooding experiment. Standard deviation in parentheses.

		Nitrate		Ammonium	
Basin		Before	After	Before	After
Grassy	Concentration	11.05	3.24	6.51	8.24
		(3.63)	(2.54)	(2.39)	(3.78)
	$\delta^{15}N$	-13.82	12.46	3.51	11.65
		(4.61)	(6.57)	(1.26)	(7.29)
Xeriscaped	Concentration	3.47	4.03	4.41	3.62
		(0.55)	(1.98)	(1.61)	(0.63)
	$\delta^{15}N$	-14.86	56.69	-6.04	28.06
		(5.35)	(32.26)	(1.86)	(16.15)

Table 4.4: Lysimeter nitrate, ammonium, N_2 (all in mg N/L), and N_2O (in ng N/L) concentrations by time since addition for grassy and xeriscaped basins.

Basin Design	Time	Nitrate ----------	Ammonium mg N/L	N_2 ----------	N_2O ng N/L
Grassy	4	0.66	0.01	n/a	n/a
	7	0.27	0.11	n/a	n/a
Xeriscaped	4.5	3.31	0.19	2.76	29.54
	5.5	2.07	0.71	n/a	n/a
	7.5	n/a	n/a	3.26	53.20
	9.25	2.00	0.14	n/a	n/a

Table 4.5: Inorganic nitrogen (iN) budget for a flood event in grassy and xeriscaped retention basins in Phoenix, Arizona. Units are g N for pools and g N/event for fluxes.

	Grassy					Xeriscaped				
	NO_3^-	NH_4^+	N_2	N_2O	Total iN	NO_3^-	NH_4^+	N_2	N_2O	Total iN
Pools										
Background	497	293			791	35	45			80.5
Post Flood	146	371			517	41	37			78.2
Difference	*-351*	*77*			*-274*	*6*	*-8*			*-2.3*
Fluxes										
N Addition (Fertilizer)	30				30	49				48.7
N Addition (Water)	11	1			12	54	4			57.6
Lysimeter Losses	-31	-3	-190	<<1	-224	-74	-5	-29	<<1	-106.8
Standing Water Gas Losses			-81	<<1	-81			Bdl	<<1	<<1
Wetted Soil Gas Losses			-2	<<1	-2			-3	<<1	-3
Loss to Drain	n/a	n/a	n/a	n/a	n/a	-1	-0.1			-0.8
Total	*10*	*-2*	*-264*	*0*	*-265*	*27.8*	*-0.6*	*-28.9*	*0.0*	*-1.6*
Recovery: Total Fluxes/Total Change in iN					0.97					0.67

Figure 4.1: Examples of stormwater retention basins in Phoenix, which typically are of the xeriscaped (top) or grassy (bottom) types that were the focus of this study.

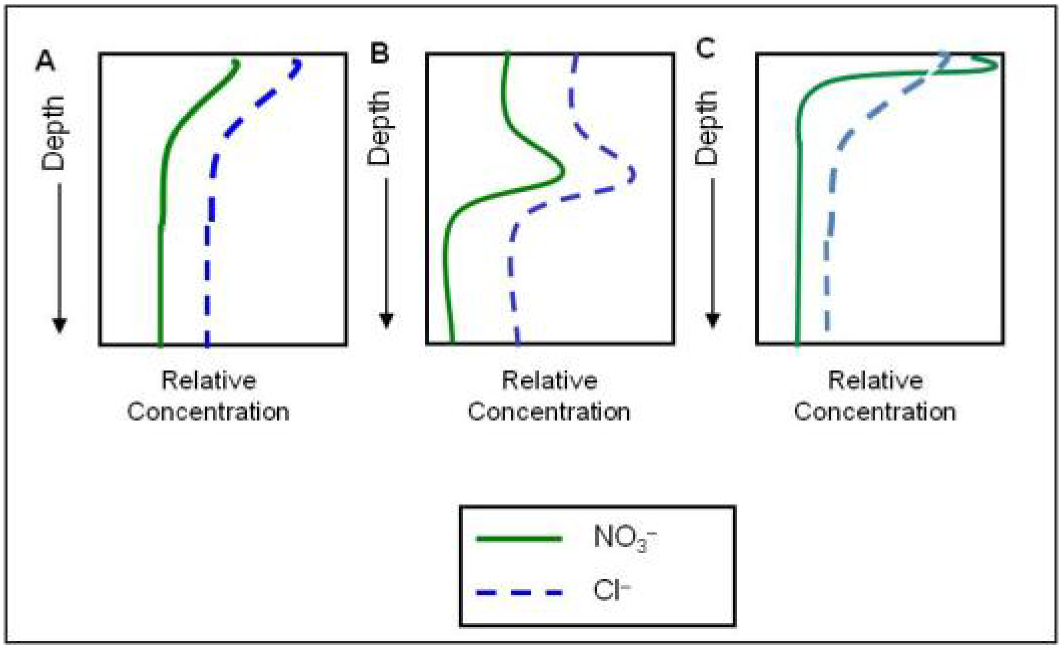

Figure 4.2: Conceptual diagrams of chloride and nitrate concentrations in soil profiles by depth. Note that the depth axis is inverted, showing the top of the soil pit as the top of the graph. If nitrate is not being stored or removed, then we would expect its profile to be similar to the profile of the conservative chloride tracer (A). If infiltration is shallow and nitrate is not removed, then we would expect to see “pooling” of both nitrate and chloride at similar depths (B). If nitrate is being removed as it moves through the profile, then its profile will be dissimilar to the chloride profile (C).

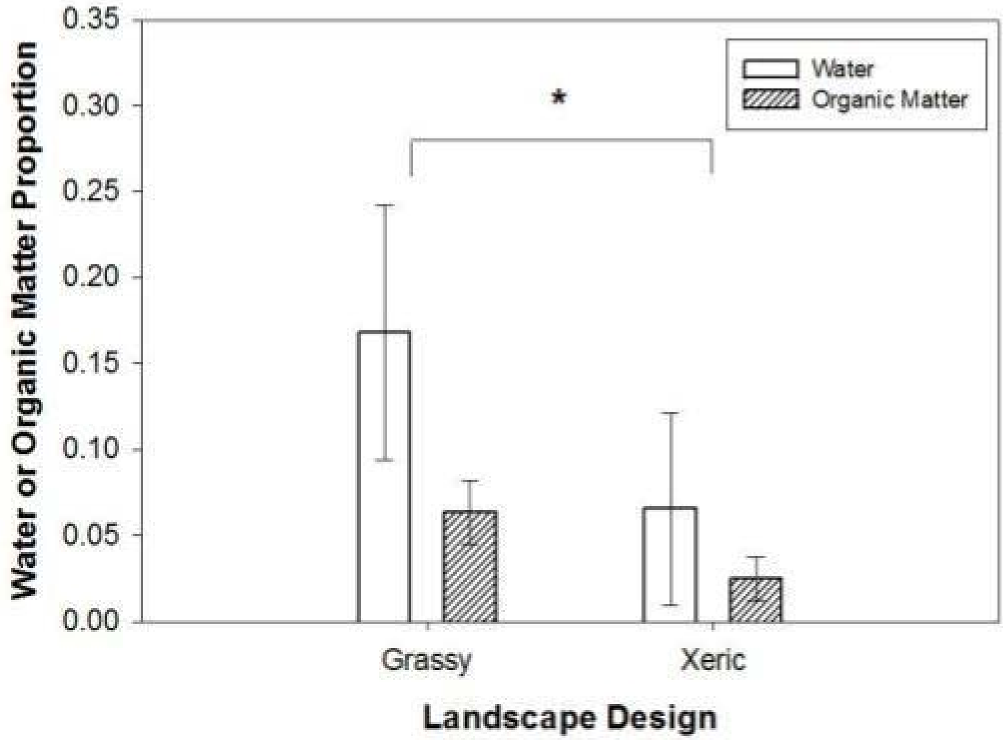

Figure 4.3: Gravimetric water and organic matter content for grass and xeriscaped basin soils. The means for both soil moisture ($F_{(1,30)}$ = 19.62, p = 0.0001) and organic matter ($F_{(1,30)}$ = 47.42, p < 0.0001) are significantly different between the two basin landscape design types.

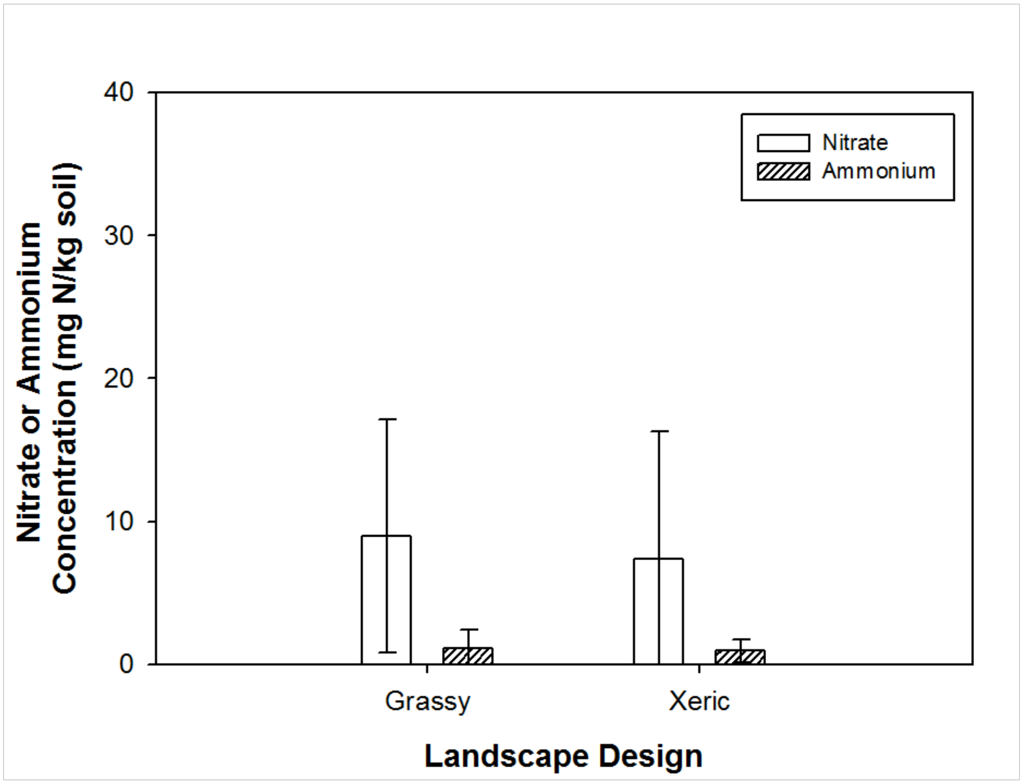

Figure 4.4: Mean NO_3^- and NH_4^+ concentrations for grassy and xeriscaped retention basins. Mean values for NO_3^- ($F_{(1,30)} = 0.45$, $p = 0.5076$) and NH_4^+ ($F_{(1,30)} = 0.034$, $p = 0.853$) were not significantly different for the two basin types; concentration of nitrate was much higher than ammonium in both basin types.

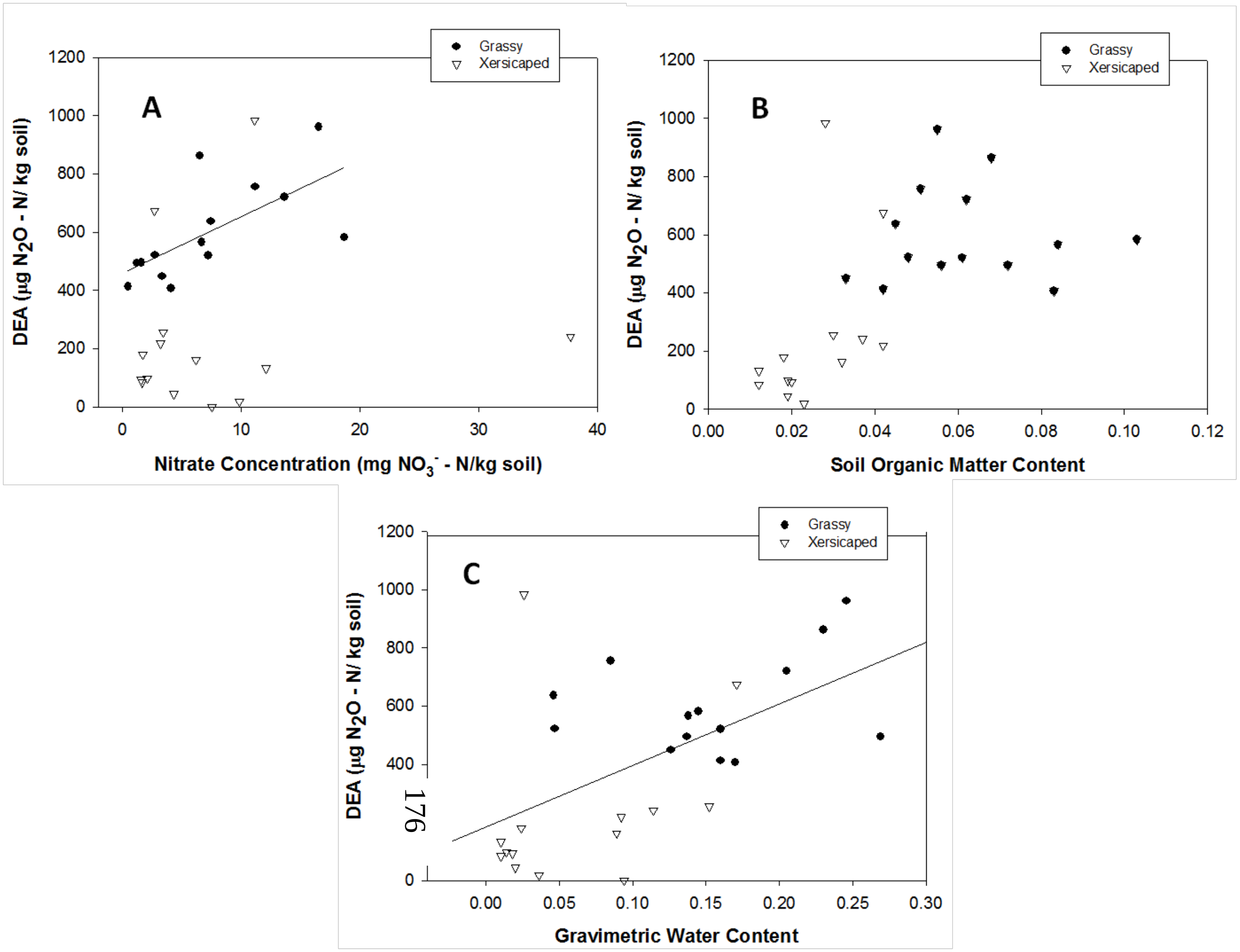

Figure 4.5: Relationships between DEA and soil properties. A) DEA in grassy basins is significantly correlated with soil nitrate concentrations (R^2= 0.637, p = 0.0015). B) DEA was not significantly correlated with soil organic matter, either when considered for the whole dataset or by basin type. C) DEA in retention basin soils is significantly correlated with gravimetric water content (adjusted R^2 = 0.439, p = 0.0004) when xeriscaped and mesic landscape design are considered together.

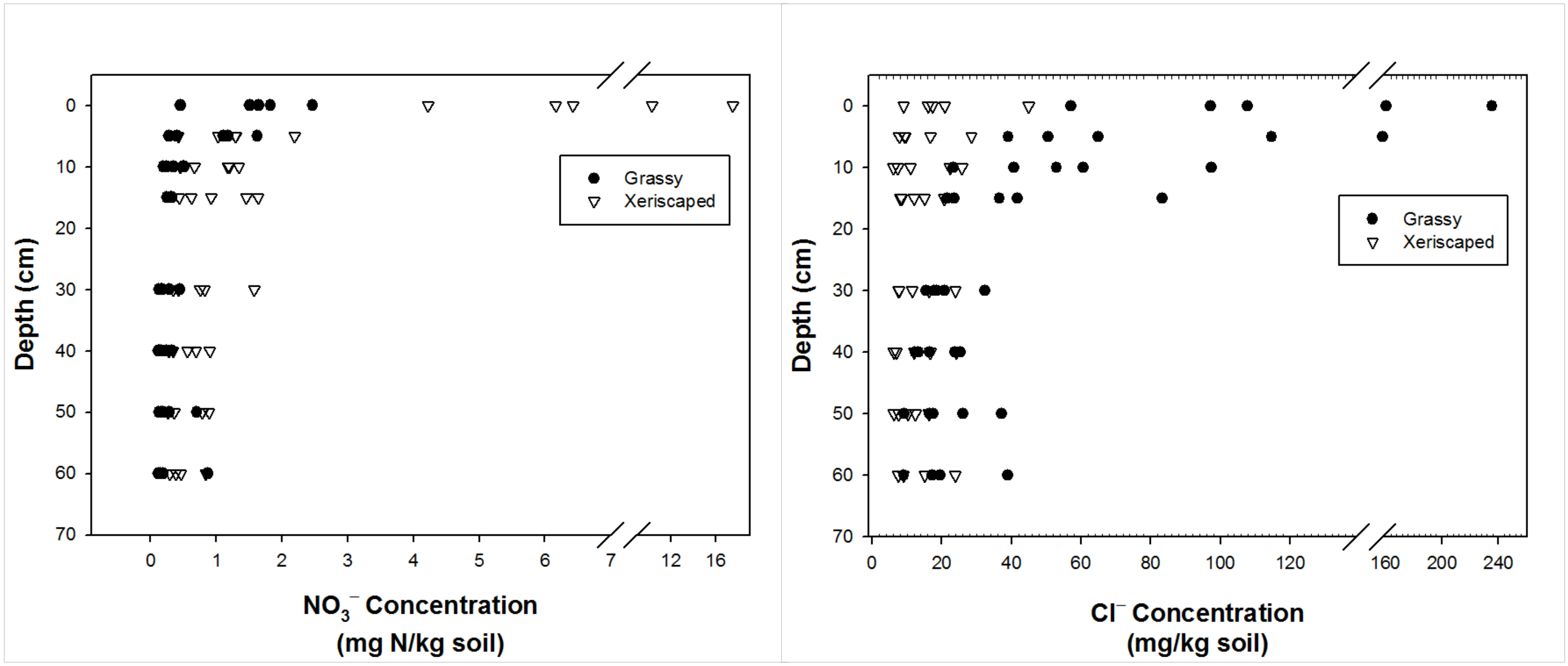

Figure 4.6: Variation in nitrate and chloride concentrations with depth in soils of grassy and xeriscaped basins. Note that the depth axis is inverted, showing the top of the soil pit as the top of the graph. Also note that the scale for chloride is different than the scale for nitrate.

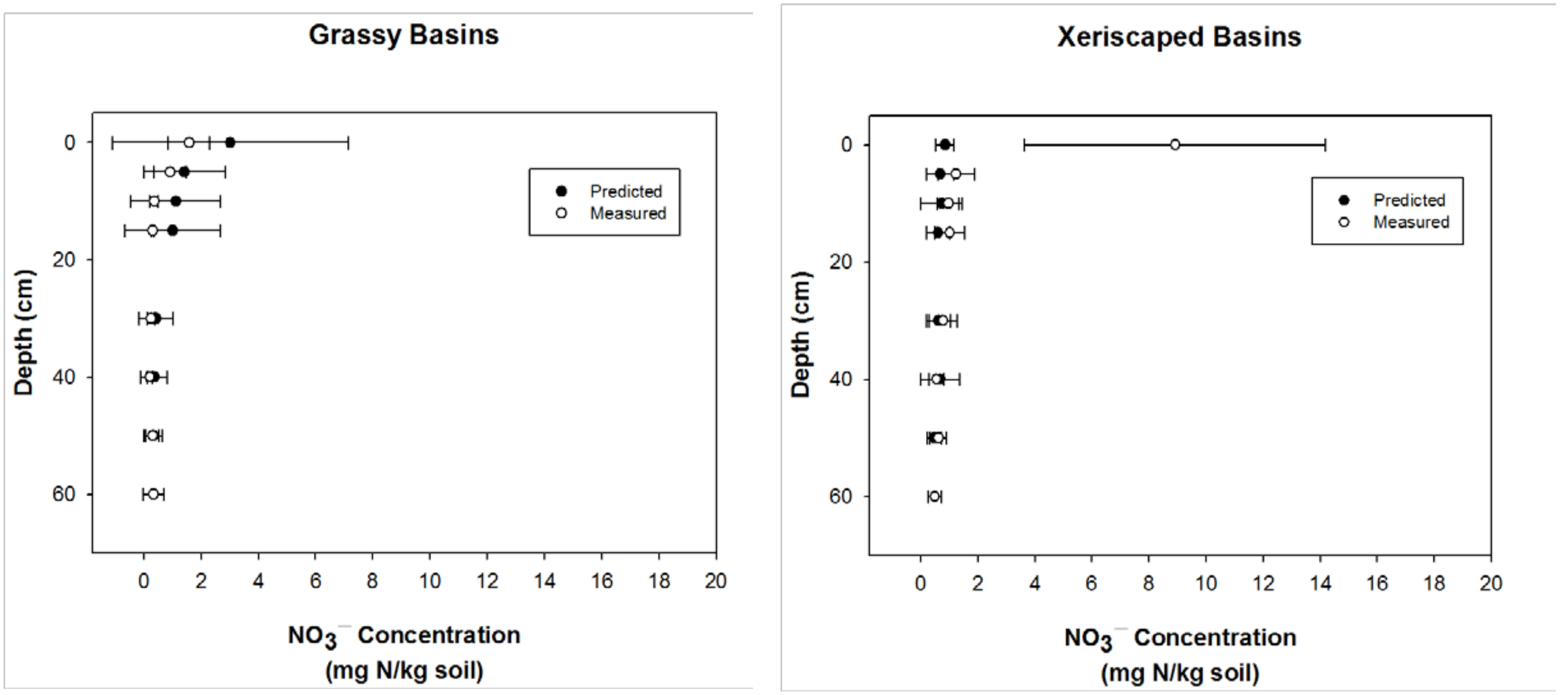

Figure 4.7: Predicted (based on chloride) and measured nitrate concentration for the two landscape designs. Note that the depth axis is inverted, showing the top of the soil pit as the top of the graph.

