Effects of Urbanization on Nutrient Biogeochemistry of Aridland Streams

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Land-use and land-cover change affect the biogeochemistry of stream ecosystems in numerous ways, both direct and indirect. Changes result from hydrologic modifications, including direct alterations of flow regimes and hydrologic flowpaths and indirect changes in hydrologic patterns via increased impervious cover in contributing areas of watersheds. Direct changes to channel morphology (i.e., reduced complexity) and to floodplains of streams and rivers also influence biogeochemistry, for example, by eliminating surface water–groundwater exchange. The nature of and strength of connections between the stream and its watershed may be altered by large-scale changes such as those brought about by urban and suburban development. Finally, in urban and agricultural areas, elevated nutrient loading is exacerbated by land-cover changes that increase the potential for erosion or overland flow, and decrease the opportunities for nutrient retention. Using comparative analysis based on published work, mined public data, and new research, we evaluate urbanization effects on stream ecosystems in the Sonoran Desert region of central Arizona. Five key characteristics of non-urban desert streams—nitrogen limitation, a flashy hydrologic regime that initiates succession, extensive groundwater–surface water interaction, episodic terrestrial–aquatic interactions, and high capacity for nutrient retention—are both dramatically altered and scarcely affected by urbanization. The similarities exhibited by aridland streams and their urban counterparts arise from large-scale constraints (e.g., episodic terrestrial–aquatic interaction is imposed by the climatic regime in both stream types), whereas the differences, like interrupted flowpath continuity in the urban landscape, likely result from the myriad direct modifications of streams and catchment land cover in cities.

INTRODUCTION

Controls of ecosystem functions operate at several scales. Constraints are those controls that occur at scales greater than those of the relevant processes, while mechanisms are controls that explain pattern based on smaller-scale processes. To understand what influences the responses exhibited by any ecosystem to land-use and land-cover change, we must

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consider four factors (Figure 1): (1) the system domain (i.e., the context in which the change is taking place, taking into account the constraints imposed by geomorphic and geologic setting, large-scale socio-political constraints, etc.); (2) the nature of the change; (3) the nature of the ecosystem being changed (i.e., intrinsic ecosystem features); and (4) the feedbacks that occur between the socio-ecological system and the agents of change.

Land-use changes that affect biogeochemical cycles occur from global to local scales. Even if we consider just one type of land-use change, urbanization, the scales that we must take into account can be quite broad because the footprint of urbanization extends far beyond the boundary of an urbanized area itself [*Folke et al.,* 1997; *Luck et al.*, 2001]. Take, for example, the ability of surrounding ecosystems to absorb the excess CO_2 produced by an urban area. The efficiency of this ecosystem service depends on intrinsic ecosystem features, like successional stage, of the ecosystem being changed. However, it also is a function of the nature of the change (how much CO_2 is produced, the number of urban inhabitants and their driving patterns, the magnitude of industrial CO_2 sources), as well as the context (domain) in which the problem is embedded (the wealth, as it affects auto ownership, of the population, available technology to reduce emissions, and air-flow patterns, which are driven by climate and topography). Finally, feedbacks that potentially alter both further urbanization and the CO_2 problem include heat island impacts on quality of life [*Baker et al.,* 2002], which could be positive (e.g., increased use of air conditioners) or negative (reduced immigration to the urban area), and institutional responses, such as stricter controls on emissions. Moreover, these feedbacks could occur at multiple scales (local, statewide controls on emissions; national air-quality regulations).

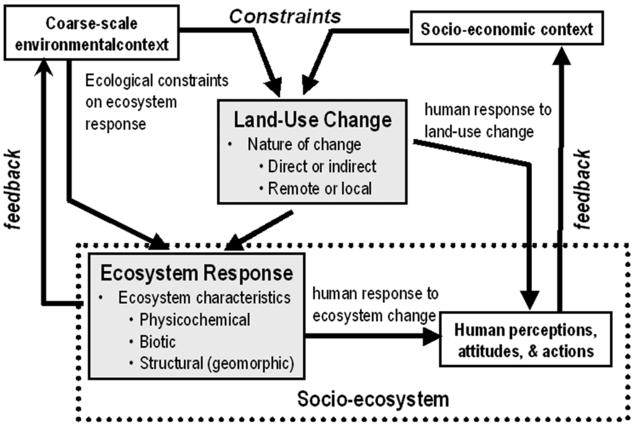

Figure 1. Ecosystem responses to land-use change occur in a domain of environmental and socio-economic contexts, which set constraints on the nature of land-use change possible [figure modified from *Grimm et al*., 2000]. Ecosystem response also is subject to these constraints, and depends on the nature of the change and the inherent ecosystem characteristics. Humans, part of the larger socio-ecosystem, respond to changes in land use and ecosystems, and their actions feed back to the socio-economic context.

Stream ecosystems are integrators of the catchments that they drain; thus, we can think of ecosystem response to land-use change at the scale of a stream reach or at the scale of a large catchment or drainage network. Several categories of land-use changes directly affect the reach or operate at the scale of the entire catchment. Feedbacks also may occur at multiple scales. In the U.S. Pacific Northwest, migrating salmon transfer marine-derived nutrients to their natal streams, where the nutrients can stimulate riparian and stream processes [*Richey et al.,* 1975; *Helfield and Naiman,* 2001]. Management choices, particularly establishment of dams along migration routes, affect this return of nutrients through impacts on spawning success of previous years. In other words, the land-use change is local (e.g., a single dam on a large river) but its effects are seen over a large spatial scale (a linear migration pathway from sea to small streams) due to the nature of the system being changed (the dendritic character of streams creates a bottleneck for migration to all small streams above an impoundment point), and feedbacks occur across multiple years (variations in spawning success as a function of streamflow regulation). Another example is seen in the fluvial response to urbanization. The geometry of the drainage networks (sewers and streets as water conveyors) is fundamentally changed, but changes in water and sediment transport also result from activities in urban catchments. Increased sediment flux from surfaces exposed during construction initially produces aggradation in downstream reaches, but as construction is completed, impervious, generally regolith-free surfaces (roads, roofs) replace debris piles and graded soils [*Graf, 1975*]. Fluvial conditions change from sediment-rich and runoff-retarded to sediment-starved and runoff-enhanced. Feedback from the change in land use to the geomorphic response thus varies temporally. The biogeochemical and other functional consequences of such geomorphic changes are poorly understood; we begin to consider them in this chapter.

Finally, streams differ depending on their biogeophysical contexts (biomes or ecoregions) [*Webster et al.,* 2003], and thus responses may be context-dependent. Streams of the U.S. desert Southwest share certain characteristics that contrast starkly with streams in forested regions, making the hypothesis reasonable that a given land-use change will have different effects in aridland streams than in forest streams.

In this chapter, we outline some key characteristics of desert streams that influence their biogeochemistry. We then contrast urban desert streams to native1 desert streams to explore the similarities and differences between them, and to ask what aspects of the interaction between land-use change and ecosystem response (i.e., context, nature of change, nature of ecosystem, or feedbacks) explain these differences. Our baseline assumption (null hypothesis) is that the same rules operate in human-dominated ecosystems as in all ecosystems, so if we understand what controls biogeochemical processes in wild ecosystems then we can make predictions of how human-altered ecosystems should respond. For example, hydraulic conductivity is a key parameter that controls the extent of surface water–groundwater (SW–GW) interaction, which in turn has a strong influence on biogeochemical function in streams [*Jones and Mulholland,* 2000]. Impervious surfaces are just that—impervious; or put another way, they have very low hydraulic conductivity. Therefore, we can make some predictions about how SW–GW interaction and hence biogeochemical processes will be altered by increasing impervious surface coverage in an area. When or if these predictions do not hold, of course, we may need to examine other controlling factors (e.g., human decisions and behavior, socio-political constraints), and determine if there is a threshold at which the "same rules" do not apply in human-dominated ecosystems.

LAND-USE CHANGE AND STREAMS

Land-use changes of particular importance to stream ecosystems are both indirect and direct. Indirect changes include land-cover modifications to a stream's catchment, which in turn influence hydrologic regimes, or diffuse loading of materials derived from certain land uses (e.g., agriculture, residential areas). Direct changes are especially important in urbanized regions, and include major modifications of stream morphology, hydrology, and riparian vegetation.

A framework for considering these types of changes, in the case of urban stream ecosystems, would include both one-way and bi-directional interactions among land use, geomorphology and hydrology, vegetation, and biogeochemistry, which collectively influence stream ecosystem function. For example, a direct modification of channel form like channelization alters the residence time of water in the system, which influences the capacity for nutrient retention. Increased impervious surface cover in the catchment feeding streams dramatically elevates flood discharges while lowering base flow because of reduced infiltration to groundwater [*Arnold and Gibbons,* 1996]. A much higher proportion of a stream's nutrient load is thereby discharged to downstream recipient systems, rather than being retained *in situ*. Losses of riparian vegetation may result in reduction of hydraulic roughness, producing higher flow velocity, and loss of bank stability, leading to erosion and elevated sediment transport.

Data and examples used to illustrate how land-use change via urbanization affects stream biogeochemistry through this complex series of interactions and feedbacks are drawn from our experience with the Central Arizona?–Phoenix Long-Term Ecological Research project (CAP LTER). The Phoenix metropolis, comprising >20 municipalities, is situated in a broad, alluvial basin where two major desert tributaries of the Colorado, the Salt and Gila Rivers, converge. The basin, which is dotted with eroded rock outcrops, rimmed by mountains, and crossed by several river channels (Plate 1), once supported a vast expanse of lowland Sonoran Desert and riparian vegetation. The CAP LTER study area includes the rapidly expanding Phoenix area along with surrounding agricultural and desert land. The region's population has increased by 47% since 1990 to over 3.5 million people [*U.S. Census Bureau*, 2000]. Growth and expansion of Phoenix has occurred mostly in the second half of the 20th century.

Since ancient times, humans in the Phoenix basin have modified river flows to access river water for irrigated agriculture. The Hohokam civilization of ca. 500–1400 A.D. developed an extensive network of canals, the channels of many of which were exploited in establishing the modern canal system. However, they were susceptible to flood damage of irrigation structures [e.g., *Ackerly et al*., 1987]. Water delivery and supply remains one of the main motivations for river and stream modification, and throughout much of the history of human occupation of the Phoenix basin agriculture has been the primary water use. On the other hand, desert streams are notorious for their flashiness [*Baker*, 1977; *Grimm and Fisher,* 1989], so flood protection also has provided an impetus for stream modification. Immediately, we see that trade-offs are necessary in balancing these two demands. Farmers and inhabitants of the city want a reliable supply of water but also adequate protection from high-discharge events. These trade-offs are made even more complex by newer motivations for stream modification associated with the rapid urbanization of the region in the past half-century: floodplain development to meet aesthetic demands or economic opportunity, water diversion for designed landscape features like artificial lakes, ponds, and golf courses, and ephemeral channel modification for groundwater recharge projects. Changes in the Salt River itself illustrate some of these conflicts.

Several north–south trending streams once flowed into the Salt–Gila River system in the Phoenix metropolitan area, notably the Verde (upstream from the city), the Agua Fria (near the city center), and the Hassayampa (downstream from the city in a largely agricultural area; see Plate 1). Upstream from the city, the waters of the Salt River are completely diverted into a network of canals that supply drinking water, irrigation, and other municipal water needs to the metropolis; hence, there is no surface flow in the Salt except during floods, and once-extensive riparian forests vanished decades ago. For decades, the riverbed was a dry, desolate wasteland home to

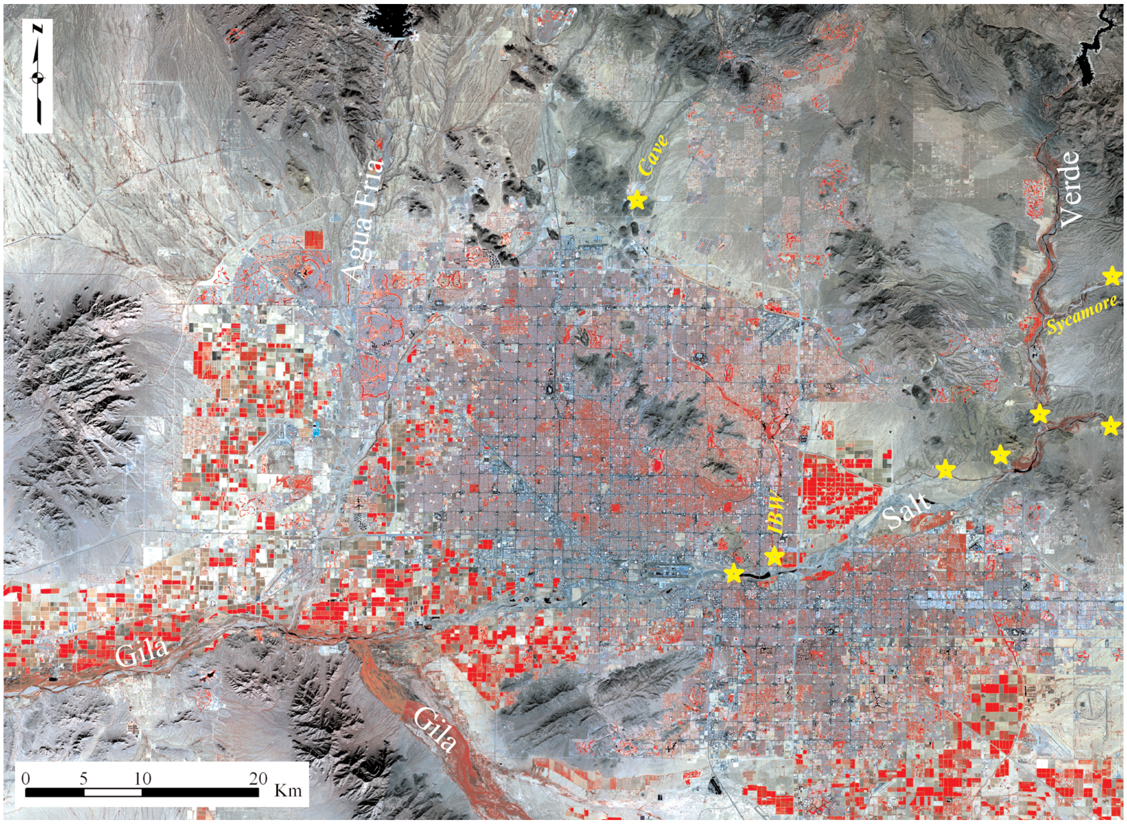

Plate 1. ASTER satellite image mosaic of the Phoenix basin acquired in 2000 and 2001, showing built-up urban cores, fringing agricultural areas, and the network of rivers (white labels). Stars denote stream-sampling sites of the Central Arizona–Phoenix Long-Term Ecological Research project (CAP LTER); small watersheds discussed in this chapter are also indicated, in yellow (IBW, Cave Creek, and Sycamore Creek). In this image, red is ASTER band 3 (reflected IR wavelengths), green is ASTER band 2 (red wavelengths), and blue is ASTER band 1 (green wavelengths). The ground cell resolution is approximately 15 m.

gravel mining operations, bordering landfills, crisscrossing roads, and surrounding industrial and agricultural land uses [*Graf*, 1998; *Stefanov et al.,* 2001]. A recent (1999) channel modification motivated by the City of Tempe's desire to improve the condition and value of its riverfront property was to convert a 4-km length of the dry riverbed to a lake, with the hope of attracting new business with shoreline parks and recreational amenities [*City of Tempe,* 2003]. Tempe Town Lake, although located in the Salt River bed and filled with mostly Salt River water, comes by this water via delivery from the nearby canals.