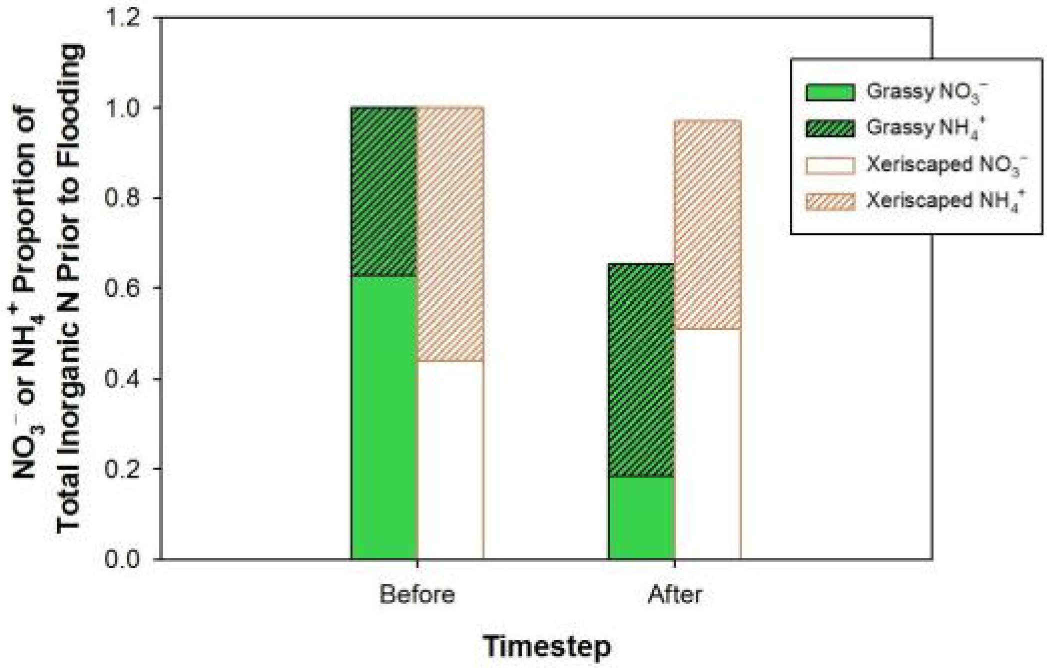

Figure 4.8: Proportion of soil total inorganic N as NO_3^- and NH_4^+ before and after experimental basin flooding, for the two basin landscape designs. Values are scaled to the total available inorganic before flooding.

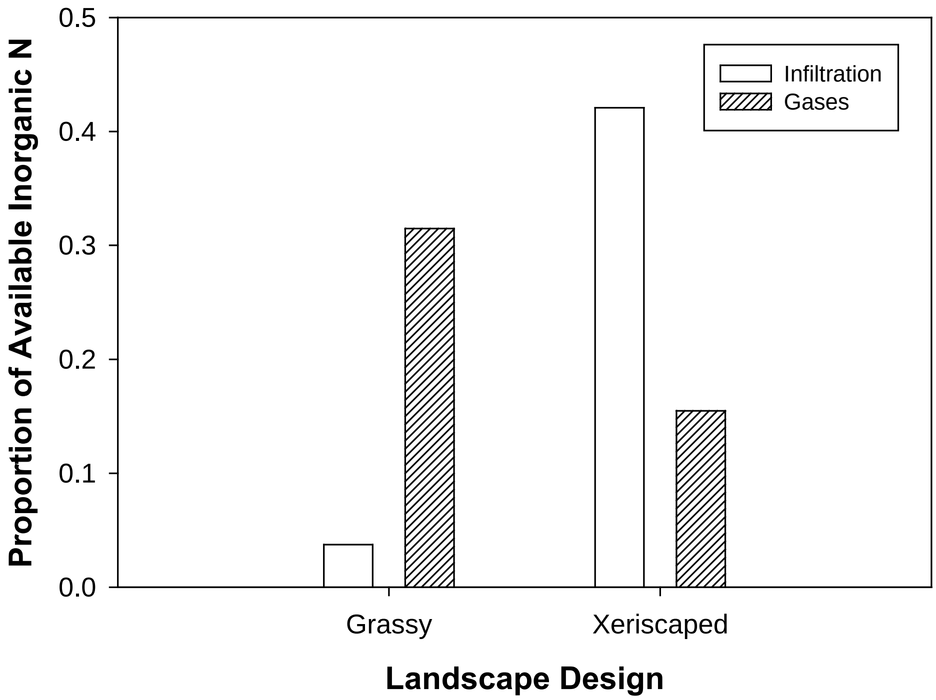

Figure 4.9: N losses during experimental flooding via infiltration (as NO_3^- and NH_4^+) or gas evasion (as N_2 or N_2O). Values are expressed as a proportion of total available inorganic N for each landscape design.

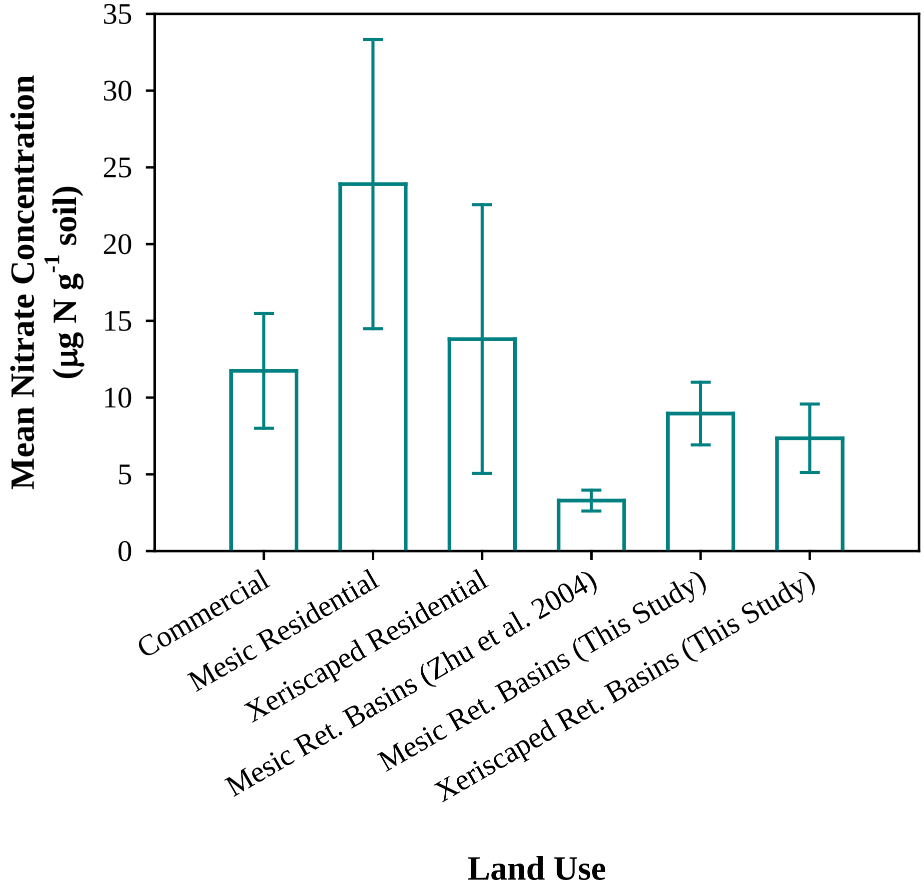

Figure 4.10: Comparison of results from this study with CAP LTER mean soil nitrate concentrations by land use with mean nitrate concentration of mesic retention basin soils studied by Zhu et al. (2004). Whiskers are ± 1 SE.

Chapter 5

LOCALLY PROVIDED HYDROLOGIC ECOSYSTEM SERVICES IN THE PHOENIX, METROPOLITAN AREA: POTENTIAL BUNDLES AND TRADEOFFS

5.1 Abstract

Humans rely on multiple natural capital stocks that generate ecosystem services essential to meeting basic needs and improving quality of life. Management of these natural capital stocks and the services they generate requires insightful planning, especially given the rapid pace of global environmental change due to urbanization and climate change. Many studies have acknowledged the potential for tradeoffs between ecosystem services of different types (e.g., provisioning and regulating services), but few have empirically quantified landscape-level interactions among services. Here we apply a framework developed by Raudsepp-Hearne et al. (2010) to examine these relationships for locally provided ecosystem services derived from hydrologic processes in the Phoenix, AZ metropolitan area. In our study, we mapped nine proxies for eight different ecosystem services, including provisioning (municipal water production, quarries, and water inputs to agricultural production), regulating (flood and climate modulation), and cultural services (recreation, aesthetic value/sense of place, and education). We examined the spatial distribution and interactions of these services at three spatial scales. Our results revealed both significant positive and negative interactions between services, and that these

were in part dependent on the scale of analysis. We found that, for some ecosystem services, the location was constrained by the hydrogeomorphology of the area, while for others human engineering and the creation of designed ecosystems have enabled the delivery of ecosystem services independent of natural hydrologic constraints. Application of this type of framework for the analysis of bundles of and tradeoffs between ecosystem services can help identify key areas of the landscape that require consideration in future planning and management efforts.

5.2 Introduction

The redistribution and growth of the global population and rapid changes in climate in the past century have highlighted and strained our dependence on natural capital (Daly 1994, MA 2005). Natural capital, when combined with the other forms of capital – human-made and cultural – generates essential real income (i.e., services flowing from capital, Fischer 1906) for human enterprise (Berkes and Folke 1991, Folke et al. 1994), or ecosystem services (Daily et al. 1997). The concentration of human population and activity in cities has resulted in complex landscapes that are emergent phenomena resulting from interactions between humans and ecological processes (Alberti 2008). The urban landscape – including its hydrology, geomorphology, and vegetation – has been dramatically altered, changing the needs and demands of residents, the availability of natural capital, and our ability to derive income from that capital. In other words, urbanization alters the capacity of the land to support the underlying ecological functions of ecosystem services, as well as modifying the suite of services demanded by its residents.

In urban ecosystems, most provisioning services (food, fuel, fiber, and sometimes water) are provided by ecosystems outside of the administrative boundaries of the city, and frequently from far beyond the immediate surrounding area (Folke et al. 1997, Luck et al. 2001). However, there remain certain services that must be locally provided, including regulating services like water flow regulation and cooling (reduction of the urban heat island), or cultural services such as recreation, aesthetic value, and sense of place. These services can be met by various combinations of natural, human-made, and cultural capital. For example, cooling can be provided by mechanical air conditioning: a combination of the human-made capital of the physical air conditioner, the cultural capital of the ingenuity of humans to invent the air conditioner and train others to construct them, and the natural capital that is fueling the creation of electricity to run the air conditioner (e.g. coal or oil). Alternatively, cooling could be provided by evapotranspiration of nearby vegetation: a combination of the natural capital of the plants, soil, and water, as well as the human-made and cultural capital that allows for the selection, installation, and maintenance of the vegetation. In the latter case, the natural capital comes from a local ecosystem in time and space, rather than distant or ancient ecosystems. For some services in some locations, it may be that the local ecosystems are the most efficient sources of the necessary natural capital. Therefore, in addition to leaving some component ecosystems relatively intact within the larger urban ecosystem, the process of urbanization also creates novel ecosystems for the express purpose of locally creating one or more ecosystem service (Palmer et al. 2004). Humans harness the “cultivated natural capital” (Daly 1994) of these ecosystems and combined it with human-made and cultural capital to generate real income (necessary and desired) for urban living.

Many ecosystems have the capacity to support the creation of multiple ecosystem services (MA 2005, Kareiva et al. 2007), but no ecosystem has all of the requisite biological functions to create all ecosystem services maximally. Efforts to enhance the production of some ecosystem services, especially provisioning services such as food, water, and fiber, have resulted in the degradation or elimination of the other functions in the same ecosystems, reducing the delivery of regulating and cultural services (Tilman et al. 2002, Grimm et al. 2004, Rodriguez et al. 2006, Bennett and Balvanera 2007). With the increasing awareness of the value of non-provisioning services and their decline in availability, recent research has focused on the relationships and tradeoffs among multiple ecosystem services (Foley et al. 2005, Kremen and Ostfeld 2005, Carpenter et al. 2006, Bennett et al. 2009, Carpenter et al. 2009, Nelson et al. 2009).

Several studies have examined the spatial and temporal congruence of different ecosystem services in particular regions or ecosystem types (Egoh et al. 2008, Naidoo et al. 2008, Tallis et al. 2008, Nelson et al. 2009, Raudsepp-Hearne et al. 2010), although uncertainties remain surrounding issues of suitable proxies (environmental characteristics representing the ecosystem services provided), comparable units across multiple services, and the desirability of congruence. Raudsepp- Hearne et al. (2010) developed a methodological framework for assessing potential tradeoffs and synergies among ecosystem services within a given region. They conducted an analysis of the spatial configuration, interactions, and bundling of several ecosystem services in the peri-urban Montreal, QC, Canada area. They were able to identify 6 distinct bundles of ecosystem services recognizable by socio-ecological characteristics, which they labeled as: corn–soy agriculture, feedlot agriculture, destination tourism, exurban, villages, and country homes. Through this work, they were able to examine how the use of and interactions between ecosystem services are influenced by both social and ecological factors. They assert that their observed patterns of bundles tradeoffs are likely to be similar to those in other peri-urban agricultural regions as well.

Little assessment has been done of potential synergies and tradeoffs of locally provided urban ecosystem services, despite the fact that the planning, design, and maintenance of urban ecosystems offer opportunities for experimentation and implementation (Grimm et al. 2000). The challenge for science and management remains in appraising the extent, coincidence, and interactions of multiple ecosystem services. In this research, we used an approach similar to Raudsepp-Hearne et al. (2010), analyzing the spatial distribution of water-related ecosystem services in the Phoenix, AZ metropolitan area. We focused on water-related ecosystem services because of their importance for sustaining this rapidly growing desert city. We selected several proxies to represent the various structures and functions of aquatic ecosystems contributing to ecosystem services to the region. Using these proxies, we evaluated possible coincidence or isolation of specific services or groups of services throughout the region. We assessed correlations between these proxies at several spatial scales to elucidate potential tradeoffs and bundling of ecosystem services. And finally, we examined the potential drivers of variation in the creation and location of hydrologic ecosystem services in this metropolitan area.

5.3 Methods

Study Area

Phoenix is home to the Central Arizona–Phoenix Long-Term Ecological Research site (CAP LTER), which encompasses a 6400- km^2 area that includes the city, surrounding suburbs, and undeveloped desert (Figure 5.1). Mean annual precipitation in this northern Sonoran desert area is less than 200m, roughly divided into two distinct seasons: a winter season associated with Pacific frontal storms, and a summer monsoon season characterized by intense, localized convective storms. Mean annual temperature is 22° C and summers are hot (mean July temperature= 34.5°C). Despite its extreme climatic conditions, the Phoenix metro is one of the most rapidly growing metropolitan areas in the US, increasing from approximately 300,000 residents in 1950 to greater than 4 million in 2007

(US Census 2008). The city sits at the confluences of the Salt and Verde, and Salt and Gila Rivers, once perennial rivers that are now dammed or diverted upstream of the metro area. Tributaries to the Salt are largely ephemeral and dammed as well, resulting in little instream flow throughout the washes, streams, and rivers of the city. Water resources for the region include the Salt and Verde Rivers (ten year range from 1997–2006 = 21–41%), the Colorado River (14–27%), groundwater (28–40%), and treated effluent (5–7%) (ADWR 2009). The large degree of variation in sources is caused by the inherent high interannual variability of rainfall in the Sonoran Desert. There has been extensive small-scale modification of the hydrogeomorphology throughout the region, with the creation of hundreds of artificial lakes, thousands of stormwater retention basins, as well as a small number of restored/designed riparian areas (See Chapter 1, Chapter 2, Roach et al. 2008).

Ecosystem Service Proxies

We selected nine proxies that potentially serve as indicators of ecosystem services from three categories of services, as defined in the Millennium Ecosystem Assessment (MA 2005, Table 5.1). The proxies were chosen both due to the availability of spatially explicit data, and because they allowed comparison of the spatial distribution of services of a variety of types.

Provisioning Services

As mentioned above, municipalities in the Phoenix area rely on groundwater for a substantial portion of their freshwater needs. The service of water provisioning is dependent on the quantity and quality of available groundwater, which are in turn dependent on the functioning of ecosystems at the land surface where recharge occurs (such as mountain fronts, recharge basins, stormwater retention basins, etc.) as well as the groundwater ecosystem itself. The Arizona Department of Water Resources (ADWR) maintains a database of all wells for the metropolitan area, as required by the Arizona State Groundwater Management Act of 1980 (ARS 1980), and we use the number of wells as a proxy for groundwater provisioning. We mapped the location of more than 8300 wells within the CAP LTER region that were identified in a 2004 ADWR database as provisioning wells for municipal consumption. Although we would prefer to include information about rates of consumption and differentiate types of water use (e.g., human consumption vs. outdoor water use), available data precluded these refinements. While the ADWR database does include some data on well pump rate, more than half of the records were missing for this variable, and data on water end use were unavailable on a per-well basis.

Sections of the Salt River and Agua Fria rivers are currently in use as quarries for river rocks, gravel, and other concrete precursors. As extraction of a natural resource, this gravel quarrying relies on the combination of geologic, hydrologic, and riverine ecosystem processes to create the natural capital used in economic consumption. The extraction this natural capital destroys the local aquatic ecosystem and has significant impacts on upstream and downstream ecosystems as well (Kondolf 1994, 1997). In this way, it can be thought of as a consumptive, non-sustainable use of natural capital. For this study, we used the area of each quarry parcel, as determined from the 2005 parcel land-use map provided to CAP LTER from the Maricopa County Assessor's (MCA) office. While this measurement may not represent the current rate of provisioning of this service, it does spatially bound the amount of total extraction possible from the floodplain ecosystem.

Irrigated agricultural lands currently occupy large areas of the Phoenix metro area. The majority of crops grown in the area are cotton, wheat, and alfalfa (MC-AQD 2010). Due to the aridity of the area, all crops rely on significant inputs of water for irrigation. Rather than use the amount of each crop produced per area, we calculate the area of irrigated agricultural lands to represent the contribution of water to this service, since it relies upon water as an essential input supporting the end product. These data are for the year 2006, and were provided to CAP LTER by the ADWR.

Regulating Services

Although climate of the region is arid, individual storms or a series of events during the winter rainy season can generate substantial rain and snow in the upland watersheds of the Salt and Verde Rivers, resulting in downstream flooding. To prevent loss of life and property, low-lying areas in and along the riverways are protected from development. In some places these floodways are maintained as a relatively natural desert river ecosystem, with meandering channels and abundant riparian vegetation, but in other areas the floodway is channelized and vegetation reduced or eliminated. The connected washes and riverine ecosystems as whole, however, provide the ecosystem service of flood abatement (water flow regulation). As one measure for estimating this ecosystem service, we use area of protected floodway as designated by the Maricopa County Flood Control District (MCFCD).

During the summer monsoon season, more localized storms can produce large volumes of runoff in a relatively small area. One common approach to flood abatement in this situation is to construct stormwater retention basins in small residential catchments. In Chapter 2 we used spatial data on the location of drywells to estimate the potential number and location of these basins in the Phoenix region, and we use those data as a proxy for flood abatement via stormwater retention. Although these are only point locations, they indicate the demand for flood abatement in the local area and are correlated with the existence of stormwater retention basin ecosystems that can provide this service. The original drywell data were provided by the Arizona Department of Environmental Quality (ADEQ); see Chapter 2 for a complete description of data source and GIS-layer construction.

In this hot, arid climate, vegetation contributes significantly to reducing the urban heat island within the city (Harlan et al. 2008, Buyantuyev and Wu 2010). In older sections of the metropolitan area, residential yards are flood-irrigated throughout the spring and summer to support lawns and mature trees. The transpiration of the vegetation and evaporation of standing water are two ways that water contributes to the ecosystem service of microclimate modulation (cooling, reducing the impact of the urban heat island). We used a GIS layer showing the location of residential flood irrigation from the year 2004, provided by the Salt River Project (SRP), a private corporation that delivers water from the Salt and Verde Rivers to residential irrigation customers in the Phoenix urban area.

Cultural Services

Water also is essential for the creation of several cultural ecosystem services in the urban area. With respect to recreation, for example, the Phoenix region has 92.8 km^2 of golf courses, a water-intensive land use that also relies on substantial inputs of human and human-made capital. We again used the parcel land-use data from the MCA to map the total area of golf courses. Human-made lakes are another example of water-dependent recreation in the urban area. Previous work (see Chapter 2) has estimated that there are approximately 1,000 artificial lakes in the Phoenix metropolitan area. We used that GIS layer to calculate the location and area of these lakes for our analysis.

Although located in the Sonoran Desert, engineering projects at a variety of scales – from the Bureau of Reclamation dams along the Colorado River to the canals managed by local water districts and the significant groundwater mining – have made water supplies relatively abundant, enabling dramatic transformations of landscape geomorphology and vegetation in the city. As mentioned above, there are many artificial lakes, golf courses, and areas that are flood-irrigated, creating novel ecosystems for the desert climate (Gober 2006). Part of the motivation for the creation of these ecosystems must be the desire for specific aesthetic value and to create a sense of place, because there are possible substitutions for the other types of services provided by these ecosystems. Therefore, urban residential irrigation, golf courses, and artificial lakes also serve as proxies for the cultural ecosystem services of aesthetics and sense of place.

One other water-related feature that provides aesthetic value and a sense of place is the restored or designed aquatic ecosystem. Several such projects exist in the Phoenix region, such as the Riparian Preserve in Gilbert (Riparian Institute 2010) and the Rio Salado Project (City of Phoenix 2010). Education is another express purpose for the creation and maintenance of these ecosystems. For this study, we manually created a GIS layer showing each of these restoration/design projects to use as a proxy for the ecosystem services of aesthetics, sense of place, and education provided by these ecosystems.

Scales of inquiry

Each ecosystem service proxy was quantified at 3 different scales for our study: cities, census tracts, and a hybrid of these two socioeconomic and political units, which we term “city hoods.” In Raudsepp-Hearne et al. (2010), metrics for ecosystem services were calculated for each municipality within the study area, and then standardized by unit area. They selected that scale of analysis because it seemed most relevant to decision-making processes that could potentially influence the provisioning of the chosen ecosystem service. The average size of the 137 municipalities included in their study as 74 km^2. For the 23 Phoenix metropolitan area cities included in this study, areas range from 2 km^2 (Guadalupe) to 1344.2 km^2 (Phoenix), with a mean of 190.8 km^2. While analysis at the city scale does make sense for our study with respect to possible planning and management outcomes, the spatial extent of many of the cities makes average calculations less meaningful, as such large cities undoubtedly cover a variety of terrain and land use. Because one of our interests was in examining correlations between the provisioning of ecosystem services and socio-economic factors, we decided to also conduct the analysis at the census-tract level. This yielded a sample size of 663 tracts, which ranged in area from 0.12 km^2 to 950.8 km^2, with a mean of 9.6 km^2. This finer resolution may better represent the distribution and potential spatial clustering of our ecosystem services, but census tracts are not easily discernable to residents and policy makers. We therefore created a medium-scale unit, "city hoods." To construct these, we first categorized the census tracts by quantile with respect to medium household income. We then aggregated tracts that fell within the same quartile and city. This resulted in 58 city hoods, ranging in area from 0.12 km^2 to 555 km^2, with a mean of 69.8 km^2. At each spatial scale, each proxy was quantified and normalized per unit area. For point data (provisioning wells and drywells), we calculated the number per unit area, for all other proxies we used the total proxy area per unit area. To allow for comparisons between proxies, all data were transformed so that the maximum provisioning for each ecosystem service per unit area was 1.

Analysis of Spatial Distributions, Interactions, and Bundling

For each level of spatial resolution, the proxies for ecosystem services were mapped using ArcGIS in order to visualize their distributions and patterns. Spatial auto-correlation of each proxy was assessed using Moran's I (Moran 1950). Data were then exported from ArcGIS and analyzed for pair-wise correlations using PASW Statistics 18 (SPSS 2010). Because the data were not normally distributed and had high levels of heteroskedasticity, we used the Spearman's non-parametric correlation test for all comparisons. In addition to examining correlations between specific proxies, we also examined the correlation between each proxy and basic socioeconomic and environmental factors known to vary throughout the metropolitan area: median resident age and income, vegetation abundance, elevation, and percent land cover in agriculture, desert, and urban categories at each spatial scale. Cluster analysis was used to identify groups of tracts/city hoods/cities that have similar sets of ecosystem services (i.e., bundles). We used the K-means cluster-analysis tool in PASW Statistics 18 (SPSS 2010), relying on dendrograms and scree plots to determine an appropriate number of clusters. The results of the clustering procedure were then also mapped in ArcGIS to visualize and assess the spatial pattern of ecosystem-service bundles. Finally, we used the star-plot function in R statistical software (R Development Core Team 2008) to visualize the provisioning of ecosystem services by proxy (hereafter, referred to as "flower" plots). For all statistical analyses $\alpha = 0.05$, unless otherwise noted.

5.4 Results

Spatial patterns of individual ecosystem services

Some ecosystem services were spatially auto-correlated (clustered), although this depended on the spatial scale of analysis (Table 5.2). At the tract level (finest resolution), all ecosystem services were spatially clustered except for education – that is, areas with high levels of a particular service tended to be near each other and vice versa. At the next level of resolution, city hoods, only flood protection (water flow regulation) via drywells/retention basins, climate regulation via urban irrigation and lakes, and recreation via lakes were spatially clustered on the landscape. And at the coarsest resolution, cities, only the level of flood protection via drywells/retention basins was spatially clustered. Note that some services, such as water provisioning, were located more frequently in peripheral areas, whereas flood mitigation was provided more often in more centrally located areas (Figure 5.2).

The loss of spatial clustering as resolution decreased (i.e., from tracts to city hoods to cities) could be the result of city hoods and cities being large enough to encompass enough census tracts to provide similar levels of ecosystem services at the aggregate level – that is, all cities were generally similar, so the autocorrelation was spread throughout the landscape. Or, it may be that city hoods and cities differed from each other in level of provisioning, but areas with high and low provisioning of a given ecosystem service were randomly distributed. It may also be an artifact of the fact that some areas of land included at the tract level were not included at the city hood and city level, as they are not incorporated (i.e., they are in the county but not any city). This creates “holes” in the city hood and city landscapes, so that areas that are similar may be noncontiguous and therefore did not form a cluster.

Interactions among ecosystem services

Several ecosystem service proxies were strongly correlated, either positively (such as urban irrigation and drywells), or negatively (such as drywells/retention basins and floodways), but for others, significant correlations were weak (such as quarries and drywells/retention basins at the tract level). Fewer significant correlations were found at the city level of analysis. Across categories of ecosystem services, provisioning services were almost always correlated with each other across scales. For flood protection, drywells/retention basins and floodways were generally negatively correlated. The climate-regulation proxies of urban irrigation and lakes were negatively correlated at the tract level, but positively correlated at higher levels. For the water-intensive ecosystems that provide aesthetic value and a sense of place, the correlations were sometimes positive (as in the case of golf and artificial lakes), were negative at the tract level for urban irrigation with respect to lakes and golf, or were otherwise neutral.

Potential ecosystem service bundles

For each spatial scale, dendrogram and scree plots indicated that the most appropriate number of clusters within the data was 3. Each cluster represents a bundle of ecosystem services characteristic of the area. The locations of tract/city hood/city bundles were spatially auto-correlated, as seen in Table 5.2 and Figure 5.3. That is, similar suites of ecosystem services tend to occur near each other across scales, despite the fact that the ecosystems these services were derived from were frequently much smaller than the unit of analysis at each scale.

The flower diagrams of standardized levels of services further demonstrated that there were characteristic bundles of ecosystem services depending on location in the metropolitan region (Figure 5.4). Each city was characterized by a distinctive mix of ecosystem services, with significant variation in the provisioning of ecosystem services at lower scales (within the cities). Even city hoods within the same city tended to be dissimilar. In the examples shown, it is worth noting that although higher mean income was associated with higher levels of some services (Table 5.4), for these cities the city hoods with moderate income levels (levels 3 and 4) generally had the lowest levels.

Potential drivers of ecosystem service creation and utilization

Some ecosystem service proxies exhibited clear relationships with socioeconomic and environmental variables (Table 5.4). All ecosystem services were significantly related to population density at the tract level, and all of these correlations were negative, except for urban irrigation and flood protection via drywells/retention basins. Increasing population age and income were positively correlated with recreational ecosystem services from human-made lakes and golf courses. Additionally, in cases where two types of ecosystems could potentially be used for the same ecosystem service, one ecosystem-based solution was more frequently associated with a given land cover. For example, protected floodways and drywells in retention basins both can be used for flood abatement, but the former were more frequently found in areas with a high percentage of desert land cover, and the latter were in areas with higher urban land cover.

5.5 Discussion

This study examined potential synergies and tradeoffs between hydrologic ecosystem services in an arid urban region and assessed potential characteristic bundles of ecosystem services for different areas of the landscape. While in some instances the location of specific types of services is constrained by the hydrogeomorphic template (e.g., quarries or floodways), the construction and maintenance of designed, water-dependent ecosystems outside of areas where natural aquatic habitats occur (e.g. human-made lakes, urban and agricultural irrigation) has enabled people to derive water-related services from cultivated natural capital. The engineering of hydrologic systems (human-made capital in the form of infrastructure) allows for the local utilization of a variety of ecosystem services across a diverse landscape.

We found that the scale of analysis had an impact on the assessment and interpretation of results. The tract level is probably too fine a scale to assess spatial autocorrelation, at least for the smaller tracts in the urban and suburban areas. On the other hand, given the large geographic size of several of the cities in the region, the city level seems too broad for meaningful assessment of the distribution and interactions of many of the ecosystem services, which tend to have their main benefits actualized at a much more local scale. The intermediate scale of city hoods used in this study may be more appropriate for assessing patterns and potential bundling of some services, but the lack of correspondence with established socioeconomic or environmental management boundaries places some limits on the insight gained from this analysis. It is likely that the appropriate scale may vary by ecosystem service or bundle of services, for several reasons. First, some ecosystems have a characteristic size that falls within a relatively narrow range (e.g., stormwater retention basins). While this scale would be appropriate for analysis of services flowing from that specific ecosystem, care must be taken when making comparisons to services from ecosystems of a different characteristic scale (e.g., floodways). These differences in spatial scale among ecosystems means that they may not respond to drivers similarly (if at all), and thus interactions and bundling of services may change under new conditions (Bennett et al. 2009). Additionally, in these complex socio-ecological systems, the selection of focal scale is further complicated by the fact that the existence, distribution, and level of ecosystem functioning of these systems are emergent from the interactions and feedbacks between human decisions (which themselves are hierarchically structured), biophysical agents, and natural processes (Alberti 2008).

Clustering analysis identified bundles of ecosystem services that also tended to be spatially clustered on the landscape. At the city level, bundle type was associated with vegetation abundance and percent desert (data not shown). However, further analysis did not allow for ready, intuitive characterization of these bundles like those in the exurban Montreal analysis. Raudsepp-Hearne et al. (2010) were able to label their bundles by drawing on the socioeconomic and environmental variables associated with each bundle type. In this analysis, across scales there were only weak associations between bundles and those types of variables. This lack of easily named ecosystem-service bundles could be the result of omitted variables, or improper scale of analysis, or of generally weak clustering of ecosystem services into bundles.

The locations and types of ecosystem processes underlying provisioning (water purification and storage, quarries, crop production) and water flow regulation services seem to coincide with general land-cover characteristics. Quarries and protection via floodways tend to be associated with higher levels of desert land cover. The higher the proportion of agricultural land, the higher the level of provisioning services (Table 5.4). In contrast, highly urbanized areas tend toward using drywells/retention basins rather than floodways for water flow regulation. The negative correlation coefficients between ecosystem types that contribute to flow regulation (Table 5.3), indicates preferential use of one ecosystem type over another depending on location. A fundamental question remains, however, as to whether these associations occur due to the local demand or the ready supply. With respect to quarries and floodways, it seems clear that existing riverine ecosystems have constrained the reliance on these ecosystem services to specific locations on the landscape. The preferential reliance of highly urbanized areas on drywell/retention basins for flood protection could be the result of both increased need due to greater runoff, as well as increased efficiency with respect to land area at providing this ecosystem service.

The other types of ecosystem services—recreation, education, aesthetics/sense of space—tend not to be associated with a particular type of land cover and generally are not spatially clustered on the landscape. The location of these "quality of life" services is associated with designed ecosystems, and seems to have arisen due to local demand, rather than populations taking advantage of antecedent ecosystems. The overall association with higher levels of income for human-made lakes and golf is indicative of the costs required to create and maintain these water-intensive ecosystems. However, this relationship is not always the case; for example, in Tempe and Chandler the lowest income quintiles have the highest provisioning of lakes and golf courses, respectively (Figure 5.4). The lack of correspondence between these services and income could be the result of intentional planning of amenities in lower-income areas. Or, it may reflect that residential location choice the result of tradeoffs among a number of factors, including ecosystem services not included in this analysis as well as non-ecological, human-made and cultural capital and services, such as proximity to employment centers.

5.6 Conclusion

In this study we applied the framework developed by Raudsepp-Hearne et al. (2010) to evaluate potential bundling and tradeoffs of hydrologic ecosystem services in the Phoenix, AZ metropolitan area. While Raudsepp-Hearne et al. (2010) focused their analysis on a largely agricultural region (exurban Montreal, QC, Canada), we wanted to investigate the potential of utilizing their methods in a heterogeneous, predominately urban area, as well as within a very different climatic context. Like Raudsepp-Hearne et al., we did find significant interactions between ecosystem services, both positive and negative, and were able to associate some of the ecosystem services with landscape socioeconomic and environmental characteristics. While the cluster analysis yielded statistically distinct ecosystem-service bundles, they were not easily interpreted and named, unlike the exurban Montreal study. In the future, perhaps better representation of the flow of these services, or inclusion of variables we might have omitted, will improve our understanding the nature and value of contribution of hydrologic processes to ecosystem services in the region. Additionally, a fuller assessment of all urban ecosystem contributions to services will help delineate the relative importance of those dependent on hydrologic processes, their interactions with other ecosystem services, and potential substitutions and tradeoffs among services. Hydrologic systems do not exist in isolation from other components of the socio-ecological system, so while focusing on water-related services may provide valuable insight, decisions should not be made without considering the context and interactions with other natural capital and services.

Our focus on hydrologic processes potentially underlying ecosystem services was based on the assumption that these processes are a vital part of maintaining and improving quality of life in this arid region subject to drought and severe flooding. However, the process of urbanization itself creates water resource and management challenges, regardless of climate, so it will be interesting to see if cities in more humid/temperate areas have similar spatial patterns of hydrologic ecosystem services and their interactions. As cities face the challenges posed by climate and demographic change, it may be a valuable adaptive strategy to alter the configurations of natural, cultural, and human-made capital providing local services to increase efficiency and sustainability. As Folke et al. (1994) note, "A frequently observed pattern, particularly in the modern world, is that ever increasing quantities of human-made capital substituting for natural resources means that ever increasing natural resources are being used elsewhere in the economy to produce that human capital," and often that remote natural capital is non-renewable. Further parsing of the differences between the utilization of ecosystem services of various types across landscapes will elucidate their drivers and interactions, creating knowledge that is essential for sustainable socioeconomic ecosystem planning and management.

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Table 5.1: Ecosystem services analyzed in the Phoenix case study. Note that some proxies may represent more than one service.

Ecosystem Service	Proxy	Data source
Provisioning		
Fresh water	# groundwater wells	ADWR*
Gravel	Area of quarries	MCA†
Food/Fiber	Area of irrigated agricultural land	ADWR
Regulating		
Flood Control	# of drywells	ADEQ°, Chapter 2
	Area of protected floodway	MCFCD§
Climate	Area of irrigated urban land	SRP^
Cultural		
Recreation	# of artificial lakes	Chapter 2
	Area of golf landuse	MCA
Aesthetics/	# of artificial lakes	Chapter 2
Sense of place	Area of urban irrigated land	SRP
	Area of restored/designed wetlands	This study
Education	Area of restored/designed wetlands and riparian areas	This study

* Arizona Dept. of Water Resources
† Maricopa County Assessor's Office
° Arizona Department of Environmental Quality
§ Maricopa County Flood Control District
^ Salt River Project

Table 5.2: Spatial clustering of individual ecosystem services and bundles by level of spatial resolution. Blue highlighting indicates clustering, pink highlighting indicates dispersal, and all other results are randomly distributed in the landscape.

		Provisioning			*Flood Protection*		*Climate Regulation*		*Recreation*	*Education*	
							Aesthetics/Sense of place				
Level		Prod. Wells	Quarries	Ag. Irr.	Dry-wells	Flood-way	Urb. Irr.	Lakes	Golf	Riparian	Bundles
Cities	Moran's I	0.03	-0.10*	0.03	0.21	0.04	0.00	-0.05	-0.08	-0.06	0.11
	z	1.31	-1.80	1.22	4.08	1.34	0.70	0.00	-0.76	-0.21	2.38
City Hoods	Moran's I	0.01	0.01	0.00	0.21	0.00	0.12	0.13	-0.02	0.00	0.07
	z	0.75	0.87	0.56	6.53	0.65	3.95	4.47	-0.22	0.66	2.44
Tracts	Moran's I	0.02	0.01	0.01	0.08	0.01	0.07	0.02	0.04	0.00	0.07
	z	7.30	7.09	5.46	31.15	5.92	31.37	9.89	16.59	-0.55	28.4

* significant at $\alpha = 0.1$

Table 5.3: Spearman correlations between ecosystem service proxies. Significant positive correlations are shaded in blue, significant negative correlations are shaded in red, and non-significant correlations are unshaded.

			Provisioning		*Flood Protection*		*Climate Regulation*	*Aesthetics/Sense of place*	*Recreation*	*Education*
	Proxy	Scale	Quarries	Ag. Irr	Drywells	Floodway	Urb. Irr	Lakes	Golf	Riparian
Provisioning	Prod. Wells	*Tract*	0.19**	0.25**	0.11**	0.11**	0.03	0.11**	0.06	0.05
		City Hood	0.18	0.34**	0.19	-0.03	0.06	-0.03	0.07	0.15
		City	0.18	0.39	0.09	0.19	0.20	0.13	0.09	-0.01
	Quarries	*Tract*		0.17**	-0.08*	0.49**	-0.07	0.02	0.06	0.22**
		City Hood		0.35**	-0.09	0.42**	0.03	-0.01	0.07	0.40**
		City		0.61**	0.29	0.07	0.36	-0.09	-0.12	0.22
	Ag. Irr.	*Tract*			-0.01	0.10*	0.06	0.03	0.04	0.04
		City Hood			0.14	-0.03	0.17	0.12	0.01	0.10
		City			0.38	-0.28	0.30	-0.12	-0.15	0.07
Flood Protection	Drywells	*Tract*				-0.23**	0.29**	-0.05	-0.29**	0.08*
		City Hood				-0.26	0.72**	0.31*	-0.15	0.19
		City				-0.50*	0.68**	0.20	-0.37	0.42*
	Floodway	*Tract*					-0.22**	0.15**	0.18**	0.21**
		City Hood					-0.13	0.02	0.12	0.28*
		City					-0.14	0.26	0.15	0.03
Climate Reg.	Urban Irr.	*Tract*						-0.13**	-0.31**	0.02
		City Hood						0.36**	-0.13	0.37**
		City						0.59**	-0.20	0.65**
Aesthetics/SOP	Lakes	*Tract*							0.37**	-0.01
		City Hood							0.36**	0.12
		City							0.16	0.41
Rec.	Golf	*Tract*								-0.06
		City Hood								0.08
		City								-0.26

** Correlation is significant at the 0.01 level (2-tailed). * Correlation is significant at the 0.05 level (2-tailed).

Table 5.4: Spearman correlations of ecosystem services proxies with socioeconomic, environmental, and land cover factors. Significant positive correlations are shaded in blue, significant negative correlations are shaded in red, and non-significant correlations are unshaded.