The decision to build the lake, and to expend city financial resources on the water to fill it, represents a victory for aesthetics (or perhaps economic incentive) in the tradeoff between floodplain development and simple water delivery and supply. The trade-off between floodplain development and flood protection in the Salt River is expressed as technological innovations, such as collapsible and inflatable rubber dams, that allow re-establishment of the lake after recession of floods. Alternate solutions that reflect this trade-off are being played out in many of the Phoenix basin's smaller rivers and streams, as new, non-structural flood management designs are implemented.

FIVE KEY CHARACTERISTICS OF ARIDLAND STREAMS

In our consideration of biogeochemical consequences of river modification, we will concentrate on two smaller watersheds within the Phoenix basin, Cave Creek and Indian Bend Wash, that can be compared with Sycamore Creek, a spatially intermittent desert stream northeast of Phoenix that has been under study since 1976 [*Fisher* 1986; *Grimm*, 1994]. This long-term study has generated a solid understanding of key controls on desert stream biogeochemistry, which provides a basis for our consideration of how urbanization changes those dynamics. Sycamore Creek drains a 500-km^2 catchment of central Arizona, northeast of Phoenix, with elevational ranges from 500 to >2000 m. Stream and watershed research focus on the low desert portions of this ecosystem (see Plates 1 and 2). This and other research has identified five key characteristics of aridland streams: they are nitrogen-limited, they undergo succession in response to frequent disturbance, they exhibit SW–GW interactions that control nutrient dynamics, they are subject to episodic terrestrial–aquatic interactions that result in episodic nutrient import, and they are likely hot spots of nutrient retention in the desert landscape.

Nitrogen Limitation

Nitrogen (N) is the limiting nutrient to aquatic primary productivity in Sycamore Creek and many other southwestern streams [*Grimm and Fisher*, 1986a]. Nitrogen availability varies in both space and time due to the desert stream characteristics described below, but during most baseflow periods, primary production is N-limited. This leads to high rates of N_2 fixation by cyanobacterial communities. No definitive tests of nutrient limitation have been done in southwestern riparian zones, but it is likely that N is the limiting nutrient to these ecosystems as well. In most ecosystems, the nutrients N and phosphorus (P) are potential limiting nutrients because their abundance is most often in short supply relative to organismal demand. Perhaps because the Southwest is a geologically youthful landscape with abundant P-bearing minerals, P appears to be in excess in most stream ecosystems and seldom reaches even the status of secondary limitation [*Grimm and Fisher*, 1986b]. The ecological stoichiometry [*Sterner and Elser*, 2002] of N and P, that is, the relative balance of these two elements, may be modified in urban ecosystems, potentially alleviating nutrient limitation or changing the identity of the limiting nutrient.

Disturbance and Succession

Desert streams like Sycamore Creek experience disturbance by flash flooding and so they are frequently in a successional state [*Fisher et al.*, 1982]. In terms of biogeochemistry, this means that actively growing, in-stream and riparian biota retain nutrients (input exceeds output) during early stages of succession [*Grimm,* 1987]. The largest flash floods remove the biota (and retained nutrients) and export these materials from the watershed to downstream ecosystems; however, many smaller events redistribute biota and other materials within the catchment. Successional rate is high, with preflood standing crops of algae and macroinvertebrates often established within a few weeks [*Grimm and Fisher,* 1989], but N limitation comes into play such that regions experiencing N inputs from groundwater show faster rates of biomass increase than those without such inputs [*Valett et al*., 1994].

Surface Water–Groundwater Interactions

Surface water–groundwater interactions are important to biogeochemical patterns and processes in many streams [*Dahm et al.,* 1998] including desert streams [*Grimm et al.*, 1991; *Valett et al.,* 1994; *Dent and Grimm,* 1999]. These interactions, from surface–hyporheic exchanges at the scale of tens to hundreds of meters to deep groundwater inputs at kilometer scales, explain patterns of variability in inorganic N concentration, which reflects the net result of N uptake and mineralization processes [*Fisher et al.*, 1998; *Dent et al.*, 2001]. The hyporheic zone, where groundwater and surface water interact, extends beneath the active channel and riparian zone and is a site of much longer-term hydrologic and nutrient storage than the surface stream. Furthermore, it is a region where N transformations (mineralization, nitrification, denitrification) balance the dominant process of N uptake in the surface [*Holmes et al*., 1994; *Jones et al.*, 1995a; *Jones et al.,* 1995c; *Holmes et al.*, 1996] . In effect, catabolic processes characterize the hyporheic whereas anabolic processes characterize the surface stream [*Jones et al*., 1995b]. Because groundwater is thus rich in N, the consequences of SW–GW interactions are extreme spatial variability in N supply and concentration at several scales (Figure 2).

Terrestrial–Aquatic Interaction

Combining our knowledge of the role of flood disturbance, N limitation of streams, and surface–groundwater interactions, we can infer that the episodic land–water interactions that characterize deserts may drive the nutrient status of streams [*Grimm and Fisher,* 1992*; Grimm,* 1994*; A. Huth,* University of Arizona, pers. comm.]. Over the long term, nitrate-N concentration (the primary form of inorganic N in streamwater) in Sycamore Creek is positively correlated with discharge; in other words, floods are associated with elevated inorganic N concentration. Because floods recharge subsurface storage reservoirs in the riparian and hyporheic zones [*Marti et al.,* 1997] this nutrient subsidy from the uplands may feed stream productivity long after floods have passed, for example through bank drainage following flood recession [*A. Huth,* University of Arizona, pers. comm.]. In these arid regions, however, hydrologic connections between uplands and stream-riparian systems effectively cease between those runoff events that reach the channel from the uplands.

Nutrient Retention

Finally, streams and their associated riparian zones are likely hot spots of nutrient retention and/or removal in the desert landscape [*Fisher et al*., 2001; *Belnap et al.*, 2004]. A hot