		Provisioning			*Flood Protection*		*Climate Regulation*		*Recreation*	*Education*
							Aesthetics/Sense of place			
Factor	Scale	Prod. Wells	Quarries	Ag. Irr.	Drywells	Floodway	Urb. Irr	Lakes	Golf	Riparian
Median Age	*Tract*	0.03	-0.07	-0.14**	-0.35**	0.14**	-0.30**	0.27**	0.41**	-0.06
	City Hood	-0.20	-0.13	-0.51**	-0.52**	-0.01	-0.53**	-0.15	0.34**	-0.09
Mean Income	*Tract*	0.09*	-0.09*	-0.06	-0.20**	0.05	-0.13**	0.15**	0.20**	-0.03
	City Hood	0.08	-0.15	-0.16	-0.12	-0.11	0.11	0.09	0.24	0.00
Vegetation Abundance	*Tract*	0.12**	0.04	0.28**	-0.39**	0.11**	0.05	0.11**	0.30**	-0.03
	City Hood	0.04	0.01	0.03	-0.70**	0.06	-0.66**	-0.36**	0.17	-0.19
Pop. Density	*Tract*	-0.29**	-0.29**	-0.29**	0.22**	-0.33**	0.31**	-0.28**	-0.38**	-0.13**
	City Hood	-0.07	-0.25	-0.24	0.44**	-0.25	0.32*	0.00	-0.24	0.05
Elevation	*Tract*	-0.10*	-0.16**	-0.18**	-0.33**	0.06	-0.45**	0.17**	0.30**	-0.08*
	City Hood	-0.07	-0.05	-0.22	-0.42**	-0.04	-0.29*	0.13	0.25	-0.04
% Agriculture	*Tract*	0.26**	0.20**	0.33**	0.02	0.11**	0.07	0.04	0.01	-0.02
	City Hood	0.03	0.12	-0.04	-0.11	0.12	0.00	-0.21	0.10	0.07
% Desert	*Tract*	0.02	0.21**	0.07	-0.25**	0.45**	-0.29**	0.01	0.19**	0.03
	City Hood	-0.02	0.26*	0.01	-0.32*	0.29*	-0.29*	-0.22	-0.12	0.05
% Urban	*Tract*	-0.15**	-0.16**	-0.22**	0.16**	-0.30**	0.19**	-0.15**	-0.30**	-0.03
	City Hood	0.01	-0.20	0.01	0.49**	-0.21	0.44**	0.21	-0.09	0.18

** Correlation is significant at the 0.01 level (2-tailed). * Correlation is significant at the 0.05 level (2-tailed).

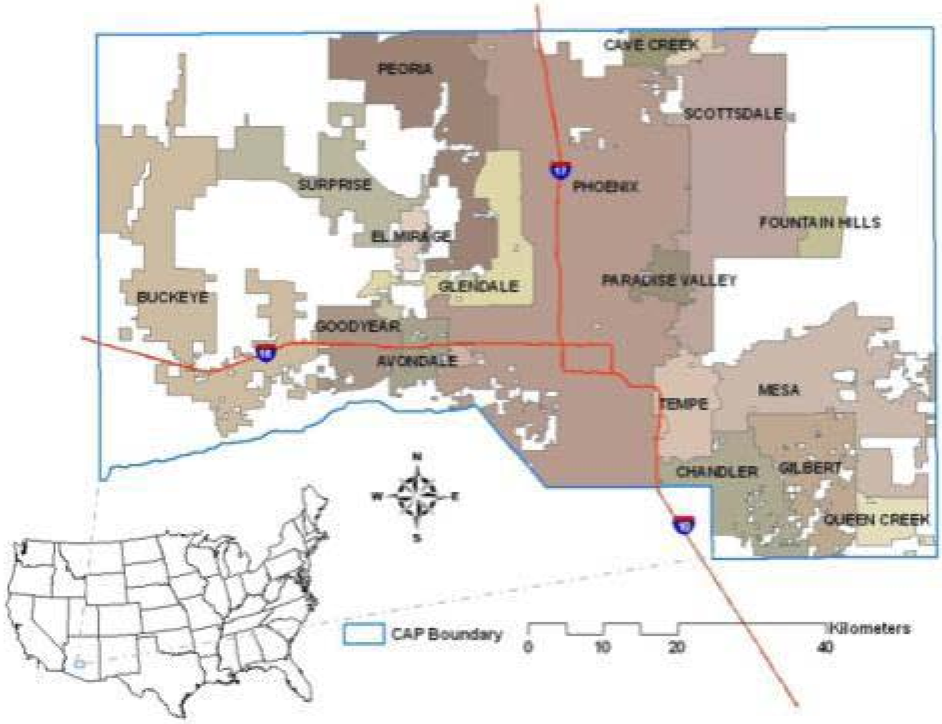

Figure 5.1: Map of the Phoenix metropolitan region. The Central Arizona – Phoenix Long Term Ecological Research project (CAP LTER) area, outlined in blue, incorporates urbanized, agricultural, and desert areas. The city level is shown here as the shaded portions of the map.

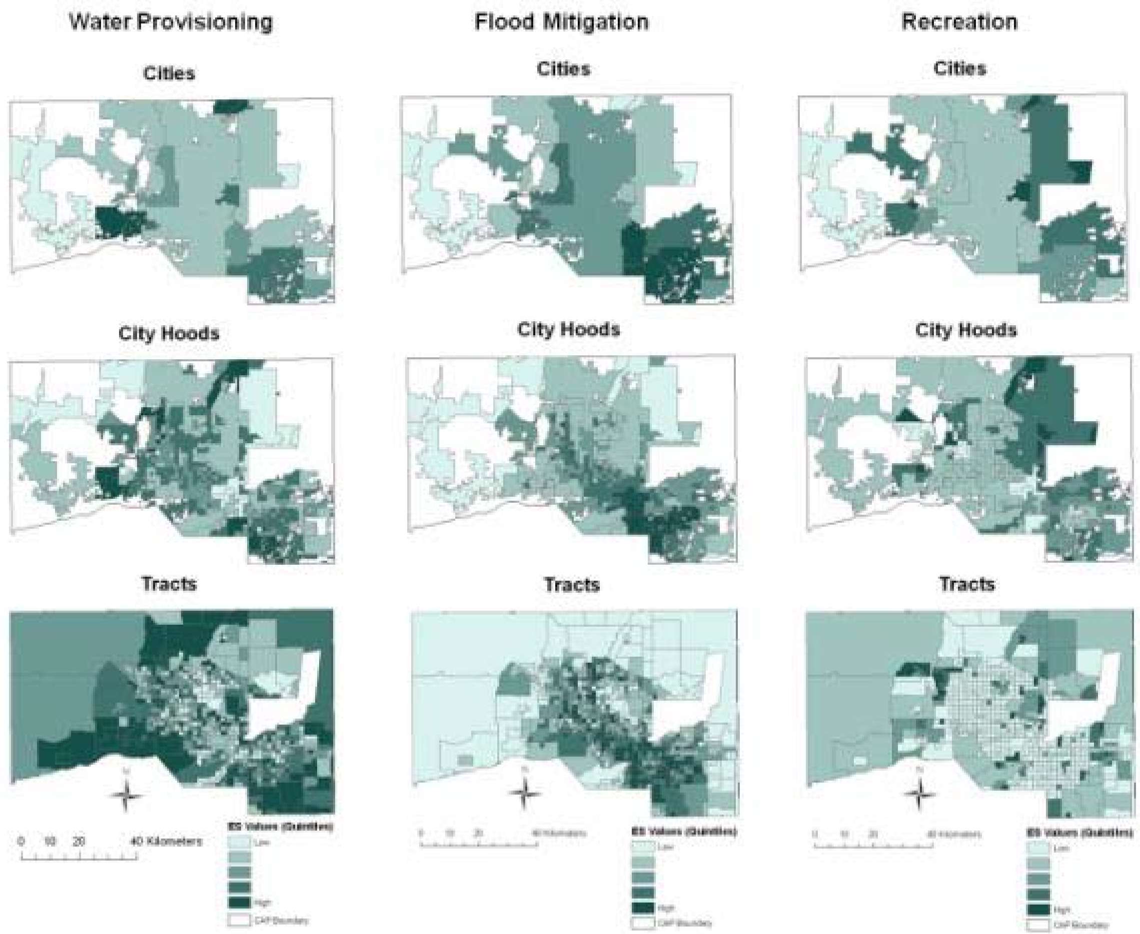

Figure 5.2: Examples of the spatial distribution of three hydrologic ecosystem services at the three levels of spatial resolution. In this example, the proxies for water provisioning are production wells, the proxies for flood mitigation are drywells, and the proxies for recreation are golf courses. Ecosystem service provisioning is represented by quintiles.

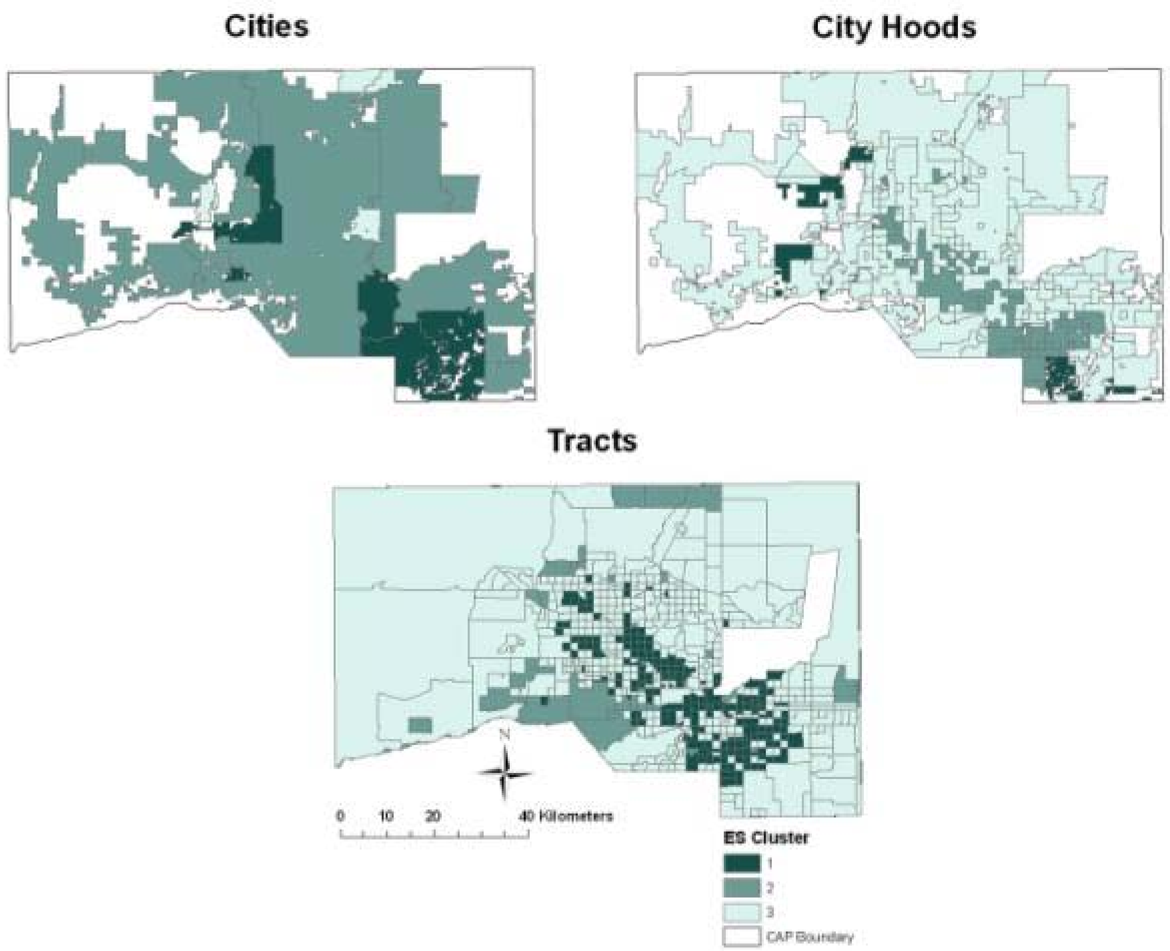

Figure 5.3: Location of ecosystem service cluster bundles at the tract, city hood, and city scales. Note that each cluster identity was determined independently for each spatial scale, thus cluster 1 for tracts ≠ cluster 1 for city hoods, etc.

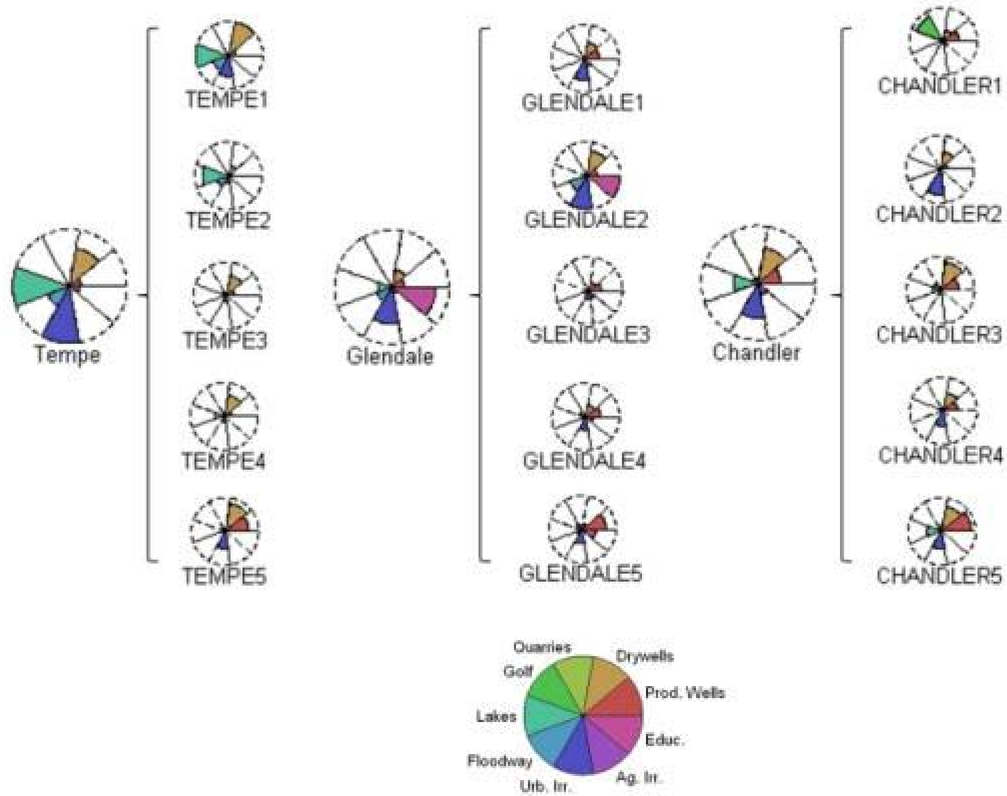

Figure 5.4: Flower plots of ecosystem service provisioning for three cities and their component city hoods. For the city hoods, each number after the name of the city indicates the mean income quintile (1 = lowest, 5 = highest).

Chapter 6

CONCLUSION

6.1 Summary

Traditionally, ecologists have focused their research on relatively pristine ecosystems. Since the industrial revolution, humans have had an ever-increasing impact on the environment, and some ecologists in the 20th century began to study the repercussions of intensive ecosystem use, such as agriculture or mining. However, humans were largely seen only as exogenous drivers of these relatively simple, homogeneous ecosystems where biogeochemical variability is damped (Collins et al. 2000). Researchers shied away from complex, heterogeneous urban systems. But as the proportion of the world's population who are city dwellers continues to increase (UNPD 2008), environmental challenges become more widespread, and new paradigms of ecosystem science emerge, more and more scientists have turned their attention to the ecology of cities.

To develop robust observations, experiments, and theories, humans can no longer be viewed as external to the system. Instead, the conceptual framework must include feedbacks between socioeconomic drivers, patterns of human activity, and ecosystem dynamics (Grimm et al. 2000, Kaye et al. 2006, Collins et al. *In press*). The research presented in this dissertation was initially based on the CAP LTER framework for urban ecosystems (Fig. 6.1), which is an adaptation of the broader LTER framework for integrating socio-ecological systems. However, one researcher cannot address all components and interactions within a complex socio-ecological system, so the focus was narrowed to designed ecosystems combining water and nitrogen, with attention paid to ecological functions such as denitrification and infiltration, processes that underlie the ecosystem service of water-quality regulation.

Although several analyses of the range of ecosystem services derived from either urban areas or freshwater ecosystems have appeared in the literature (Postel and Carpenter 1997, Bolund and Hunhammar 1999, Wilson and Carpenter 1999, Tratalos et al. 2007, Phaneuf et al. 2008), none has explicitly considered designed ecosystems or urban aquatic ecosystems. Chapter 1 describes the range of designs and manipulations that have occurred to meet the needs of urban populations in arid ecosystems, and introduces the concept of "flowers" of ecosystem services (Table 1). Because management of water is a vital component for the provisioning of a wide range of ecosystem services – from vital needs such as clean drinking water and waste removal to aesthetic, cultural, and recreational desires for lush vegetation and lakes – there are numerous potential types of designed ecosystems, some intended for more than one purpose, others with a more narrow use. Within the larger urban ecosystem, there needs to be a mixture of component ecosystems that can provide the range of needed and desired ecosystem services, balancing tradeoffs between services among locations. We proposed that use of flowers of ecosystem services could aid in planning, assessment, and management of urban regions.

Aquatic ecosystems of various designs are located across a diverse set of land covers and uses that comprise the urban ecosystem, yet very little research has focused on these ecosystems nor quantified their relative abundance or spatial distribution. We described the pattern of designed aquatic ecosystems within the larger urban ecosystem in order to understand their contextual relevance. In this research we focused on two common designed ecosystems: artificial lakes and stormwater retention basins. In Chapter 2, we demonstrated the ways in which human engineering and institutions have acted upon the geomorphic template to create novel, designed ecosystems in the urban desert of the Phoenix metropolitan area (Table 6.1). The extent of relatively small-scale changes to the hydrogeomorphology of the region is dramatic, made even more so by the lack of official (municipal agency/planning organization), spatially explicit data for both types of ecosystems. Given that climate-change models for the US Southwest predict the region to become drier and for rainstorms to be more frequent and intense (Karl et al. 2009), accurate awareness of the existence of these ecosystems will be vital for adaptation planning.

If aquatic ecosystems are providing important ecosystem services within the urban environment, they must be both prevalent and support ecosystem process rates that underlie the desired services. We investigated biogeochemical processes associated with nitrogen (N) removal in these designed ecosystems. For retention basins, we conducted an extensive survey and intensive measurements, described in Chapter 4 (Table 1). For artificial lakes, there are numerous design elements that could potentially influence process rates, including lake area and volume, water source, loss to groundwater vs. evaporation (or removal for irrigation), landscape context, algae management, and fish stocking. We were not able to conduct a large enough field study to adequately control for all of these variables, and our analysis of GIS data and government records indicates that for many lakes in the region this basic information is unknown. For the lakes we did sample (reported in Chapter 2), while we found relatively high rates of potential denitrification in lake sediments, it is likely that this process is limited by the rate of diffusion of the nitrate into the sediment (Roach 2005, Seitzinger et al. 2006). Further assessment of the contribution of lakes to the larger urban ecosystem N budget would require investigation of range and frequency of design and management practices along with evaluation of N biogeochemistry.

Some of the characteristic design elements of retention basins can be estimated from GIS analyses of drywell locations, and our survey of 32 basins in the Phoenix area (reported in Chapter 4) revealed that landscaping choice (grassy vs. xeriscaped) is likely to be one of the most important drivers of N biogeochemistry in these systems. However, there are other unknown drivers that are likely to have a significant impact on N removal and export to groundwater from retention basins, including local soil compaction and infiltration rates, irrigation and fertilization regimes, mowing practice (frequency and possible removal of clippings), N load per storm event (likely to be influenced by time since prior storm, Lewis and Grimm 2007), etc., and it is possible that these factors vary substantially throughout the valley. Nevertheless, our analysis shows that the spatial pattern of retention basins is likely to be skewed more toward xeriscaped basins than grassy ones, indicating that on a region-wide scale these locations are unlikely to contribute significantly to overall N removal via denitrification, but may instead be exporting N to groundwater systems.

Synthesizing the work on spatial distribution, abundance and biogeochemical processes, I conclude that grassy retention basins may be important providers of the ecosystem service of N removal, whereas xeriscaped basins and artificial lakes are not. The dynamics of N biogeochemistry in artificial lakes and stormwater retention basins are the result of both natural and human drivers. The human effects on ecosystem processes are intentional and unintentional, stemming from engineering practices, socio-economic considertions, cultural values, and historic legacies. These complex interactions typify the "unique urban biogeochemistry" identified by Kaye et al. (2006).