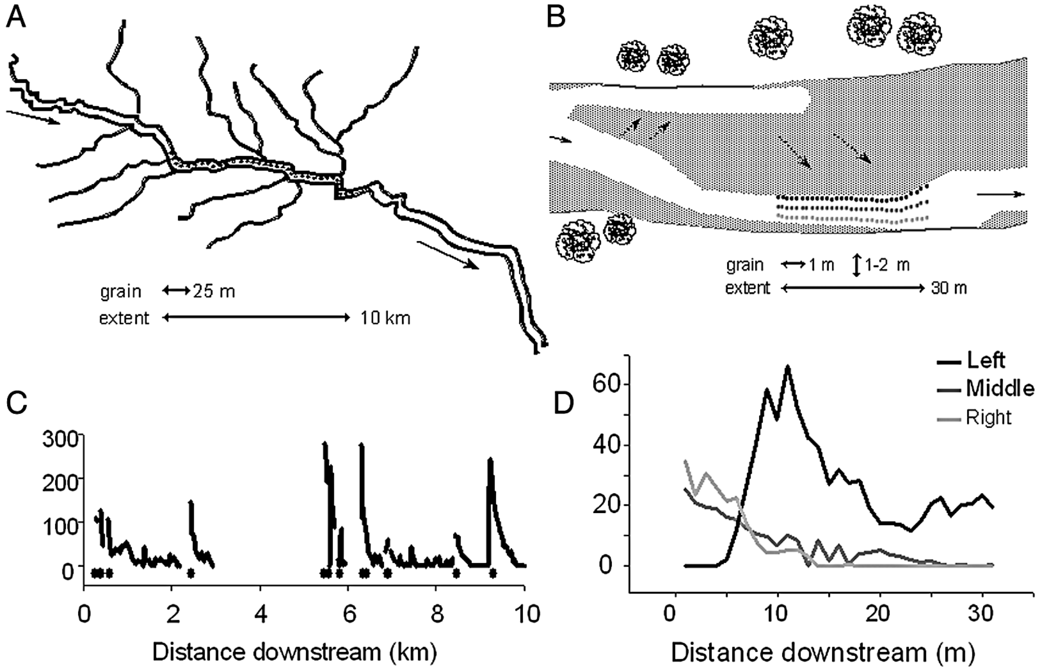

Figure 2. Effects of groundwater or subsurface inputs on nitrate-N concentration at two scales in Sycamore Creek, Arizona. A) Map of the 10-km stream section (extent=10 km) that was sampled every 25 m (grain = 25 m); B) Pattern of nitrate-N concentration (μg/L) along the stream section showing increases at consistent sites of GW discharge (red dots). C) Longitudinal plot of nitrate-N concentration (μg/L) showing enrichment on the side where subsurface water enters from the gravel bar, and D) schematic diagram of a 30-m reach (extent = 30 m) that was sampled along the left margin, middle, and right margin of the stream (grain = 1–2 m). Adapted from *Dent and Grimm* [1999; A, B] and *Dent et al.* [2001; C, D].

spot is a localized area where biogeochemical reaction rates are much higher than the surrounding matrix, and often forms where reactants and products are brought together through convergence of flow paths [*McClain et al.,* 2003]. Biogeochemical reaction rates are very high in riparian bank soils [*Jones et al.* 1995c; *Holmes et al.*, 1996], particularly at the interface between riparian zone and gravel bar, or between open and plant-colonized gravel bars [*Schade et al.*, 2001]. A role for aridland riparian ecosystems in N removal at the watershed scale has yet to be documented, however, since the total flux of material through these interfaces is unknown.

ARE URBAN DESERT STREAMS SIMILAR OR DIFFERENT IN THESE FIVE CHARACTERISTICS?

The CAP LTER study site lies at the confluence of the Salt and the Gila Rivers in Arizona's Basin and Range Lowlands Hydrogeologic Province [*Montgomery and Harshbarger,* 1989; *Anderson*, 1995]. The province is characterized by large, gently sloping basins filled with alluvial material eroded from the surrounding mountains. The mountains are composed of igneous, metamorphic and sedimentary rocks, ranging in age from Precambrian to Cenozoic [*Cooper and Cone*, 1969]. The geologic history of the region sets the stage for human occupation and modification of the environment. Geologic processes are responsible for the particular combination of landscape features that have made the Phoenix basin a desirable setting for human occupation. Periods of extensive aggradation in the early and mid Pleistocene (~1,000,000 to ~100,000 years ago), followed by slight downcutting and entrenchment in the late Pleistocene and early Holocene (~20,000 to ~10,000 years ago), created the wide expanses of the basin floor and stabilized the drainage system within well-established channels. Both prehistoric and modern populations have used these surfaces and drainage systems for irrigated agriculture. The growth of the greater Phoenix regional metropolis was promoted and sustained by the irrigation potential of the basin.

The Cave Creek watershed encompasses more than 90 km^2 including the towns of Cave Creek and portions of north Phoenix (see Plates 2 and 3). The main ephemeral channel runs for more than 50 km with the lower 30 km flowing through urbanized or rapidly urbanizing areas. The pre-development stream channel ran through what is now downtown Phoenix, but all flood flows are now diverted upstream from the urban core. Slopes in this drainage are typically on the order of 1%

to 5%; however, below the town of Cave Creek the average stream gradient is even less (see Plate 3). In these lower reaches, the main channel cuts through a sequence of river terrace deposits that vary in age from middle Pleistocene (~2 Ma) to the present, but flow is ephemeral. The terraces suggest perennial flow and frequent channel migration prevailed during the Holocene. Dams help to reduce the chance of frequent damaging floods, but also can create a false sense of security in downstream areas. One such dam was constructed in 1922 to prevent flooding in west-central Phoenix. During the high precipitation years of the late 1970s, this dam was at full capacity repeatedly, prompting construction of the Cave Buttes Dam less than 1 km downstream from the original dam.

Indian Bend Wash (IBW) is a tributary to the Salt River and drains approximately 584 km^2 of desert, agricultural and suburban lands as it flows from the McDowell Mountains northeast of Scottsdale, Arizona, to its confluence with the Salt River (see Plates 1 and 2). The mainstem of the wash runs north–south through downtown Scottsdale (Plate 1). Historically, it was dry most of the year but prone to heavy flooding during storms. To protect the city from floods and provide recreational opportunities for its populace, the Army Corps of Engineers and the City of Scottsdale built the IBW flood control project [*City of Scottsdale*, 1985]. The floodway was designed to accommodate a 100-year storm discharge of 850 m^3/s [*U.S. Army Corps of Engineers,* 1975]. In the lower portion of the wash, a greenbelt was constructed, the most striking feature of which is a series of several shallow permanent lakes that sit in the larger, protected floodplain. When it is not flooding, lake levels are artificially maintained with a mix of canal and groundwater.

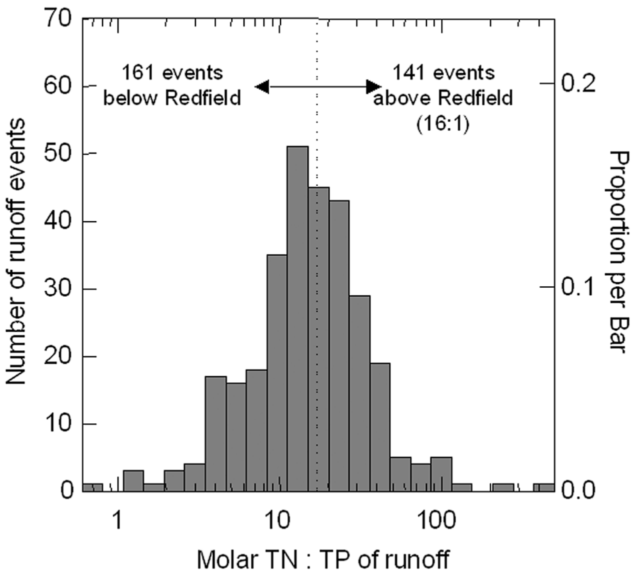

Figure 3. Frequency distribution of atomic nitrogen to phosphorus ratios (N:P), based on total N and total P concentrations in stormwater runoff from urban catchments of metropolitan Phoenix, Arizona. Vertical line indicates the Redfield Ratio, the N:P above which the potentially limiting nutrient theoretically switches from nitrogen to phosphorus.

Nutrient Limitation

The relative balance of N and P is likely to be strongly modified in Phoenix's urban waterways as a result of the massive imports of N to the ecosystem in the form of food, fertilizer, and fuel [*Baker et al.*, 2001] and heavy utilization of groundwater, which is N-enriched [*Lauver and Baker*, 2000]. It is somewhat surprising, then, that nearly half the storms from several urban catchments exhibit N:P < 16, the Redfield Ratio at which potential nutrient limitation may shift from N to P (Figure 3). Such stormwater inputs would be expected to alter the chemistry of recipient waters and establish the baseline nutrient conditions. These data suggest that for many urban recipient systems, either N or P could be the limiting nutrient. In support of this interpretation, a preliminary survey of urban streams at baseflow shows that mean inorganic N concentrations are much higher than those of Sycamore Creek, but as the phosphorus concentration often is also higher, mean N:P remains below the Redfield Ratio (Table 1).