The work described thus far has focused on the identification of landscape patterns and exploration of their influence on ecosystem processes, or Box A in Figure 6.1. But, beyond creating patterns of novel ecosystems with unique processes, humans also respond their environment, preferentially using or avoiding some areas, changing management practices, and potentially creating new designed ecosystems. The signals to which people respond are changes to amenities or disamenities that ecological components help create or ameliorate—in other words, ecosystem services. One way to assess the impact of these signals is to conduct an economic analysis of the value of ecosystem services. However, many do not have existing markets, and so require alternative methods of valuation. In Chapter 3, we used hedonic price modeling to estimate the value of environmental amenities in the Phoenix metropolitan area. The price of a single family residence is determined not only by the physical attributes of the house itself, but also environmental and locational aspects embodied in the house as well. We explored the value of several amenities and disamenities, both by city and for the entire region. Of particular note is the universal value added by increases in vegetation abundance – a water-intensive activity in this desert environment. Assessing the value of this amenity is relatively easy because a there is an existing market proxy.

To evaluate the ecosystem service of N removal and water-quality improvement is more difficult. One approach would be to use the replacement cost approach, where the value is estimated by the cost incurred for water-quality improvement by engineering. However, there are limitations to this approach. First, while the research presented in Chapter 4 was able to describe significant differences between grassy and xeriscaped basins, there are still many unknown factors that constrain the degree to which we fully understand the biogeochemical processes, as mentioned above. Without sufficient knowledge of the underlying ecosystem functions, quantification, much less valuation of ecosystem services is impossible.

Secondly, it is unclear if the ecosystem services potentially provided by N removal are relevant within this context. The demand for water-quality improvement is highly location-specific. If the water entering retention basins is not being used in a way that requires high water quality, then there is no demand for that ecosystem service at that time and location. We did find relatively high levels of N infiltrating to deeper soil layers from retention basins, which could be affecting groundwater quality. Since groundwater is used in some locations of the Phoenix region for drinking water, it is possible that N removal by retention basins could offset treatment costs further down the line. However, quantifying this ecosystem service and evaluating its benefit would require more investigation into the quality of infiltrating water, as well as a sufficient spatially and temporally explicit understanding of groundwater dynamics, which was beyond the scope of this dissertation.

Finally, the spatial component of ecosystem services was explored in Chapter 5, which mapped a selection of hydrologic ecosystem services in the Phoenix region (Table 1). This work demonstrates that certain ecosystem services are location-specific, whether constrained by geomorphology or as the result of human engineering. That is, the capacity of an ecosystem to provide ecosystem services (via the underlying ecosystem functions occurring in the system) does not necessarily mean that the services are being utilized; in other words, ecosystem functions are not equal to ecosystem services. It is desired ecosystem functioning in a specific desired location – a combination of pattern and process –

that yield ecosystem services. Therefore, I have developed a new framework representing my approach in this dissertation (Figure 6.2). In this conceptual framework, human engineering and institutions have interacted with the general geomorphic template of the region to create a spatially explicit pattern of component ecosystems (or patch types, in landscape ecology). The pattern of these ecosystems interacts with ecological processes within this landscape. When specific processes occur within specific ecosystems, then ecosystem services arise. Note that not every interaction of pattern and process will yield an ecosystem service. Then, it is through valuation of these ecosystem services (via traditional markets or not) that human institutions are influenced to maintain systems as they are, or to potentially alter the geomorphic template further. This conceptual framework represents the scope of my research interests and outlines a potential research program for identifying the patterns of designed ecosystems in urban socio-ecological systems, quantifying the ecological processes within them, and assessing the value of the ecosystem services that arise. Additionally, an historical or long-term analysis could include evaluation of the changes in human engineering practices and institutions that alter the landscape. Although this program was not fully implemented for the nitrogen-removal service across all designed ecosystem types in Phoenix metro, the combined approaches could be applied to other ecosystem services for which extant data are more readily available, enabling researchers and managers to close the loop for the integration of social and ecological processes.

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Wilson, M. A., and S. R. Carpenter. 1999. Economic valuation of freshwater ecosystem services in the United States: 1971-1997. Ecological Applications **9**:772-783.

Table 6.1. Citations and current status of publication products from this dissertation.

Chapter	Citation	Status
1	Larson, E. K., S. R. Earl, E. Hagen, R. Hale, H. Hartnett, M. McCrackin, M. McHale, and N. B. Grimm. In review. Beyond restoration and into design: hydrologic alterations in aridland cities.*in* S. T. A. Pickett, M. L. Cadenasso, B. McGrath, and K. Hill, editors. Resilience in Urban Ecology and Design: Linking Theory and Practice for Sustainable Cities.	In review
2	Larson, E. K., and N. B Grimm. Small-scale and extensive hydrogeomorphic modification and water redistribution in a desert city and implications for regional nitrogen removal	In preparation for Urban Ecosystems
3	Larson, E. K., and C. Perrings. The value of water-related amenities in an arid city: The case of the Phoenix Metropolitan Area	In preparation for Landscape and Urban Planning
4	Larson, E. K., and N. B. Grimm. Landscape design and the fate of nitrate in stormwater retention basins in the Phoenix, AZ Metropolitan Area	In preparation
5	Larson, E. K., J. M. Anderies, and N. B. Grimm. Locally provided hydrologic ecosystem services in the Phoenix Metropolitan Area: Potential bundles andtradeoffs.	In preparation

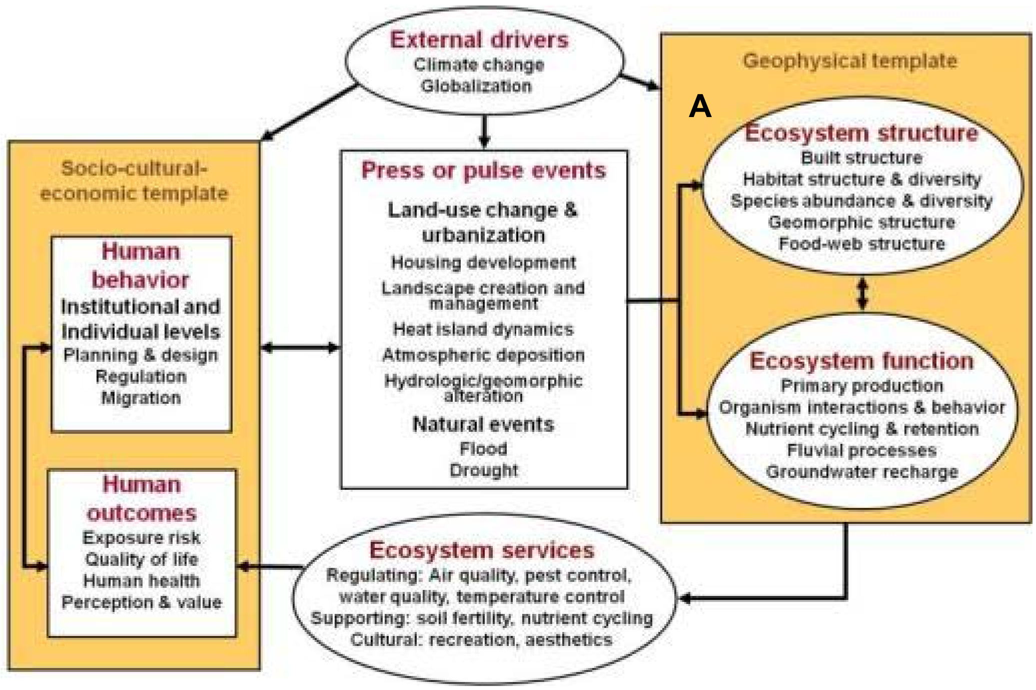

Figure 6.1: Conceptual framework for the CAP LTER program.

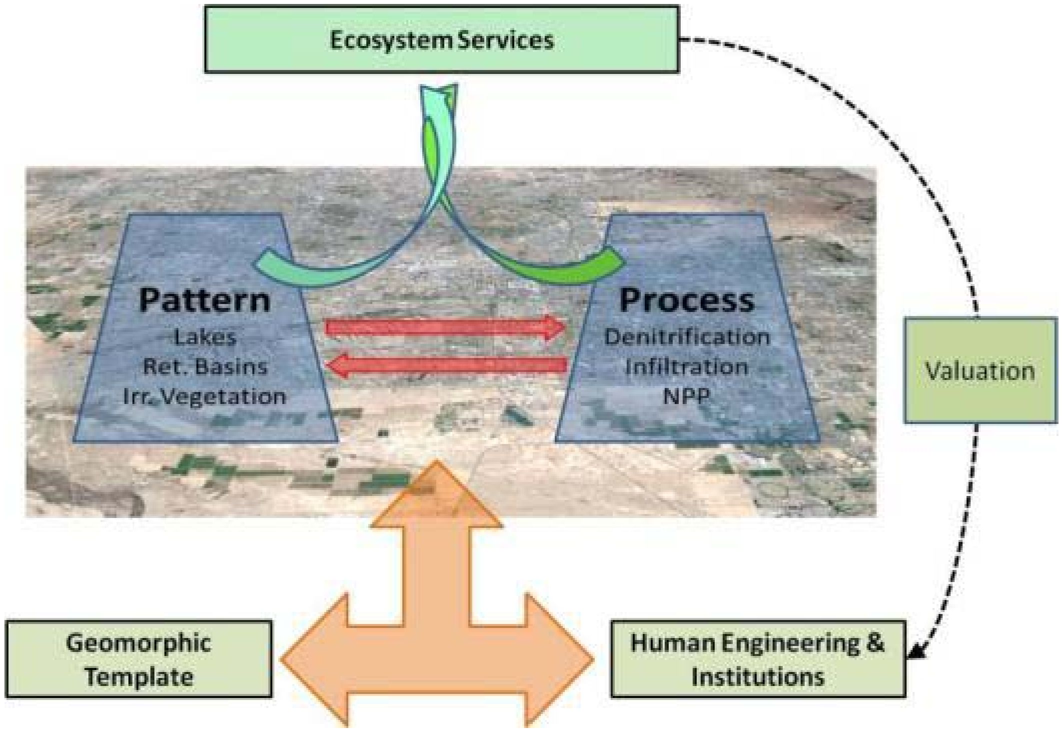

Figure 6.2: Conceptual framework developed in this dissertation.

APPENDIX I. Characteristics of retention basin soils by design and depth. Samples were taken at sequential depths from a pit, up to 60 cm (or the deepest a pit could be dug). Nitrate and ammonium data are from 2M KCl extracts, Chloride data from water extracts. H_2O is gravimetric water content, oM is organic matter, calculated as loss on ignition.

Site	Design	Depth Cm	NO_3^- µg N/g soil	NH_4^+ µg N/g soil	Cl^- µg/g soil	H_2O	oM
GW	Xeric	0-5	4.22	3.87	16.10	0.01	2.08
GW	Xeric	5-10	0.42	0.78	16.68	0.02	1.71
GW	Xeric	10-15	0.67	1.07	25.74	0.01	2.09
GW	Xeric	15-20	0.62	0.93	15.07	0.12	2.71
GW	Xeric	30	0.75	1.86	16.33	0.14	3.50
GW	Xeric	40	0.69	2.86	24.14	0.13	3.27
GW	Xeric	50	0.79	1.09	10.30	0.12	3.54
RR	Xeric	0-5	6.43	1.27	44.89	0.21	8.34
RR	Xeric	5-10	2.19	1.32	28.50	0.15	3.03
RR	Xeric	10-15	1.34	1.17	22.34	0.09	1.11
RR	Xeric	15-20	1.46	1.42	20.75	0.11	1.27
RR	Xeric	30	1.57	1.56	23.94	0.22	2.24
RR	Xeric	40	0.90	1.81	16.71	0.21	2.21
RR	Xeric	50	0.78	1.03	16.35	0.21	2.07
RR	Xeric	60	0.46	0.92	23.94	0.21	2.13
PMX	Xeric	0-5	6.17	2.17	17.27	0.04	4.66
PMX	Xeric	5-10	1.29	1.17	9.29	0.04	2.56
PMX	Xeric	10-15	0.45	0.81	6.13	0.04	0.94
PMX	Xeric	15-20	0.44	0.95	8.13	0.12	2.55
PMX	Xeric	30	0.42	1.02	7.77	0.11	3.21
PMX	Xeric	40	0.34	0.99	6.24	0.09	3.36
PMX	Xeric	50	0.36	0.98	7.69	0.08	3.28
PMX	Xeric	60	0.38	1.22	8.99	0.11	3.11
POL	Xeric	0-5	10.31		9.07	0.04	5.99
POL	Xeric	5-10	1.02		9.51	0.10	1.59
POL	Xeric	10-15	1.17		7.38	0.16	2.38
POL	Xeric	15-20	1.64		8.46	0.17	2.40
POL	Xeric	30	0.82		7.61	0.16	2.82
POL	Xeric	40	0.56		6.99	0.17	2.82
POL	Xeric	50	0.88		6.24	0.16	3.00
POL	Xeric	60	0.84		7.54	0.15	2.93

Site	Design	Depth Cm	NO_3^- µg N/g soil	NH_4^+ µg N/g soil	Cl^- µg/g soil	H_2O	oM
VV	Xeric	0-5	17.50	6.08	20.90	0.01	3.23
VV	Xeric	5-10	1.30	0.29	7.75	0.06	0.68
VV	Xeric	10-15	1.20	0.34	10.99	0.09	2.45
VV	Xeric	15-20	0.92	0.31	11.97	0.11	2.74
VV	Xeric	30	0.35	0.59	11.49	0.12	3.01
VV	Xeric	40	0.28	0.47	12.10	0.12	2.69
VV	Xeric	50	0.27	0.66	12.44	0.14	3.44
VV	Xeric	60	0.30	0.78	15.13	0.13	3.25
PMG	Grassy	0-5	1.51	3.01	107.68	0.08	8.17
PMG	Grassy	5-10	0.40	1.61	39.06	0.09	5.51
PMG	Grassy	10-15	0.35	1.45	40.77	0.10	3.87
PMG	Grassy	15-20	0.33	1.23	41.77	0.09	3.08
PMG	Grassy	30	0.45	1.00	15.51	0.13	3.22
PMG	Grassy	40	0.33	0.98	12.34	0.14	3.23
PMG	Grassy	50	0.70	0.89	9.21	0.15	3.25
PMG	Grassy	60	0.87	0.80	9.05	0.15	3.30
ER	Grassy	0-5	1.64	3.55	97.13	0.14	14.83
ER	Grassy	5-10	1.17	1.49	50.55	0.11	8.32
ER	Grassy	10-15	0.49	1.00	23.25	0.08	3.58
ER	Grassy	15-20	0.33	0.94	21.52	0.08	3.89
ER	Grassy	30	0.29	1.30	18.59	0.08	1.90
ER	Grassy	40	0.25	1.38	13.33	0.10	2.47
ER	Grassy	50	0.29	1.50	26.04	0.15	2.93
WW	Grassy	0-5	2.47	7.82	160.50	0.40	19.64
WW	Grassy	5-10	1.62	3.37	158.08	0.29	13.75
WW	Grassy	10-15	0.52	2.09	60.52	0.15	5.44
WW	Grassy	15-20	0.31	1.23	23.60	0.13	3.03
WW	Grassy	30	0.18	0.93	17.87	0.12	2.57
WW	Grassy	40	0.16	1.18	16.48	0.12	2.83
WW	Grassy	50	0.20	0.76	16.50	0.12	2.00
WW	Grassy	60	0.19	0.62	17.27	0.12	2.09
CV	Grassy	0-5	0.46	1.79	57.18	0.17	2.58
CV	Grassy	5-10	0.28	1.54	64.82	0.17	2.28
CV	Grassy	10-15	0.19	0.99	97.33	0.20	2.14
CV	Grassy	15-20	0.25	0.56	83.27	0.19	2.34
CV	Grassy	30	0.13	0.23	32.34	0.09	1.14
CV	Grassy	40	0.12	0.33	25.39	0.10	1.39

Site	Design	Depth Cm	NO_3^- μg N/g soil	NH_4^+ μg N/g soil	Cl^- μg/g soil	H_2O	oM
CV	Grassy	50	0.13	0.13	37.14	0.08	1.11
CV	Grassy	60	0.12	0.17	38.88	0.09	1.11
NC	Grassy	0-5	1.82	9.90	235.41	0.32	22.66
NC	Grassy	5-10	1.11	3.21	114.68	0.16	20.95
NC	Grassy	10-15	0.24	2.41	52.92	0.16	4.10
NC	Grassy	15-20	0.31	2.37	36.50	0.16	2.97
NC	Grassy	30	0.16	2.44	20.77	0.16	2.67
NC	Grassy	40	0.14	3.09	23.75	0.17	2.54
NC	Grassy	50	0.16	2.73	17.45	0.16	2.59
NC	Grassy	60	0.15	2.90	19.53	0.15	2.63

APPENDIX II: Isotope results for gas samples from basin flooding experiments. At 2 retention basins (1 grassy, 1 xeriscaped) simulated a 1-in. storm and flooded the basin. The water had a concentration of 1 mg N/L at 5000‰ $\delta^{15}N$. Prior to flooding, we collected soil samples to assess background N concentrations and $\delta^{15}N$. During the flood event, we collect standing water and lysimeter samples. After most of the water had infiltrated, we installed chambers in the soil. Lids were attached to create a chamber, and gas samples taken initially and after a period of 30 minutes. We also collected soil samples again to evaluate changes in N concentrations and $\delta^{15}N$. Gas samples were analyzed for $[N_2]$ and $[N_2O]$, as well as $\delta^{15}N - N_2$ and $\delta^{15}N - N_2O$, soil extracts were analyzed for $[NO_3^-]$, $[NH_4^+]$, as well as $\delta^{15}N - NO_3^-$ and $\delta^{15}N - NH_4^+$.

Table 1. Isotope results for standing water and lysimeter samples.

Timestep	**Station**	**Water temp.** °C	**δ^{15}N-N$_2$** per mil	**Sample N$_2$ mass** mmoles	**Standing Water N$_2$ conc.** mmols/L	**δ^{15}N-N$_2$O** per mil	**Sample N$_2$O mass** nmoles	**Standing Water N$_2$O conc.** nmoles/L
Grassy								
0.00	E	32.6	-1.31	0.089	0.109	311.04	0.293	13.93
0.50	W	30.4	-1.13	0.126	0.166	355.67	0.367	18.19
0.50	E	30.8	-2.46	0.063	0.077	19.98	0.043	2.00
1.00	W	28.6	-1.84	0.058	0.077	130.80	0.301	15.42
1.00	E	29.4	-1.56	0.107	0.131	140.45	0.417	19.60
4.0	W	34.3	-1.63	0.092	0.104	203.61	0.493	21.26
4.0	E	35.1	-0.98	0.175	0.199	102.54	0.471	20.53
4.0	M	33.4	-1.45	0.101	0.114	83.77	0.494	21.31
6.0	W	38.2	-1.47	0.073	0.095	306.81	0.468	22.75
6.0	E	37.1	-1.16	0.114	0.139	265.67	0.467	21.64
6.0	M	37.6	-1.21	0.071	0.085	298.15	0.475	21.18

Timestep	**Station**	**Water temp.** °C	**δ^{15}N-N_2** per mil	**Sample N_2 mass** mmoles	**Standing Water N_2 conc.** mmols/L	**δ^{15}N-N_2O** per mil	**Sample N_2O mass** nmoles	**Standing Water N_2O conc.** nmoles/L
Xeriscaped								
0.0	S	20.0	-1.23	0.130	0.133	220.42	0.355	14.23
0.0	M	20.0	-0.84	0.211	0.246	-10.78	0.213	9.74
0.5	S	21	-1.26	0.131	0.142	21.20	0.547	23.18
0.5	M	21	-1.51	0.076	0.088	11.28	0.448	20.25
0.5	N	21.0	-1.40	0.107	0.107	-9.32	0.505	19.52
2.0	S	23.0	-2.14	0.107	0.163	-2.1	0.420	25.38
2.0	M	24.8	-1.84	0.064	0.082	5.93	0.443	22.06
2.0	N	22.0	-1.65	0.066	0.084	-152.49	0.490	24.57
4.0	S	25.5	-1.83	0.078	0.096	29.60	0.657	31.04
4.0	M	26.1	-1.72	0.089	0.109	1096.86	0.536	25.00
4.0	Lysimeter	26.1	-1.44	0.087	0.099	2884.71	1.080	46.43
7.0	S	29.0	-0.30	0.111	0.113	4058.50	0.752	28.70
7.0	M	29.0	0.78	0.072	0.086	6121.30	0.658	29.52
7.0	Lysimeter	29.0	3.84	0.110	0.117	8194.47	2.202	83.61

Table 2: Chamber N_2 isotope data. Time indicates the hour and whether the sample is the initial or final sample. Chamber lids placed on the cylinders, an initial sample taken right away, and then a final sample taken after 30 minutes.