In Cave Creek and Indian Bend Wash, different configurations and modifications lead to opposing predictions regarding nutrient limitation. As Cave Creek is not significantly modified in its upper reaches, we expect N limitation as in Sycamore Creek. A preliminary longitudinal survey in spring 2000 showed that N:P was <2 along most of the 11-km length

Table 1. Comparison of urban streams surveyed in 2003 within the Phoenix, Arizona, metropolitan area with long-term (time) and large-scale (space) surveys of Sycamore Creek, a native desert stream near Phoenix. All concentration units in μg/L.

Parameter	Urban Stream Survey		Sycamore Cr (time)		Sycamore Cr (space)	
	Mean	Limits	Mean	Limits	Mean	Limits
ammonium-N	35	9-94	na	na	nd	Nd
nitrate-N	1377	12-3850	95	0-450	35	0-279
SRP[1]	64	23-107	48	20-80	28	2-59
N:P[2]	8	0.5-88	~6	0-60	4	0-29
n[3]	7		211		260	

[1] SRP = soluble reactive phosphorus

[2] N:P = atomic ratio of inorganic N (ammonium+nitrate) to SRP; mean is average of individual N:P values for each site (or time).

[3] n = sample size

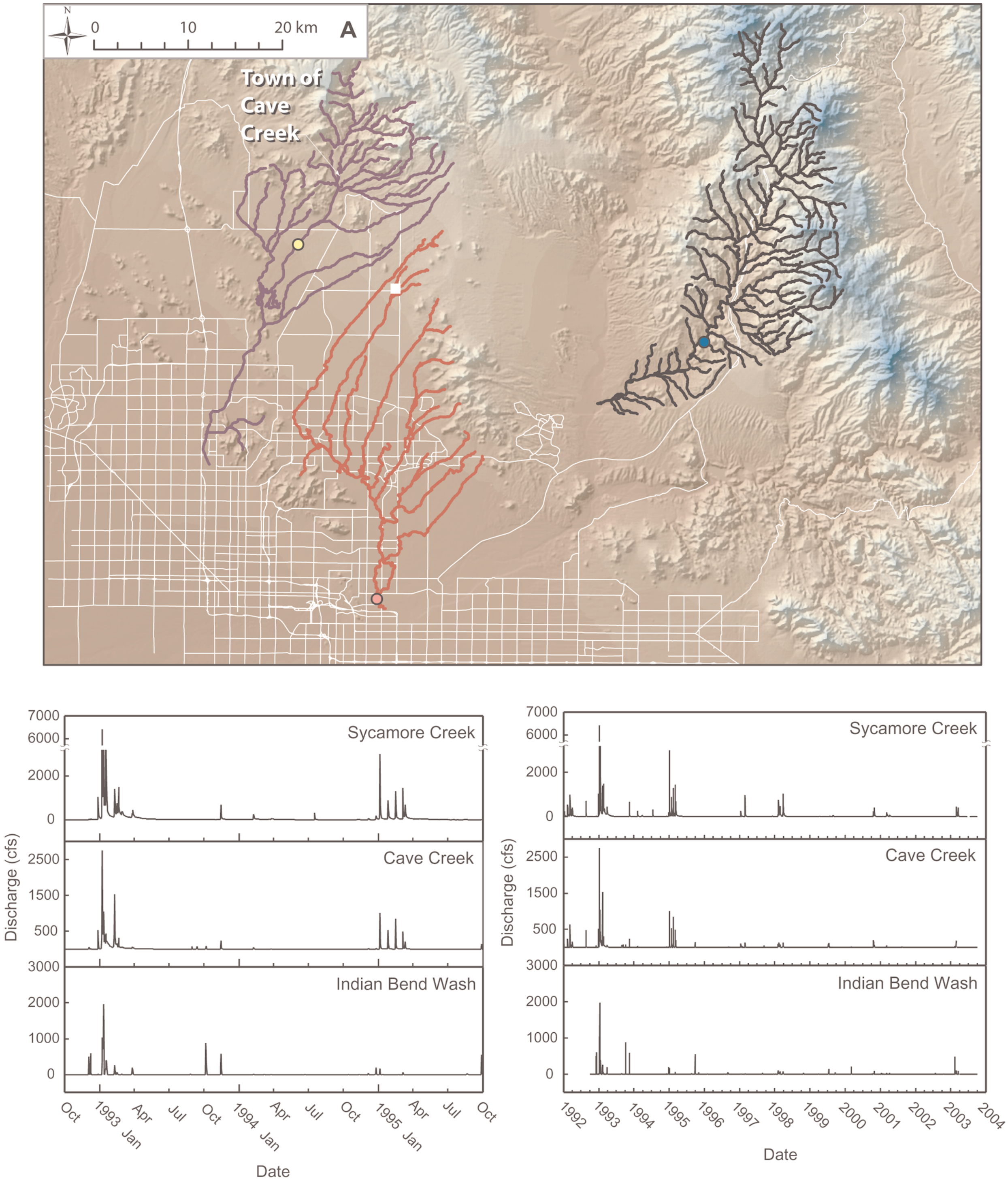

Plate 2. Discharge patterns for urban and regional desert streams. A) Map showing major drainage networks for Cave Creek (blue), Indian Bend Wash (red), and Sycamore Creek (black). Note the Phoenix area road network as an indicator of the degree of urbanization that has occurred in the watersheds. Colored circles show gauge locations for B. B) Discharge pattern for these drainages illustrating their overall narrow, peaked form with steep rising limbs. Variations in discharge are due to drainage basin structure and connectivity relative to gauge location (IBW station is low in drainage basin, but much of the contributing area for discharge is truncated by upstream diversion).

surveyed, with a dramatic increase, followed by just as dramatic a reduction, below a point-source input (waste water treatment plant effluent) in the town of Cave Creek [*N.B. Grimm*, unpublished data; see Plate 2 for location]. Indian Bend Wash has been shown to be P-limited during summer, when N additions from groundwater pumping are exceptionally high [*A. Goettl, W.J. Roach, and N.B. Grimm*, unpublished data]. This unique type of surface–groundwater interaction has a dramatic influence on the chemistry of Indian Bend Wash, and will be discussed further in that context.

Disturbance and Succession

Stream disturbances in the arid southwestern United States occur as monsoonal flash floods (summer) and spates arising from winter frontal storms, which rise quickly but have slower returns to base flow. Event hydrographs of desert streams have been called “parking lot” hydrographs because their characteristics are similar to those generated by rainfall on impervious surfaces. Thus the dogma of urban hydrology, that peak flows become flashier with urbanization [*Leopold,* 1968; *Arnold and Gibbons,* 1996], may depend on the context of that urbanization. Certainly, the event hydrographs of the unimpounded, small streams of Indian Bend Wash and Cave Creek resemble those of Sycamore Creek (Plate 2).

At larger scales, beyond the single event on a stream reach, the downstream change in stream discharge does appear to be altered by flood management practices and water delivery infrastructures. Along Cave Creek, the Cave Buttes Dam reduces the expected peak flood from about 1500 m^3/s to an outflow of 14 m^3/s. This two orders-of-magnitude decrease in flood discharge clearly allows for downstream development and additional stream diversions [*MacLeod,* 2003; Plate 3]. Furthermore, Cave Creek has been completely truncated downstream at the southeast–northwest trending Arizona Canal where it is diverted outside of its original drainage basin to the Agua Fria River (Plate 1), rather than its original path through what is now downtown Phoenix [*MacLeod*, 2003; Plate 2]. Severe flooding in Cave Creek’s original channel in 1921, no doubt, spurred subsequent dramatic flow path alterations.

In Indian Bend Wash, the Arizona Canal cuts across the lower third of the wash, severing storm-water delivery pathways and effectively reducing the contributing area. This effect manifests as a downstream reduction in total discharge associated with individual flood events (Figure 4). We used discharge records with high temporal resolution from five gauges operated by the Flood Control District of Maricopa County (FCDMC) to examine longitudinal changes in volume and peak flow of several floods in IBW since 1999. The Central Arizona Project Canal and Arizona Canal truncated all contributing runoff downstream, although the Arizona Canal did permit conveyance in the IBW channel. The volume per flood event (integral of the hydrograph) was reduced as water passed through the Arizona Canal section, where total drainage area remains constant (Figure 4). Presumably, the total flood volume dropped because of transmission loss. The peak discharge also decreased for the same reason. An increase in total flood volume observed between the lowermost two gauges may have resulted from increased contributing area (notably, tributary input) over this reach. Further effects are likely to include a reduced sediment supply to the stream, resulting in bank erosion, and reduced surface water–groundwater (SW–GW) interactions.

Succession has not been studied in urban streams of CAP LTER, or in urban streams elsewhere, to our knowledge. Yet, if the disturbance regime is similar, we might expect similar patterns in biomass accumulation of algae and macrophytes and consequent nutrient retention because sunlight and nutrient supplies are ample. One management activity that likely changes these patterns, however, is the control of algal and macrophyte growth using algicides and herbicides. In fact, Indian Bend Wash receives such inputs on a regular basis as frequently as once a week during the summer growing season. These inputs reduce rates of primary production and may limit the ability of the system to retain nutrients because autotrophic assimilation decreases. These treatments are at least in part necessitated because of stimulation of primary productivity by nutrient subsidies from the urban landscape. Thus, feedbacks between biological and social processes (i.e., aesthetic values) may mediate the ability of the system to transform and retain nutrients. Finally, recruitment and succession of riparian vegetation, which respond to longer-term cycles of flood and drought in native streams, are likely severely curtailed either by intensive landscape management along watercourses, or by water stress from reductions in surface flow and water table elevation.

Surface Water–Groundwater Interaction

Both overall urbanization of watersheds and direct land-cover change along streams have altered the potential for SW–GW interaction, and the nature of that interaction. In the Cave Creek channel, progressive channelization and ultimately canalization from the upper mainstem to the most downstream diversion channel traces in space increasingly severe modifications [Plate 3; *MacLeod,* 2003]. At its lower reaches, Cave Creek consists of a concrete-lined flume primarily intended to convey floodwater; here SW–GW interactions are completely precluded.

Indian Bend Wash has been modified over time as the City of Scottsdale has grown. Earliest land uses were small patches

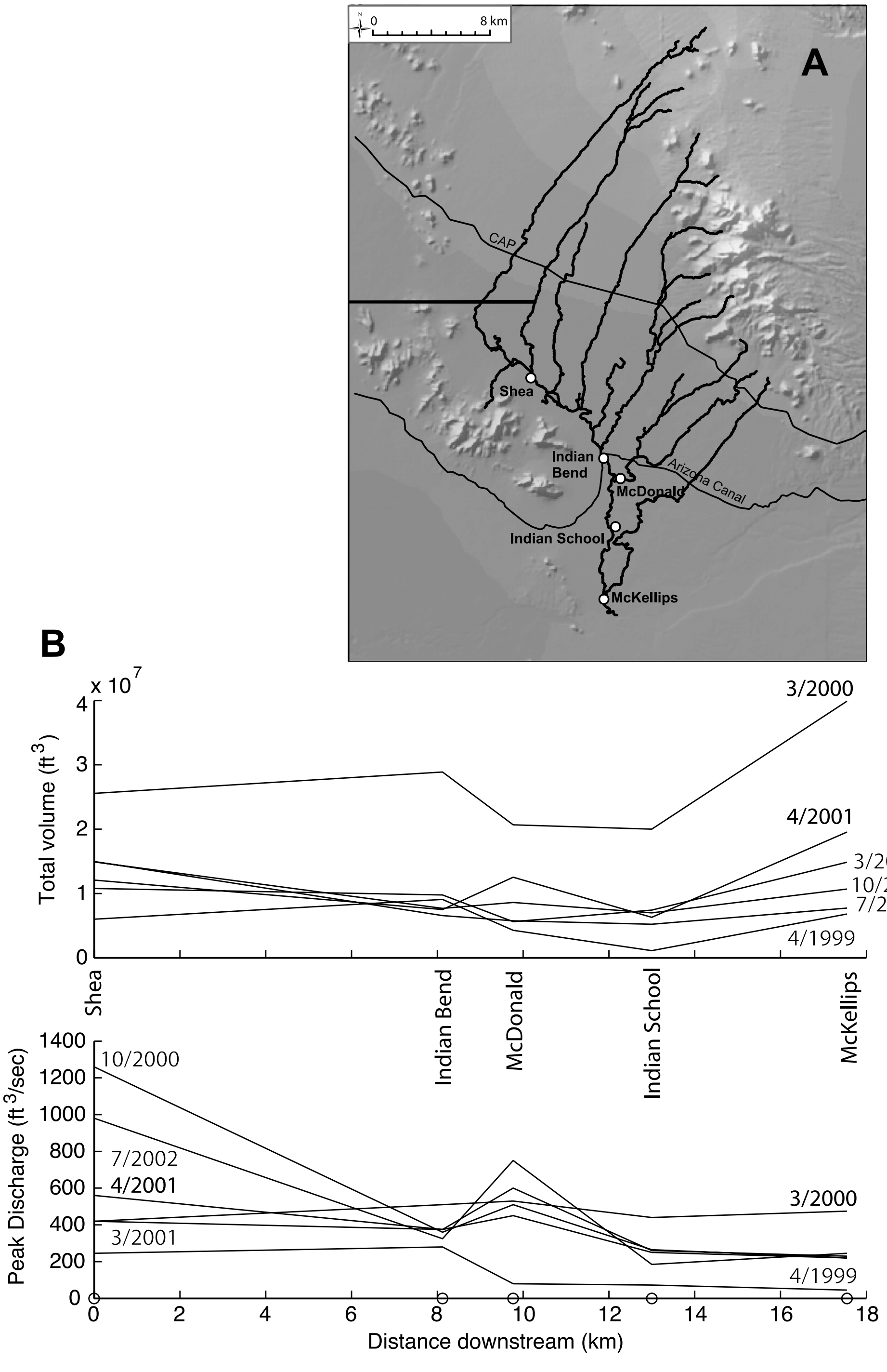

Figure 4. A) Stream gauges (white circles) along lower Indian Bend Wash of the Flood Control District of Maricopa County (http://www.fcd.maricopa.gov/Services/ALERT/). The Central Arizona Project and Arizona Canals partition the Indian Bend Wash watershed (thin lines); the network of IBW tributaries is shown on the backdrop of a topographic image. B) Total volume and peak discharge for five representative flood events at each of the five gauges. As floods move through the watershed, total volume consistently decreases in the area of contributing-area cutoff (McDonald and Indian School). Peak discharge in general decreases as hydrographs attenuate and transmission losses mount.