Design	**Station**	**Time**	**Temp**	**µmol N_2**	**$\delta^{15}N$**	**Rate N_2 flux ($\mu g\ m^{-2}\ h^{-1}$)**	**$\Delta[N_2]$ $\mu mol\ h^{-1}$**
Grassy	E	8 init.	33.0	406.33	5.21	41.28	0.15
		8 final		406.48	5.57		
Grassy	M	8 init.	34.6	422.46	5.43	59.73	-17.98
		8 final		404.48	5.73		
Grassy	W	8 init.	33.7	418.06	4.89	21.64	-6.43
		8 final		411.63	5.04		
Grassy	E	11 init.	36.1	408.26	5.33	378.79	3.64
		11 final		411.90	5.72		
Grassy	M	11 init.	36.1	416.13	4.97	194.85	-7.38
		11 final		408.76	5.71		
Grassy	W	11 init.	36.3	415.34	5.53	1703.75	-3.71
		11 final		411.63	5.75		
Xeriscaped	N	9 init.	34.7	411.95	3.40	213.64	5.74
		9 final		417.69	5.61		
Xeriscaped	M	9 init.	34.7	401.09	3.31	388.62	12.60
		9 final		413.69	6.29		
Xeriscaped	S	9 init.	34.7	433.57	3.36	254.07	-21.22
		9 final		412.35	5.76		
Xeriscaped	N	12 init.	38.5	414.83	4.20	72.38	-16.12
		12 final		398.71	5.25		
Xeriscaped	M	12 init.	38.5	404.53	3.87	120.80	-8.91
		12 final		395.62	5.33		
Xeriscaped	S	12 init.	38.9	399.12	3.60	171.28	11.19
		12 final		410.31	5.51		

APPENDIX III. Infiltration data, determined by tension infiltrometer. Variables include the Hydrolic Head (H1 and H2), Infiltration rate (Q1 andQ1), and saturated hydraulic conductivity (ksat). For most of the locations, I was unable to get convergent ksat calculations from the data.

Site	Design	*Head 1*	*Head 2*	*Q1 (cm3/h)*	*Q2 (cm3/h)*	*ksat (cm/hr)*	*ksat (cm/yr)*
Greenway	Xeriscaped	-10.5	-8.2	4663.49	4663.49		0.00
		-10.5	-3.4	4663.49	5596.19	3.26	50.90
		-8.2	-3.4	4663.49	5596.19	4.66	72.66
Polaris	Xeriscaped	-9.4	-6.2	9326.98	11192.37	15.68	244.65
		-9.4	-2.8	9326.98	18653.96	36.02	561.95
		-6.2	-2.8	11192.37	18653.96	48.96	763.82
Roadrunner	Xeriscaped	-10.2	-3	2798.09	7461.58	18.48	288.27
		-8.2	-3	2798.09	7461.58	24.98	389.64
Vista Verde	Xeriscaped	-6.3	-3.2	9326.98	11192.37	13.59	212.00
		-10.7	-3.2	7461.58	11192.37	12.63	196.96
		-10.7	-6.3	7461.58	9326.98	11.64	181.62
Whispering Wind	Grassy	-10	-6.2	1865.40	2798.09	7.87	122.79
		-10	-3.2	1865.40	7461.58	28.08	438.06
		-6.2	-3.2	2798.09	7461.58	48.68	759.44
Eagle Ridge	Grassy	-11.3	-3.1	932.70	6528.88	28.23	440.46
		-6	-3.1	932.70	6528.88	139.89	2182.34
North Canyon	Grassy	-11	-6	932.70	1865.40	7.11	110.95
		-11	-2.2	932.70	3730.79	9.29	144.93
		-6	-2.2	1865.40	3730.79	10.45	163.02
Cactus View	Grassy	-6.2	-3.4	3730.79	5596.19	15.51	242.00
		-10.2	-3.4	2798.09	5596.19	11.20	174.77
		-10.2	-6.2	2798.09	3730.79	6.70	104.47

APPENDIX IV. Plot-level isotope addition data. At 10 retention basins (5 grassy, 5 xeriscaped) applied 3L of water with a concentration of 1 mg N/L at 5000‰ $\delta^{15}N$ to two plots delineated by a circular column embedded in the soil. The area of the column was 491 cm^2, and the height above the surface ranged from 10 – 14 cm high (depending on how deeply the columns were installed). We collected nearby soil samples to assess background N concentrations and $\delta^{15}N$. After most of the water had infiltrated, lids were attached to create a chamber, and gas samples taken initially and after a period of 30 minutes. The next day and 8 days later, we collected soil samples from within the plot to evaluate changes in N concentrations and $\delta^{15}N$. Gas samples were analyzed for [N_2] and [N_2O], as well as $\delta^{15}N$ – N2 and $\delta^{15}N$ – N2O.

Table 1. Soil inorganic N content and $\delta^{15}N$ by basin design, plot number, and time since addition.

Site	Design	Plot	Time	NO_3^- mg	$^{15}NO_3^-$ mg	$\delta^{15}N$ - NO_3^-	NH_4^+ mg	$^{15}NH_4^+$ mg	$\delta^{15}N$ - NH_4^+
WW	Grassy	1	Background	434.9	1.6	-10.8	47.9	0.2	-6.6
WW	Grassy	1	24 h	514.7	1.9	4.1	169.2	0.6	-9.8
WW	Grassy	1	Day 8	954.9	1.6	-7.0	168.4	0.1	-12.9
WW	Grassy	2	Background	386.6			162.4		
WW	Grassy	2	24 h	291.3	1.1	-9.2	142.8	0.5	-4.2
WW	Grassy	2	Day 8	267.0	2.9	-1.5	745.0	0.2	-2.6
CV	Grassy	1	Background	196.7	0.7	-13.9	294.9	1.1	-69.5
CV	Grassy	1	24 h	274.1	1.0	-4.4	43.2	0.2	-16.9
CV	Grassy	1	Day 8	115.4	1.5	-14.5	249.7	0.1	-10.8
CV	Grassy	2	Background						
CV	Grassy	2	24 h	242.3	0.9	16.4	67.2	0.2	-19.0
CV	Grassy	2	Day 8	218.8	2.6	-2.8	1065.0	0.2	1.5
NC	Grassy	1	Background	91.1	0.3	-8.3	52.2	0.2	-2.5
NC	Grassy	1	24 h	510.5	1.9		366.0	1.3	

Site	Design	Plot	Time	NO_3^- mg	$^{15}NO_3^-$ mg	$\delta^{15}N$ - NO_3^-	NH_4^+ mg	$^{15}NH_4^+$ mg	$\delta^{15}N$ - NH_4^+
NC	Grassy	1	Day 8	208.3	2.2	238.3	1190.9	0.1	0.2
NC	Grassy	2	Background	91.1	0.3	0.7	52.2	0.2	-1.7
NC	Grassy	2	24 h	280.4	1.0	-6.6	620.5	2.2	-15.2
NC	Grassy	2	Day 8	61.9	2.6	-12.5	650.6	0.2	0.9
PM	Grassy	1	Background	111.9			129.2		
PM	Grassy	1	24 h	10.3		-9.0	56.2		-14.3
PM	Grassy	1	Day 8	151.6	0.1	#DIV/0!	494.8	0.2	-5.7
PM	Grassy	2	Background	70.7	0.3	-11.3	206.6	0.7	-14.4
PM	Grassy	2	24 h	188.1	0.7	-7.9	56.2	0.2	-6.9
PM	Grassy	2	Day 8	30.9			879.6	0.2	-1.4
VV	Xeric	1	Background	776.5	2.8	-6.0	692.5	2.5	-5.9
VV	Xeric	1	24 h	412.9	1.5	-0.8	38.2	0.1	5.8
VV	Xeric	1	Day 8	227.2	1.5	-6.7	68.2	0.1	-13.3
VV	Xeric	2	Background	571.9	2.1	-9.2	338.4	1.2	-18.5
VV	Xeric	2	24 h	718.8	3.4	-1.3	55.1	0.2	1.9
VV	Xeric	2	Day 8	323.2	2.6	-4.2	40.4	0.2	-27.4
RR	Xeric	1	Background	296.6	1.1	-3.9	167.4	0.6	-8.5
RR	Xeric	1	24 h	114.6	0.4	-7.3	97.5	0.4	-9.9
RR	Xeric	1	Day 8	137.2	24.9	0.3	123.7	0.4	8.7
RR	Xeric	2	Background	222.2	0.8	-9.3	112.1	0.4	-7.4
RR	Xeric	2	24 h	131.7	0.5	-8.2	68.4	0.2	-10.0
RR	Xeric	2	Day 8	512.7	0.5	2.2	191.0	0.2	-5.5
PO	Xeric	1	Background	357.9	1.3	-6.5	313.2	1.2	10.1
PO	Xeric	1	24 h	227.0	0.8	3.1	181.9	0.7	2.4

Site	Design	Plot	Time	NO_3^- mg	$^{15}NO_3^-$ mg	$\delta^{15}N$ - NO_3^-	NH_4^+ mg	$^{15}NH_4^+$ mg	$\delta^{15}N$ - NH_4^+
PO	Xeric	1	Day 8	274.1			77.8		
PO	Xeric	2	Background	351.9	1.3	15.0	466.8	1.7	1.1
PO	Xeric	2	24 h	172.0	0.6	22.2	181.9	0.7	22.7
PO	Xeric	2	Day 8	219.4	0.6	-6.1	215.5	0.7	0.5
GW	Xeric	1	Background	582.1	2.1	-12.9	259.0	0.9	-35.6
GW	Xeric	1	24 h	252.6	0.9	20.9	421.9	1.6	6.5
GW	Xeric	1	Day 8	225.4	0.9	4.6	79.0	1.5	-8.8
GW	Xeric	2	Background	468.9	1.7	-9.7	405.9	1.4	-33.2
GW	Xeric	2	24 h	230.2	0.9	13.2	165.6	0.6	8.5
GW	Xeric	2	Day 8	226.4	0.8	6.0	139.9	0.6	-4.5

Table 2. Average of gas sample isotope results by site, design, and time since addition.

Site	Design	Time	^{15}N2O - N ng m^{-2} h^{-1}	N_2O – N μg m^{-2} h^{-1}	N_2 – N μg m^{-2} h^{-1}
RR	Xeric	0.5	9E-07	1387.69	2752.89
RR	Xeric	1		2199.84	2764
RR	Xeric	2		7506.98	-85.922
RR	Xeric	4	1.1E-06	9618.97	26.4429
RR	Xeric	10	3E-05	-2877	70.4735
RR	Xeric	24	3.4E-05	1451.72	2081.52
PO	Xeric	0.5	3.7E-05	7586.31	5041.37
PO	Xeric	1	1.3E-05	9918.6	-13.997
PO	Xeric	2	7.5E-06	11406.1	-10.783
PO	Xeric	4	8.4E-07	7317.1	-3.0166
PO	Xeric	10		1265.58	3855.6
PO	Xeric	24	5.1E-07	0.00327	8.38276
GW	Xeric	0.5	3.1E-05	1642.8	164.713
GW	Xeric	1	2.7E-05	3453.28	65.9838
GW	Xeric	2	4.9E-06	2802.01	4425.39
GW	Xeric	4	1.4E-06	9795.58	780.105
GW	Xeric	10	3E-07	678.528	725.433
GW	Xeric	24	1.2E-05	94.6146	22.8123
VV	Xeric	0.5	1.7E-05	999.301	0.43212
VV	Xeric	1		2628.82	216.521
VV	Xeric	2	9.9E-07	-145.75	-26.012
VV	Xeric	4	1.6E-05	72.7467	1557.2
VV	Xeric	10	9.7E-08	-857.23	462.874
VV	Xeric	24	1E-05	-7887.2	-11.971
CV	Grassy	0.5		-80.312	21.9164
CV	Grassy	1		424.808	19.8936
CV	Grassy	2	3.6E-05	1740.08	-25.067
CV	Grassy	4	2.6E-08	-3495.5	-102.7
CV	Grassy	10	1.5E-06	3147.64	-23.986
CV	Grassy	24	1.8E-05	-2340.6	-192.02
PM	Grassy	0.5	3.7E-05	2920.74	39.9267
PM	Grassy	1	1.2E-05	4223.21	18.5752

Site	Design	Time	$^{15}N2O$ - N ng m^{-2} h^{-1}	N_2O – N µg m^{-2} h^{-1}	N_2 – N µg m^{-2} h^{-1}
PM	Grassy	2	3.9E-05	17629.4	183.154
PM	Grassy	4		26003.2	9.88545
PM	Grassy	10	1.5E-05	-1059.3	91.8459
PM	Grassy	24	4.4E-06	4073.19	0.06919
WW	Grassy	0.5		4229	0.70536
WW	Grassy	1	1.2E-07	11574.3	958.176
WW	Grassy	2	0.00014	13608.8	-150.67
WW	Grassy	4	1.6E-09	20977.8	61.088
WW	Grassy	10		7922.19	2627.75
WW	Grassy	24	2.4E-06	5256.36	-0.2429
NC	Grassy	0.5	6.8E-07	5272.11	104.967
NC	Grassy	1	5.7E-05	-4107.1	1921.44
NC	Grassy	2	2.9E-05	1419.25	-2352.4
NC	Grassy	4	0.00041	-9114.1	-84.577
NC	Grassy	10		784.694	14.8042
NC	Grassy	24		711.73	28.8426

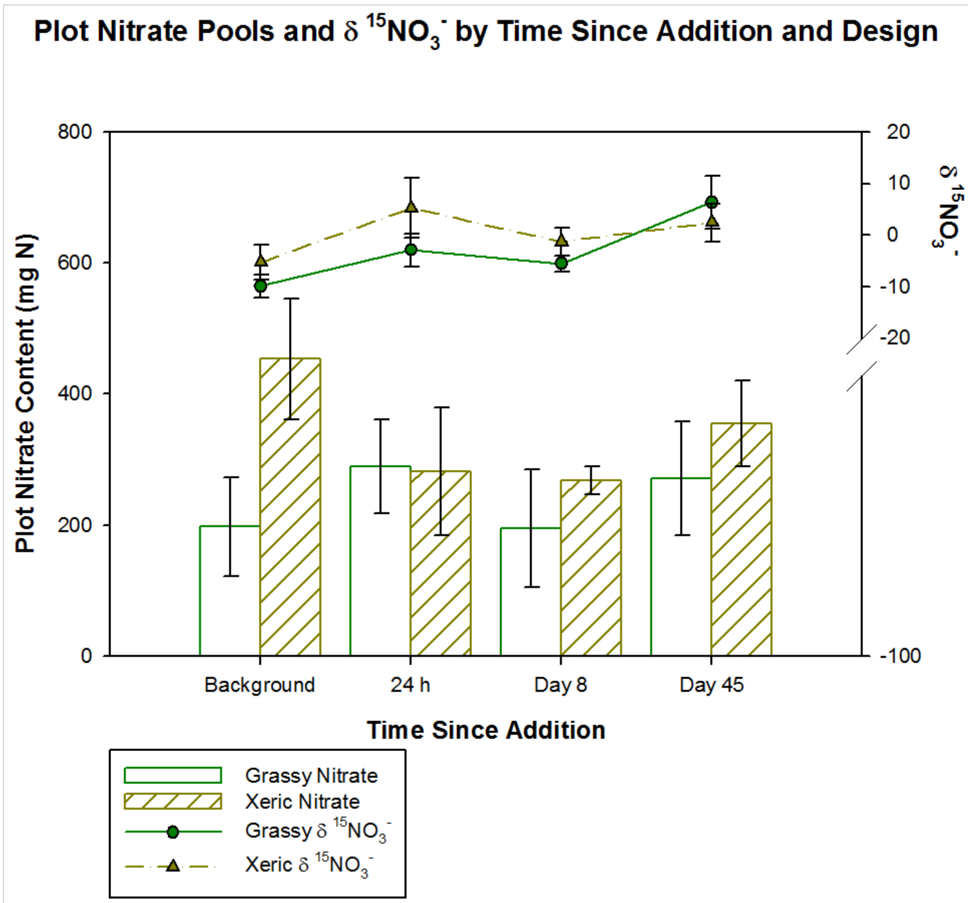

Figure 1: Nitrate pools and $\delta^{15}NO_3^-$ by time since addition and design. Whiskers are ±SE.

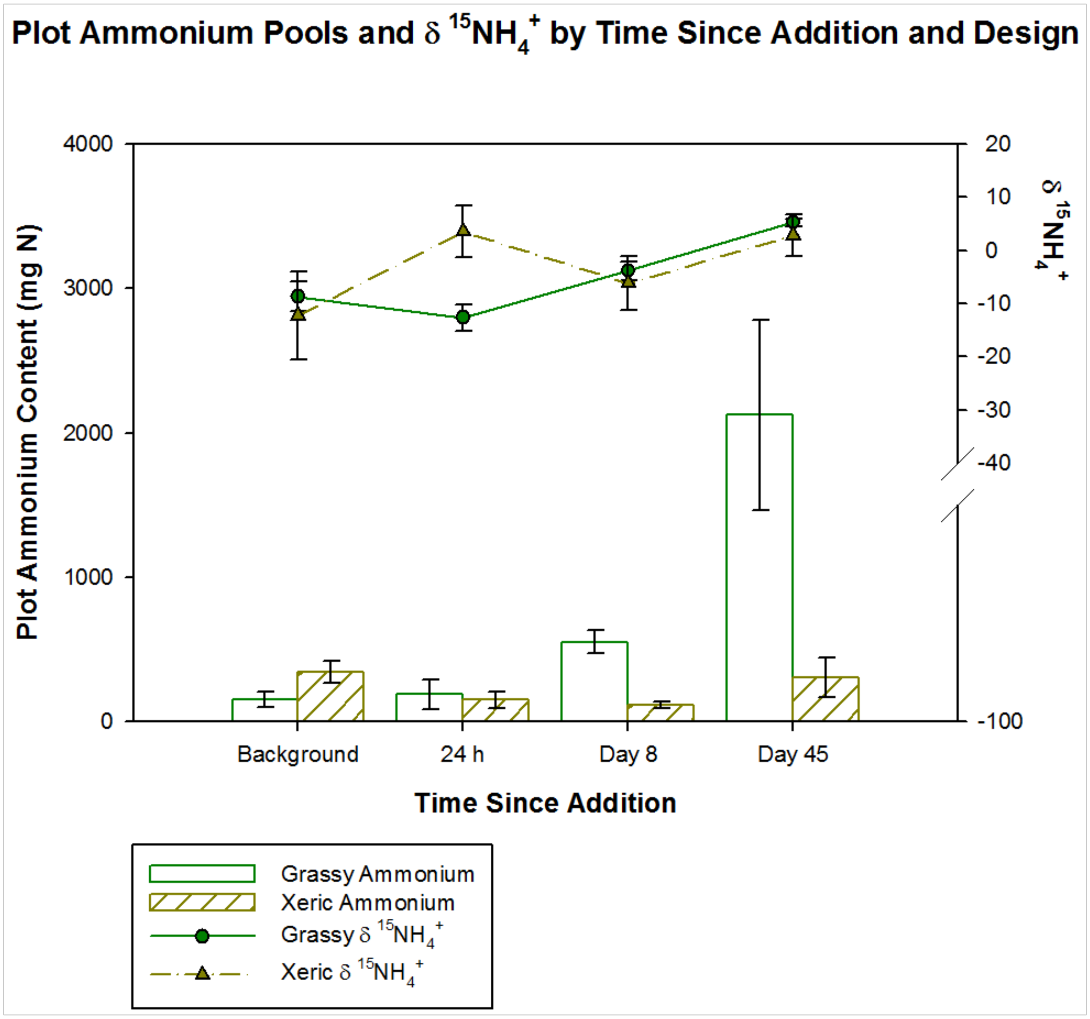

Figure 2: Ammonium pools and δ15NH_4^+ by time since addition and design. Whiskers are ±SE.