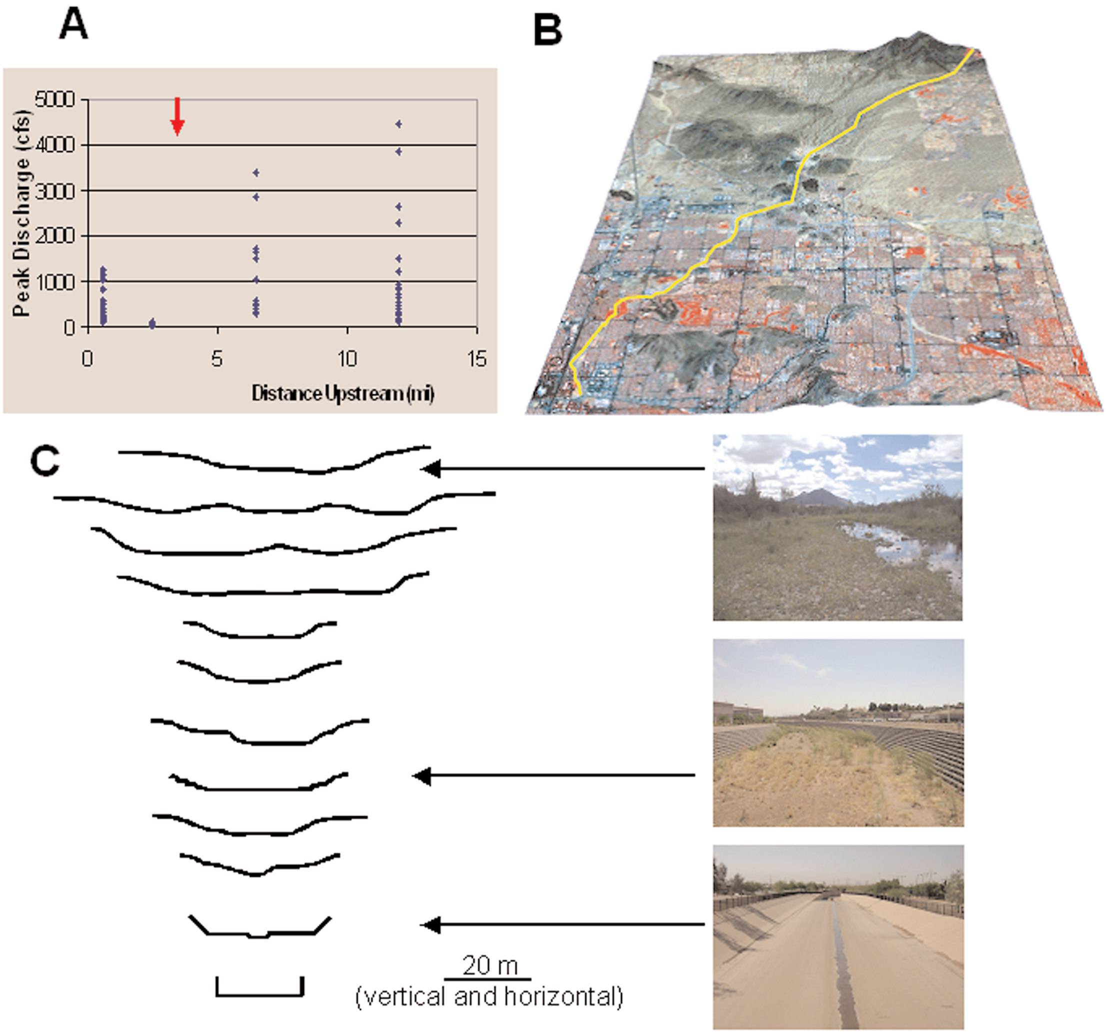

Plate 3. A) Peak flood discharges from 1999–2002 in Cave Creek, Arizona. Arrow denotes location of Cave Buttes Dam, located between upper and lower gauges (red dots on map, B). B) Road grid included to illustrate the degree of urbanization of the catchment. Note dramatic attenuation of peak discharge below the dam. C) Cross sections of Cave Creek, illustrating the dramatic alteration of channel form from the upper, relatively unmodified section to the lower, channelized reaches in the urban area.

of farms and large expanses of desert; today, the majority of the watershed has been urbanized with only 45% remaining as (doomed) desert [*W.J. Roach et al.,* unpublished data]. Modifications of the channel itself have undoubtedly reduced SW–GW interactions: the straightened channel permits little gravel bar–stream exchange, lined stream channels (concrete, clay) and lake bottoms limit vertical water loss (and thus exchange), lakes have been constructed, which means water has less contact with sediments, and groundwater has dropped. In 1900, the water table was just a few meters below the land surface at the IBW–Salt River confluence and significant SW–GW exchange was likely there. Upstream, the land surface increased in elevation more rapidly than the water table so that 16 km upstream from the confluence, the water table was almost 50 m below the land surface. Significant pumpage for agricultural and municipal purposes occurred during the 20th century, with a peak in about 1955. The water table dropped significantly in response and by the 1970s and 1980s, and later as pumpage declined, the depth to water at the IBW–Salt River confluence averaged about 60 m below the land surface, with a peak of 95 m [*Schumann and Associates*, 1998].

Surface water–groundwater dynamics and chemistry in the Salt River changed dramatically with the filling of Tempe

Town Lake after 1999 [*Dahlen*, 2000; *Fergason,* 2001]. To prevent infiltration and loss of lake water to groundwater, slurry walls to the shallow bedrock minimize infiltration losses on the western end of the lake, while the eastern portion is lined with clay and outfitted with 10 recirculation wells that return water to the surface. These management activities have resulted in mounding of the water table under and near the west end, and the development of a composite cone of depression around the recirculation wells and a net lowering of the water table to the east (beyond the seasonally modulated regional water table fluctuations) [*Fergason,* 2001]. Groundwater chemistry monitoring during the first year of the lake's existence demonstrated the clear lack of isolation of the lake and its infiltrated and recirculated water from the regional native groundwater, and evidence for contaminant mobilization, dilution of shallow groundwater, and the capture of recharged wastewater treatment plant effluent from an upstream plant [*Dahlen*, 2000].

The extensive modifications of both IBW and the Salt River/Tempe Town Lake have dramatically changed SW–GW interactions, introducing a new constraint on the way these systems can interact. Because groundwater levels have dropped, the only time groundwater interacts with surface flows is when it is pumped to the surface. The chemical character of surface waters is then much more like that of groundwater. Shallow surface–subsurface interactions that characterize desert streams like Sycamore Creek have been largely eliminated as these exchanges have come under human control.

The reduction of SW–GW exchange associated with urban streams is likely to have a number of biogeochemical consequences. Because shallow groundwater zones often serve as transient storage sites, hydrologic residence time may be reduced in urban streams, decreasing the potential for both biotic and abiotic processes to influence nutrient chemistry [*Triska et al.*, 1989; *Morrice et al.*, 1997]. In addition, SW–GW interaction allows the coupling of oxic and anoxic processes (e.g., nitrification-denitrification) [*Baker et al*., 2000]. Reduction of SW–GW exchange may, therefore, alter the relative importance of these processes. Finally, reduction of SW–GW exchange may alter the spatial structure of nutrient availability, creating a longitudinal rather than a patchy organization. In sum, all of these consequences serve to increase the importance of advective transport and decrease the retentive capabilities of stream ecosystems. These changes point to the relevance of intrinsic native ecosystem characteristics. Because desert stream biogeochemistry is driven by SW–GW exchange [*Dent et al.,* 2001], modifications associated with urbanization may dramatically influence instream processes. In streams where SW–GW interactions are less prevalent, modifications of hydraulic conductivity may be less important.