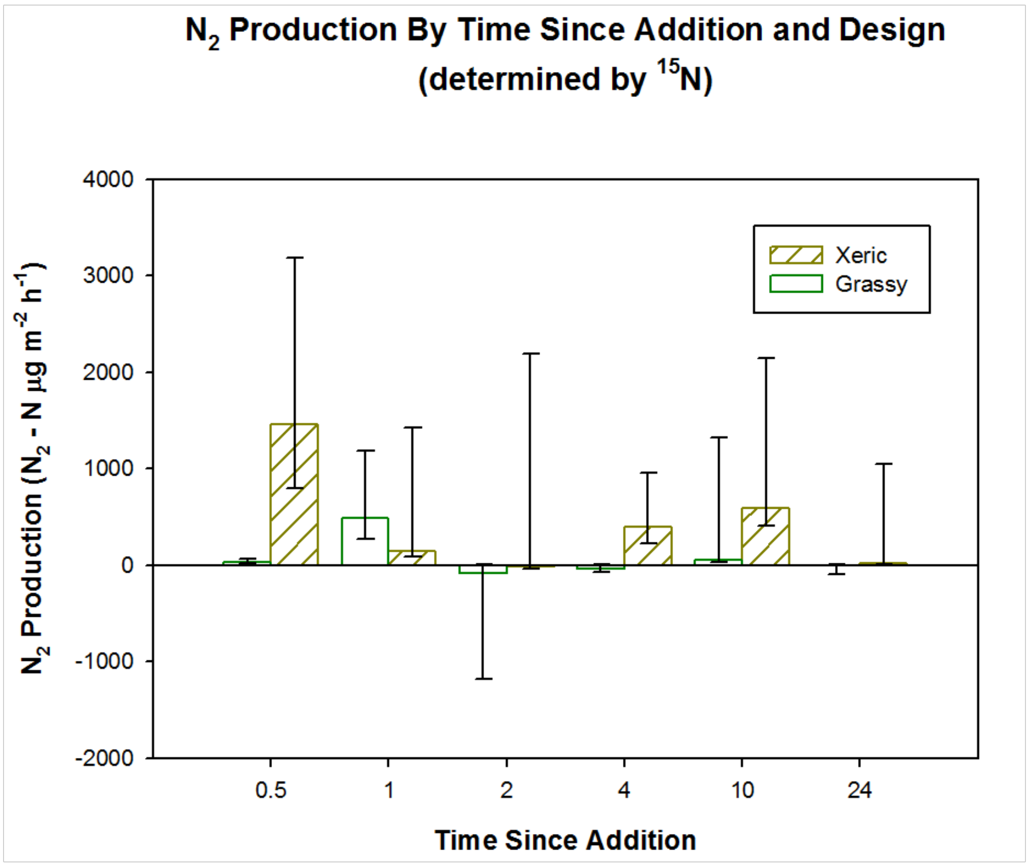

Figure 3: Plot N_2 production by time since addition and design. Whiskers are ±SE.

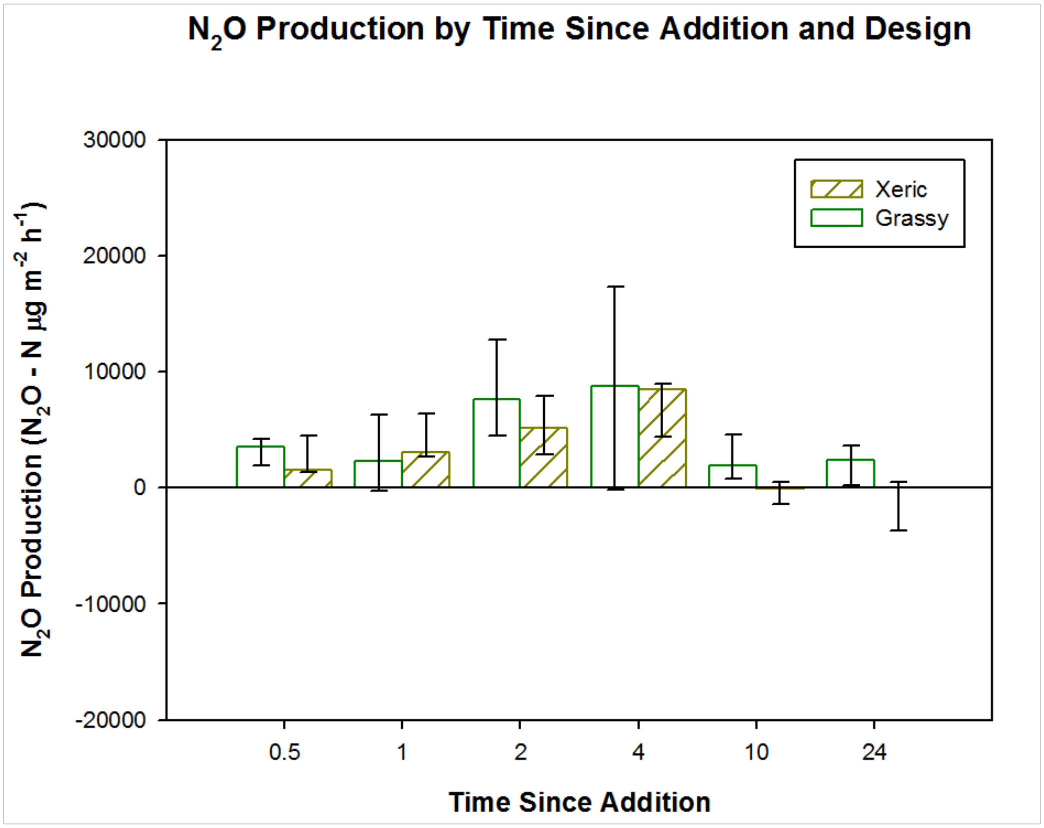

Figure 4: Plot N_2O production by time since addition and design. Whiskers are ±SE.

ECOHYDROLOGY & HYDROBIOLOGY

Vol. 5
No 4, 287-296
2005

Aquatic Habitats
in Integrated Urban
Water Management

The paradoxical ecology and management of water in the Phoenix, USA metropolitan area.

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Abstract

One of the fastest growing cities in the US, the desert city of Phoenix has appropriated significant surface and ground-water resources from regions near and far to support not only basic needs but also various cultural amenities, such as golf courses. Rapid expansion of the metropolitan area has resulted in loss of native ecosystems including desert riparian areas, and creation of new, designed ecosystems that are frequently water-intensive. This article reviews current water resources and management practices, along with resultant ecological impacts. Future legal, socioeconomic, cultural, and environmental challenges to the sustainability of the current lifestyle are highlighted.

Key words: urban ecology, sustainability, aquatic habitat, Arizona, semi-arid ecosystem.

1. Introduction

Located in the northern Sonoran Desert of the southwestern USA, the Phoenix metropolitan area receives approximately 180 mm of precipitation a year, with an average January temperature of 12^{o}C and an average July temperature of 34^{o}C (Baker *et al.* 2004). Most rain is concentrated in two seasons: a summer "monsoon" season with short, intense, localized thunderstorms and a winter rainy season characterized by frontal storms of longer duration and lower intensity. Given its hot, dry climate, the area experiences an average of two meters real evapotranspiration annually. Since this is much higher than the annual precipitation, additional sources of water must be utilized for human habitation. The city is situated in an alluvial valley surrounded by rugged mountain ranges typical of Basin and Range topography (Jacobs, Holway 2004).

Despite its arid climate, humans have lived here since prehistoric times. The valley is situated at the base of more humid upland watersheds. Dryland rivers, the Salt and Verde, provided adequate surface water to support settlement in an area where precipitation falls short of evapotranspiration substantially. The complex civilization of the Hohokam, which was based on irrigated agriculture, persisted for more than 1000 years (Fitzhugh, Richter 2004). Early modern Phoenicians in the late 19th and early 20th centuries resurrected and expanded the ancient canal system, creating vast areas of agricultural production, including citrus, dairy, alfalfa, and cotton crops.

Throughout the 20th century new tactics for stabilizing and increasing water supply to the valley included the establishment of large dams, both within the local watershed and on other distant rivers, requiring trans-boundary water transfer, and an extensive canal network throughout the region. These structural solutions essentially eliminated in-stream flow of the region's rivers, except during extreme flood events.

Now, Phoenix is one of the most rapidly growing cities in the US, increasing from approximately 300 000 in 1950 to greater than 3.7 million in more than 20 municipalities in 2004. Models predict that in 2025, the population will be exceed 6 million, representing a 280% change since 1980 (Jacobs, Holway 2004), and nearly all of the undisturbed and agricultural lands will have been developed to urban land uses (Jenerette, Wu 2001). With few geographical barriers to expansion, growth has been largely in an outward direction, estimated at almost 0.8 km per year (Gober, Burns 2002). Most new construction has been the result of conversion of agricultural to residential use, but increasingly, new areas of desert are being transformed into housing developments.

Clearly, unlimited population growth rates are unsustainable due to accompanying environmental impacts and resource limitation. However, analysts predict the population of Phoenix to level off around 7 million (Gammage 2003). Are there enough resources to support a population of this size without incurring serious environmental damage, impairing resources for future generations? Does the rate of growth affect long-term sustainability? Questions and concerns about the sustainability of Phoenix's rapid expansion are inextricably linked to sociological and ecological processes across many scales, from daily individual and household decisions to long-term climatic patterns and change. As Gammage (2003) notes, "because water's absence is the defining characteristic of a desert, its management becomes the defining activity of living in the desert." In addition to describing current socio-ecological conditions in the metropolitan area, in this article we also address future prospects for maintenance and growth of urban Phoenix, given water as a limiting resource. Phoenix is not alone in addressing these questions; Fitzhugh and Richter (2004) estimate that "41% of the world's population lives in river basins where the per capita water supply is so low that disruptive shortages could occur frequently." Evaluation of Phoenix's sustainability, and implementation of steps to achieve it, will benefit not just Phoenix and the US Southwest but also rapidly growing cities throughout the world.

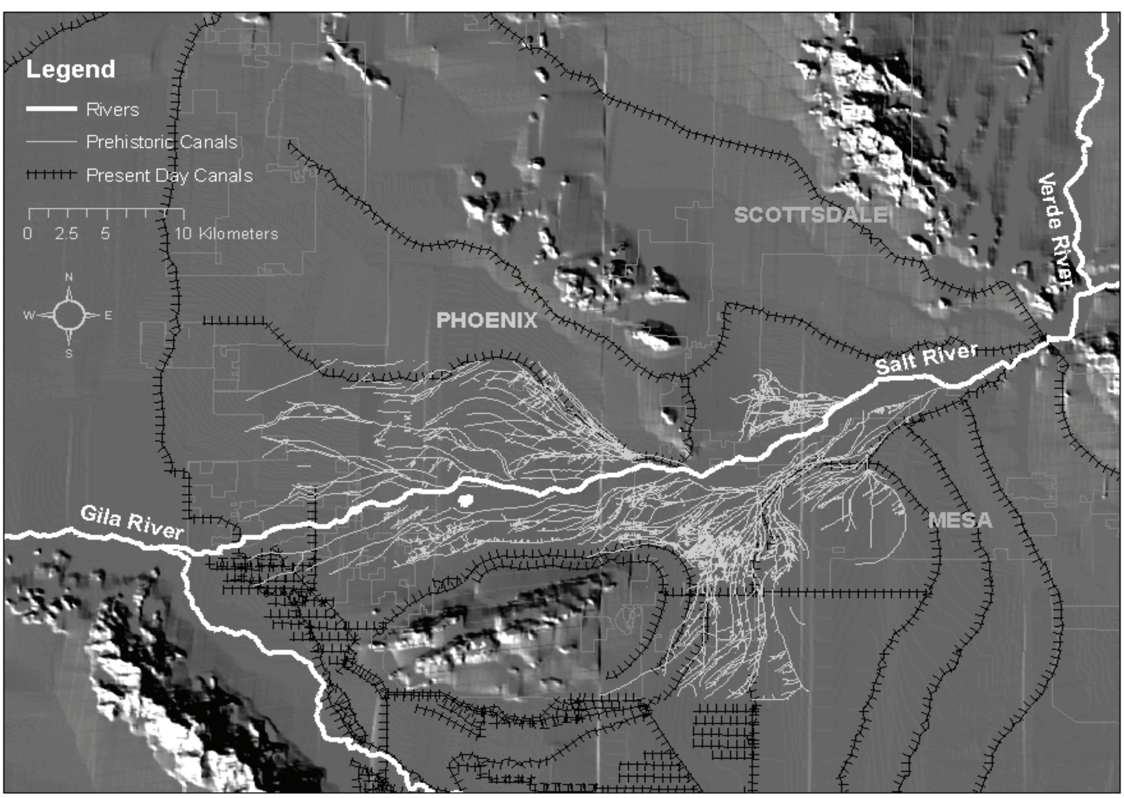

Fig. 1. Major water features in the Phoenix Metropolitan Area, including prehistoric Hohokam Canals. The Salt and Verde Rivers are tributaries to the Gila River, which in turn feeds into the Colorado River (hundreds of km away).

2. Current water resources and management

Phoenix currently has a relatively diverse array of water resources available. The valley has access to 2.8 billion cubic meters of water from its watershed, groundwater, and the Colorado River. The Hohokam were the first in the area to actively manage water resources to enhance agriculture. They built a system of canals on the north and south sides of the Salt River, just downstream from the confluence with the Verde River (see Fig. 1). When Phoenix was re-established as a farming community in the late 1800s to support area mining and military outposts, the ruins of Hohokam canals were discovered, excavated, and expanded upon (Gober 2006).

The magnitude of earlier efforts is small in comparison to the billions of m^3 moved throughout the modern-day system. The transition from relatively small-scale farming to bustling metropolis, although rapid, did not occur instantaneously. Rather, changes were incremental and in response to rising pressures and opportunities. The first step was the construction of dams. Like other rivers of the American Southwest (Baker 1977), the flows of Salt and Verde Rivers are quite variable and flash flooding can have deleterious effects on settlements and agriculture. Damming the rivers reduces flooding and ensures a more stable water supply year-round. In the early 20th century, several dams were constructed along the Salt and Verde Rivers, most of them funded by the US Bureau of Reclamation (Fig. 2). There are now seven dams with six reservoirs (one dam is for diversion only) with a total storage capacity of 4.4 billion m^3 (ADWR 1999). The eventual dam and canal system, managed by what came to be known the Salt River Project (SRP), was able to supply water to greater than 800 km^2 of irrigated farmland (Gammage 2003). As the urban population of Phoenix grew, farms and ranches were converted to residential and commercial areas that retained the prior water rights. These land-use types use less water than agriculture, depending on landscape choice and household conservation. Today, SRP delivers more than 1.2 billion m^3 per year to its service area (Jacobs, Holway 2004).

However, substantial development has occurred outside of the SRP service area, where users must find an alternate source of water. The Phoenix valley has several groundwater sub-basins that are hundreds of meters deep and have been used to supplement surface water supply since the early 1900s. The history of intensive agriculture throughout much of the area has diminished the quality of this groundwater, especially with respect to pesticides and nitrate, a component of fertilizer (ADWR 1999). For example, the median concentration of nitrate is >10.0 mg NO_3^--N dm^{-3} (Baker *et al.* 2004), just above the maximum limit for drinking water established by the US Environmental Protection Agency. Treatment to remove contaminants during the water-treatment process is generally considered cost-prohibitive except when no other water source is available.

Another substantial difficulty resulting from groundwater use is overdraft of the aquifer (removals > recharge). Declining water table levels have been occurring in some places since the 1940s, although early legislation (the 1948 Critical Area Groundwater Code) proved insufficient to slow the trend of increased well drilling. Arizona did not undertake serious measures to curb groundwater use until the federal government issued an ultimatum to the state in 1977: the US Secretary of the Interior threatened to eliminate funds for the Central Arizona Project (CAP), a canal being constructed to deliver water from the Colorado River in the west, eastward across the state some 450+ km and uphill more than 700 m, to the cities of Phoenix and Tucson. In response, the state created the Groundwater Management Act (GMA) in 1980, a com-

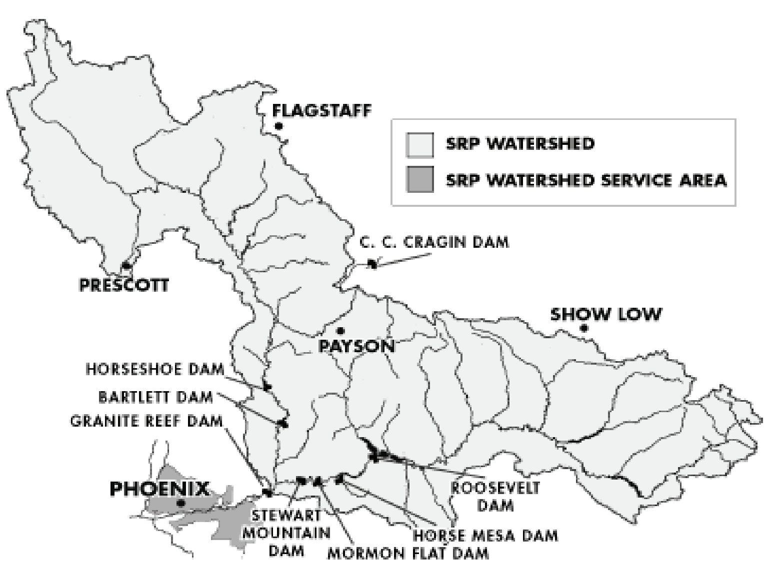

Fig. 2. Salt and Verde River Watersheds serving the Phoenix Metropolitan area. The Salt River Project (SRP) is a quasi-municipal agency that manages water deliveries from these rivers to its service area, as well as providing electricity generated by hydropower. Source: http://www.srpnet.com/water/dams/default.aspx

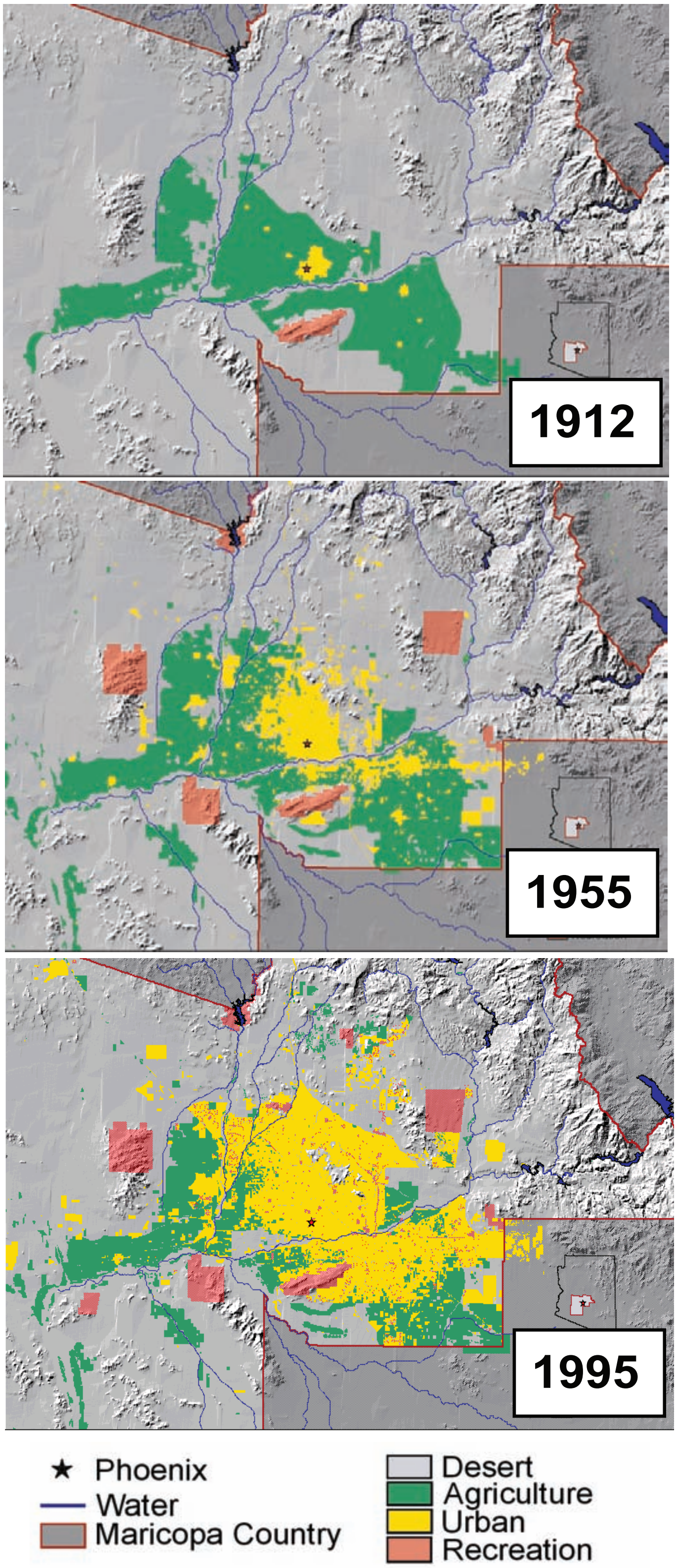

Fig. 3. Changing Land Use in Phoenix's History (After Knowles-Yanez *et al.* 1999 modified).

plex and ambitious regulatory plan to achieve "safe-yield" by 2025. As an "Active Management Area" (AMA), the Phoenix metropolitan area is subject to several management approaches, including supply- and demand-side management, as well as technical planning and assistance (Jacobs, Holway 2004). To date the GMA has had mixed success across the state. In the Phoenix AMA, the groundwater overdraft was reduced by approximately 40% between 1985 and 1995 (ADWR 1999), but 0.44 billion m^3 per year are still overdrafted today (Baker *et al.* 2004). While there have been significant reductions in the Phoenix AMA, groundwater use in the entire state of Arizona has been reduced by only 15% from 1950 to 2000 (Konieczki, Heilman 2004).

The transition from reliance on groundwater to utilizing other sources has been aided significantly by the construction of the CAP canal. In the 1960s, states in the lower Colorado River Basin, California, Nevada, and Arizona began discussing the prospect of apportioning the river water between them. With an allotment of 3.4 billion m^3, Arizona received federal funding to build the CAP canal, designed to bring 1.7 billion m^3 to Phoenix and Tucson. The canal began deliveries of Colorado River water to Phoenix in 1985, but for the first decade Arizona appropriated less than half of its share of water. Initially intended to support agriculture, CAP water instead increasingly went to municipal and industrial users as the urban centers of Phoenix and Tucson grew and agriculture declined. Arizona only recently began to use its full share, largely due to the establishment of the Central Arizona Groundwater Replenishment District (CAGRD) in 1993 and the Arizona Water Banking Authority (AWBA) in 1996. These programs assume that water in the seven underground basins is interchangeable, and that recharge in one place compensates for pumping in other places. Under these plans, developers without access to renewable water sources (i.e., surface water) are able to pump groundwater at the development site in exchange for purchasing an equivalent amount of CAP water to be recharged at an existing recharge facility. This water is legally considered to be surface rather than groundwater. Thus urban growth can occur at sites that otherwise would not have been able to satisfy the 100-year assured water supply criterion (Jacobs, Holway 2004).

The above description of water sources to the Phoenix area is merely a broad overview; the intricacies are extremely complex and often opaque. Although the ADWR is charged with enforcing state regulations and has a general, regional perspective on the Phoenix AMA that includes surface water sources, its primary focus is on groundwater. More than 20 municipalities and agencies in the Phoenix metro in fact make the practical management decisions; there are no standardized methods for accounting and no integrated management approaches that consider all water sources. The summary statistics for the Phoenix area for 1995 are that water sources comprised approximately 44% surface water (Salt and Verde Rivers), 39% groundwater, 12% CAP water, and 5% treated wastewater effluent (ADWR 1999).

The end users of this water supply have changed substantially over time. In the first half of the 20th century most of the water was used for irrigation of cropland. After World War II and the invention of air-conditioning, the urban population of Phoenix began to grow rapidly (Gober, 2006). The proportion of agricultural land use relative to the total area has significantly dropped over the past forty years (Fig. 3). But agricultural water demands are so high in comparison to municipal needs that even with the overall regional decline in agr

agricultural water still represented 58% of the total water demand in 1995 (ADWR 1999), and declined to 42% in 2000 (Authors' calculations based on data from the Arizona Department of Water Resources 2005). The average liters per capita per day (LPCD) for municipal use in Phoenix has only decreased somewhat since 1980 (Fig. 4), and is still above the national average of 693 LPCD. Seventy percent of the municipal use is used for landscape irrigation (Baker *et al.* 2004), indicating not only a potential area for considerable increases in water use efficiency, but also the remarkable ecological transformation that has accompanied urban development in Phoenix. These include introduction of numerous non-native species, destruction of desert habitat and riparian areas, and construction of artificial lakes and stormwater management structures.

3. Current aquatic habitats/ecosystem services

Cultural preferences, along with the relatively easy access to a variety of water resources, have drastically changed the ecology of the Phoenix valley. Demand for agriculture and later municipal uses have had a significant impact on contributing watersheds and downstream systems. Dams on all of the major tributary rivers to the Gila River have eliminated pre-dam seasonal patterns of in-stream

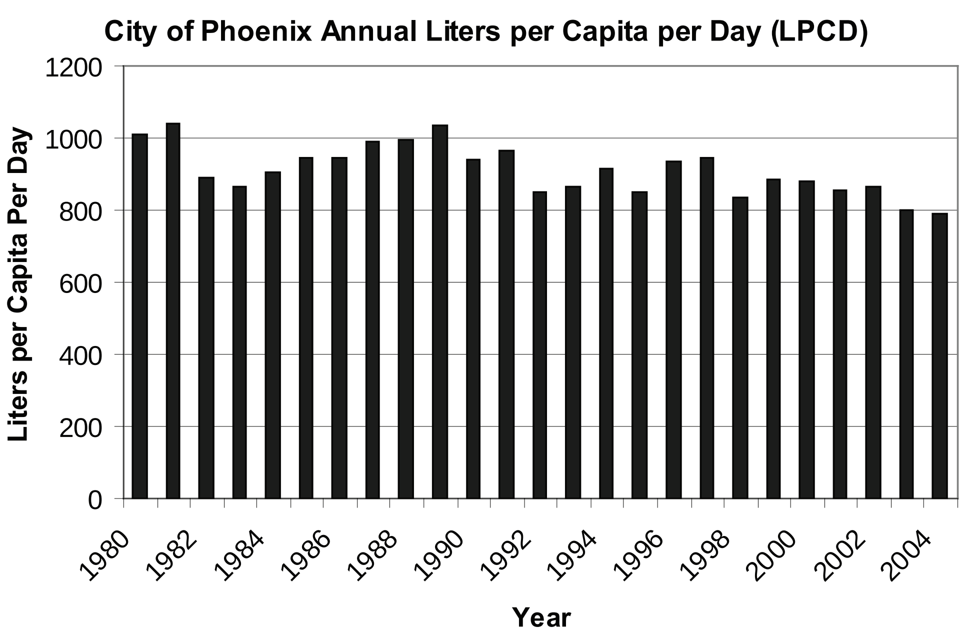

Fig. 4. Annual Averages of daily per Capita Consumption [Units: $dm^3\ capita^{-1}\ day^{-1}$] 1980 - 2004. Source: City of Phoenix Water Services Department

hard-engineering solutions including the lining of riverways. These days, a significant amount stormwater runoff is diverted to stormwater retention and detention basins associated with housing and commercial developments. These basins serve several roles, providing flood mitigation, groundwater recharge, and recreational areas. The built environment has eliminated many natural flowpaths, and although newer developments are designed to handle floods of a particular magnitude, flooding does still occur, especially in older neighborhoods.

Historical modifications have resulted in an overall loss of riparian areas in some places, and a general shift in riparian community composition via bank stabilization, the introduction of non-native plant species, and the decrease in total woody plant volume (Green, Baker 2003). Some river and riparian habitats exist downstream of wastewater treatment plants, in ephemeral river washes receiving stormwater runoff, and sites designated for groundwater recharge, but it is only recently that agencies have begun to consider ecological factors in management of aquatic systems in the Phoenix area. For example, the Rio Salado Project, funded by the City of Phoenix, the Flood Control District of Maricopa County, the Arizona Water Protection Fund, and the US Army Corps of Engineers (US ACE) began the "restoration" in 2001 of 240 hectares of riverbed and riparian areas in central Phoenix. They are using native riparian species, such as cottonwood (*Populus fremontii*), willow (*Baccharis salicifolia*) and mesquite (*Prosopis* spp.), and when completed, the riparian system will include 57 hectares of mesquite bosque and 17 hectares of cottonwood/willow habitats, as well as 16 acres of wetland marsh. However, because the river flow regime has not been restored and groundwater levels have been lowered, there is not enough water naturally available to support these communities. Therefore, the project will include groundwater pumps, canals, and reservoirs to ensure adequate supply (City of Phoenix, 2005).

Meanwhile, further downstream at the 91st Avenue Wastewater Treatment Plant (WWTP), billions of liters of treated effluent are released into the Salt River annually. This nutrient-rich water supports an extensive riparian area, but little groundwater recharge is occurring because the area already has high groundwater levels. From the management perspective, this water is going to waste: a Bureau of Reclamation officer says "we just can't keep dumping it in the stream and letting it go downstream." So an $80 million project is in the works to pipe the water northwest and uphill to the dry Agua Fria riverbed to facilitate groundwater recharge. Officials note that, in addition to recharging the aquifer, the addition of this water will help restore native riparian habitat along the Agua Fria (Landers 2004). However, no mention is made of the potential impact of water removal on the riparian communities of the Salt, which has

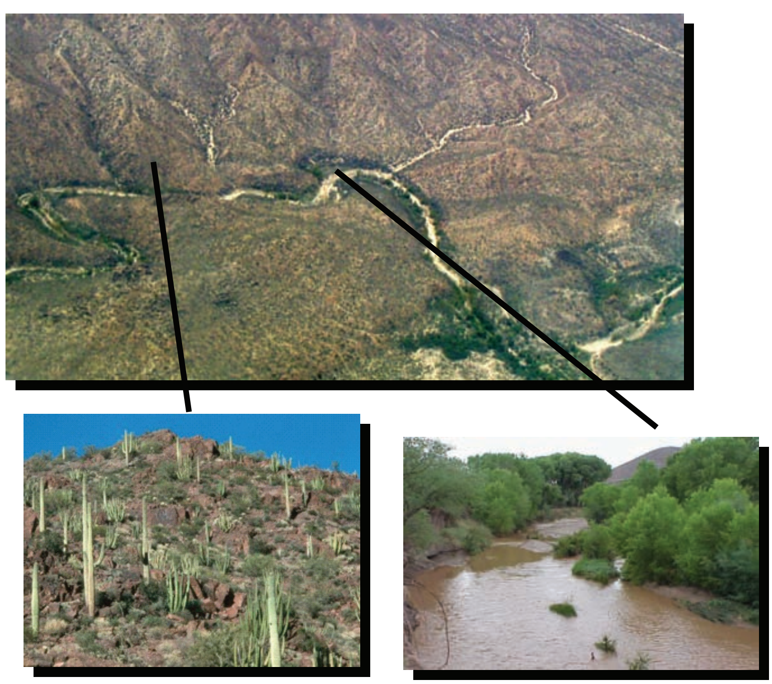

Fig. 5. Impacts of Urban Development on Riparian Habitat. Pictures of Sycamore Creek, NE of Phoenix. Natural Sonoran desert riparian areas have sinuous streams with gallery forests comprised primarily of cottonwood (*Populus fremontii*), willow (*Baccharis salicifolia*) and mesquite (*Prosopis* spp.). Upland areas typically have Saguaro cacti (*Carnegiea gigantean*) and creosote bushes (*Larrea tridentate*).

been receiving effluent from the WWTP for decades.

Thus, inevitably there are complex tradeoffs within the urban ecosystem between various water management and environmental objectives (Grimm *et al.* 2004). The culmination of historical decisions leads more and more frequently to the creation of "designed ecosystems" to satisfy particular goals: water recharge, habitat restoration, aesthetics, recreation. For example, the Phoenix metropolitan area now has greater than 650 artificial lakes (E.K. Larson, unpubl.). In Gilbert there is a Riparian Institute and "water ranch": 18 recharge ponds for treated effluent and constructed "riparian" habitat (the area is not an historical wash or river) designed to attract birds and other wildlife (Edwards 2001). The City of Scottsdale along with US ACE and Flood Control District of Maricopa Country, instead of installing a concrete-lined channel in Indian Bend Wash for flood mitigation, created a series of lakes connected by streams, surrounded by a grassy floodplain including parks and golf courses (Fig. 5 and Fig. 6). All of these require substantial management efforts: water replenishment, algal control, fertilization for maintenance of grass. Current management goals focus on desire for lush, green playing fields, and clear ponds, rather than long-term sustainability.

Very few locations throughout the urban area are typical desert aquatic habitats. Water, as a premium commodity, is moved miles to create desirable landscapes. What is desired is strongly influenced by the cultural backgrounds and experiences of stakeholders. Immigrants from more temperate climates, especially the US Midwest, represent a large proportion of Phoenix residents. With them they may bring memories of lush grasses, abundant vegetation, small ponds and lakes. Two aspects of city life reinforce the perception that such landscapes are sustainable. First, regulation of water flows via damming and reservoirs has damped the strong "pulse" regime of desert hydrology, allowing available water to support vegetation year-round. Second, before the invention of air-conditioning, the cooling effect of increased evapotranspiration was an essential way to contend with extreme summertime temperatures. Thus, throughout Phoenix's history, new arrivals have seen a steady supply of water and abundant green growth, and many were attracted in the first place by promotions of the city as an idyllic place to retire,

Fig. 6. Impacts of Urban Development on Riparian Habitat. Pictures of Indian Bend Wash, Scottsdale, AZ. Urban washes have been radically transformed. No buildings are allowed within the 100-year floodplain, instead a series of artificial lakes, streams, and wide grassy park areas provide flood abatement. Note the severe down-cutting in the stream.

famous for its golf courses. There is no reason, given that such perceptions are actively encouraged, for newcomers to think of water as a limiting resource. Essentially, many residents no longer perceive or appreciate that they are living in the desert; for them, the desert exists only outside of the city (Farley-Metzger, personal communication; Gober 2006).

Another effect of urbanization on the ecosystem is the "urban heat island." In Phoenix, the average minimum temperature has increased on average about 0.1º C per year over the past fifty years, due to nighttime attenuation of cooling by the reradiating built structures (Baker *et al.* 2002; Brazel *et al.* 2000). Additionally, the number of "misery hours per day" (hours in which the temperature is above 38º C) has doubled since 1948 (Baker *et al.* 2002). While the only direct effects of the heat island on aquatic ecosystems are increased evapotranspiration rates and plant stress within the city, the overall demand in the city for water increases as cooling and irrigation needs rise, possibly resulting in less water availability in contributing watersheds.

4. Sustainability concerns, challenges for the future

Are the water supplies and management practices for the Phoenix metropolitan area sustainable? The crux of this question is that the answer depends on the interaction of a multitude of uncertain ecological, economic, social, and cultural variables. Assessment of these variables is ongoing, but is hampered by significant technological, organizational and informational difficulties. At the most basic level, research is still needed on human population trajectories, ecological impacts, climatic change, etc. There is a paucity of data on environmental outcomes of urban development, water use, landscaping practices, etc. at scales ranging from individual households to municipalities to watersheds to the Colorado River basin. Even well established and government-supported programs, such as the Central Arizona Groundwater Replenishment District, are based on assumptions and simplifications of the complex system that have had insufficient investigation. For example, there is no science to support the feasibility recharged surface water in one location to compensate for groundwater pumping in another, much less any investigation into resultant water quality. Given the groundwater overdraft, subsidence, and lowered water levels, does recharging work on a basin scale? Does recharged water remain perched? Is it reasonable for housing developments to recharge surface water at one location in the basin, and expect the water they pump at their location to be "surface water" (or at least counted that way)? What are the long-term impacts of the spatial and temporal discontinuities of groundwater banking? The Phoenix Active Management Area of the Arizona Department of Water Resources, along with other research and public interest groups, are striving to address these issues with extensive data collection and modeling. After extensive research, the Governor's Water Management Commission (2001) conceded that, given the continued and projected trend of groundwater overdrafting, it is unlikely that the Phoenix AMA will achieve safe yield by the 2025 deadline. Even if it were possible to reach the goal of safe yield, Jacobs and Holway (2004) note that "the safe-yield goal… does not account for potentially diminished surface water flows or localized areas of depletion. Thus safe-yield is not necessarily synonymous with sustainability, as defined by the Brundtland Commission…"

With respect to renewable (non-groundwater) sources, new analyses continue to emerge. Some authors, such as Gammage (2003), argue that populations as high as 7 million will be sustainable as long as there is a corresponding decrease in agriculture (a water-intensive land use). His view does not incorporate any climatic variability. Morehouse *et al.* (2002) conclude that the variety of water resources available to Phoenix provide more of a buffer to short-term drought conditions than Tucson, but that "even if agricultural demand were eliminated entirely, drought conditions would still force the AMA to rely on non-renewable supplies to meet 43% of its needs." However, one of the most basic underlying assumptions about the flexibility of Phoenix water resources, that Colorado River water will provide when the Salt and Verde are experiencing drought and vice versa, was recently challenged by a joint University of Arizona/Salt River Project report. The report used tree-ring analysis to reconstruct drought cycle synchrony between the two basins, and found that only two events in the 443 years analyzed showed asynchronous flow (Hirschboeck, Meko 2005), leading the manager of water resources at SRP to opine, "our thought that the Colorado River would be able to bail us out is not a safe assumption anymore" (McKinnon 2005c).

The current political environment is also changing. The 2004 Arizona Water Settlements Act returns 800 million m^3 to the Gila River Indian Community (GRIC), to compensate for lost access to surface water by appropriation by European settlers in the 19th and 20th centuries. This water will be allotted from CAP water, reducing the amount available to the cities of Phoenix and Tucson. The GRIC will lease 49 million m^3 back to the municipalities and 82 million m^3, previously undistributed, will also be allotted (2004). The tribe will also have the option to lease a greater portion of their water back to the municipalities, but it is unknown if they will elect to do so, many expect they will use the water for their own agricultural needs (King 2005). The final impact of the Act has yet to be fully realized.

Additionally, the legal status of Arizona's claim to CAP water is not secure. The US Secretary of the Interior has instructed the Colorado Basin states (Colorado, Utah, New Mexico, Wyoming, Nevada, California, and Arizona) to come up with a drought and water-shortage management plan (McKinnon 2005b). Arizona, as the most junior party, could lose some of its allotment more easily than more senior states in times of crisis, and thus has started a legal defense fund in anticipation of upcoming disputes. Such clashes might not be far on the horizon, as Upper Basin states protest Lower Basin states' use of water in Colorado River tributaries that does not count in their total allotment. If the US court system were to decide that tributary flows should be included, Arizona could lose up to half of its CAP allocation (McKinnon 2005a).

Finally, accurate and precise assessment of water supply and demand for the valley remains elusive. With more than 20 municipalities and agencies making management decisions, there is an acute lack of consistency in the way that water use is calculated. On top of that, the history of western US water-rights law makes stakeholders reluctant to disclose all information, for fear that other agencies will dispute claims and annex resources. There is no good, integrative, regional understanding of actual usage; for example, in a report published by the City of Phoenix Water Services Department (1995) designed to inform city residents of future prospects, there is only tangential mention that, if the city were to assert its total allotment rights during a time of drought, other municipalities would likely suffer shortages. Such gaps in communication lead to conclusions such as Bush's (2005) that there is enough renewable water only "if the context of institutional arrangements and water entitlement is ignored." As the city is in the midst of an ongoing decade-long drought, coupled with continued rapid growth, the pressure for institutional transformation is increasing. Efforts with varying foci and scales have been initiated by several research institutes, municipal agencies, and collaborative organizations. For example, the East Valley Water Forum is a partnership of tribal, public, and private water agencies working together to assess the status of current water resources and develop plans for meeting future water needs reliably. Efforts such as these are nascent; considerable work is still needed.

Conclusions

Beyond the traditional boundaries of basic natural science, urban ecological questions pose new challenges for researchers, as they necessitate interdisciplinary work between the natural and social sciences (Grimm *et al.* 2000). Anthropological and sociological questions about what makes Phoenix Phoenix are intimately tied to the changing environmental setting. For instance, research has shown that plant diversity within the city is closely correlated with socioeconomic factors such as family income and housing age (Hope *et al.* 2003). But socio-cultural values, like ecosystems, are mutable. For the Phoenix metropolitan area, water is perhaps the foremost integrator of these issues. Adequate assessment of regional sustainability and the means to achieve it require comprehending how values, economics, and the environment feedback to one another and change over time. The Central Arizona - Phoenix Long Term Ecological Research project (CAP LTER), a nationally funded program now in its eighth year, seeks to expand and develop the necessary research tools and data to understand the long term, regional dynamics of the urban ecosystem (Grimm, Redman 2004). Key areas of research include Land Use/Land Cover Change; Climate-Ecosystem Interactions; Fluxes of Materials and Socio-Ecosystem Response; Human Control of Biodiversity; and Water Policy, Use and Supply. Additional vital insight will be provided by the newly funded Decision Center for a Desert City (DCDC), a research institute at Arizona State University focused on establishing relationships between climatic conditions and water decision making.

Phoenix was named explicitly after the mythological bird that rose again from its ashes. With its astronomical continuing growth, Phoenix again burns bright, but will it maintain enough water to prevent another incineration?

Acknowledgments

This paper summarizes background information on Phoenix water from work on the Central Arizona-Phoenix Long-Term Ecological Research (LTER) project and the Decision Center for a Desert City project, both funded by the National Science Foundation (award #s DEB-0423704 and SES-0345945, respectively); EKL is also supported by National Science Foundation award # DGE-9987612; support is gratefully acknowledged.

Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author and do not necessarily reflect the views of the National Science Foundation.

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