Land–Water Interaction: Nutrient Inputs

Urbanization does not have a major effect on regional climate that drives rainfall characteristics, with some exceptions [*Cerveny and Balling*, 1998], so we argue that the episodic nature of land–water interaction is largely unchanged. Rainstorms do result in dramatic, rapid changes in discharge in urban streams, just as in the desert. How the storm runoff actually traverses upland systems until it reaches streams and other recipient systems, on the other hand, may be dramatically altered by urbanization. Upland flowpaths are redirected, shortened, diverted into retention structures, and in many cases rechanneled into underground pipes. In Scottsdale, canals running parallel with contour lines have divided Indian Bend Wash into three separate basins with severely reduced hydrologic connections (Figure 4). As many investigators have pointed out, the network of urban flowpaths becomes much more rectangular, rather than dendritic, in appearance [*Paul and Meyer*, 2001]. New recipient systems also appear in urbanized landscapes, many created explicitly to slow down the movement of water from urban surfaces to surface waters. An example is the neighborhood retention basin, which collects storm water and quickly moves it to groundwater via deep drywells. Similarly, gravel-mining operations in some of the semi-urbanized channels increase transient storage zone size, decreasing peak flows. New recipient systems like these have no real parallels in the desert landscape; thus, we expect they will have different biogeochemical characteristics.

The chemistry of episodic runoff is influenced by the nature of the land surface over which it runs. Like the desert, CAP's urban stormwater is enriched both in N and P; however, the extent of enrichment in N is not as great as one would predict based on known storage on impervious surfaces [*Hope et al.,* 2004; Figure 3] suggesting retention of N during transport (see below). The chemistry of stormwater runoff in the Phoenix metropolis further suggests that land–water interactions are altered by the urban variety of land-cover and land-use change. Land–water interactions can be examined by investigating the chemistry of stormwater runoff because stormwater runoff integrates the materials stored on the terrestrial surface of a catchment and delivers them to streams and downstream recipient systems. Consequently, the effects of land-cover and land-use change on land–water interactions can be assayed by comparing nutrient load in stormwater runoff exported from catchment ecosystems that differ in terms of impervious surface amount, development type, and the spatial arrangement of patches germane to nutrient transport (e.g., preferential runoff flowpaths, retentive pervious patches, and stored sources of N and P). If urban varieties of land cover and land use alter land–water interactions (i.e., nutrient delivery from catchment uplands to stream), then nutrient loads in runoff should

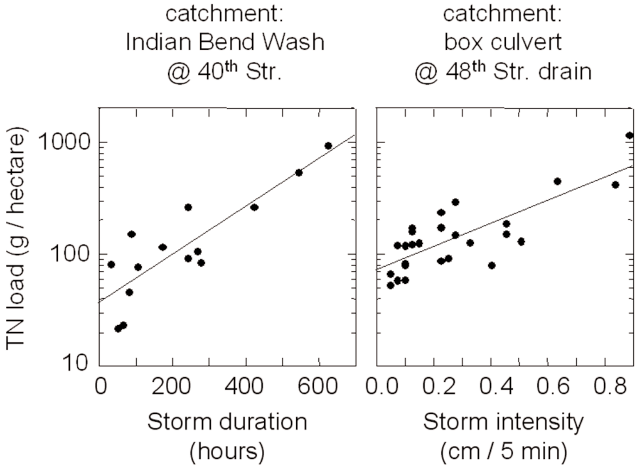

Figure 5. Load of total nitrogen (unfiltered samples) exported from two catchments in stormwater runoff. Each data point represents a single storm event (i.e., occurrence of rain in a catchment). Storms occurred during the period October 1991–October 2000. The Indian Bend Wash (IBW) catchment is 246.5 hectares (ha), is located in north-central Phoenix, is a tributary of IBW, is dominated by residential development, and is 37% impervious. The box culvert catchment is 15.8 ha, is located in central Phoenix, is a tributary of the Salt River, is dominated by industrial development, and is 80% impervious. In the IBW catchment, log10-TN load was a function of storm duration ($N = 14$, $p < 0.001$, $R^2 = 0.76$) but not of storm intensity ($p = 0.96$). In the box culvert catchment, log10-TN load was a function of storm intensity ($N = 26$, $p < 0.001$, $R^2 = 0.58$) but not of storm duration ($p = 0.17$). Storm intensity is the amount of rain to fall during the most active 5-minute window of the storm.

differ among contrasting urban catchments (even when attributes of storms generating the runoff are otherwise similar). Indeed, we observe that export of total N (TN) responds differentially to storms when compared between two contrasting Phoenix-area catchments. In the catchment located at Indian Bend Wash and 40th Street, TN responds to storm duration, whereas at the 48th Street drain catchment, TN responds to storm intensity (Figure 5). These data suggest that some intrinsic features of these catchments mediate how overland runoff from storms delivers nutrients to streams. Studying the export of eight nutrients from 12 catchments in 144 storms, Lewis and Grimm [unpublished data] document this phenomenon; namely, that features of the catchment mediate the delivery of nutrients from catchments to stream channels. It is currently under study whether such "features of the catchments" are shape, orientation, and size, or whether they are true reflections of anthropogenic land-use and land-cover changes (e.g., percent impervious, internal patch and flow network arrangement, local nutrient deposition, etc.).

Non-storm inputs of nutrients from the watershed to recipient systems may be greater in the city than in desert streams. Human activities like lawn watering or patio washing are relatively continuous on a large scale, and may move nutrients derived from fertilizer additions or accumulated dry deposition to streams. Groundwater is pumped from the deep, nitrate-rich aquifer in order to fill artificial lakes and to supply irrigation, as mentioned previously. Stormwater is in some cases captured, retained, and used to fill artificial lakes. Finally, wastewater from municipal uses is treated and discharged into the dry Salt River downstream from the center of Phoenix, where it contributes 100% of surface discharge. Biogeochemical consequences of this are huge: nitrate, phosphate, and organic carbon concentrations of this effluent-dominated river are an order of magnitude higher than concentrations upstream from the metropolitan area [*Edmonds*, 2004].

Nutrient Retention

As in desert landscapes, the CAP ecosystem shows an imbalance (inputs of ca. 76 kg ha^{-1} y^{-1} >> outputs of ca. 62 kg ha^{-1} y^{-1}) in its N budget [*Baker et al.*, 2001]. Although some of this can be accounted for by accumulation in groundwater and landfills, uncertainty exists in estimates of gaseous loss and sequestration in vegetation and soils. Whether streams and riparian zones could be hot spots of N retention or removal in this urban ecosystem depends on whether the mechanisms known to be associated with streams are intact. These mechanisms can be attributed to the stream itself or to surrounding riparian areas, and include assimilation and denitrification.

In-stream N retention, as measured by nutrient spiraling techniques, has been reported for very few urban streams, although a current research effort stemming from the LINX project [e.g., *Mulholland et al.*, 2000] has extended to urban streams in eight biomes. Uptake length, a measure of the distance traveled by an average N atom in water before it is taken up, is one measure of retention that incorporates both mechanisms; for Sycamore Creek, uptake lengths for ammonium vary from 20–80 m and for nitrate from 10–200 m [*Marti et al.,* 1997; Table 2]. Our preliminary results from short-term nutrient addition experiments [see *Mulholland et al.,* 2002, for methodology] show that urban streams have much longer uptake lengths (are less retentive) than does Sycamore Creek. Furthermore, although only four streams were measured, uptake length was longer in the two concrete-lined channels than in the two earthen channels (Table 2).

In riparian zones, hot spots of N removal are created when flowpaths with high nitrate intersect locations of high denitrification potential [*Hill*, 1996; *Gold et al.*, 1998; *McClain et al.*, 2003]. Clearly, flowpaths need to be intact for this to occur, so all of the changes in SW–GW interaction discussed above are relevant to whether urban riparian zones may be sinks for N. However, new recipient systems like neighborhood

Table 2. Uptake lengths for nitrate from 4 urban streams in Phoenix, AZ, and Albuquerque, NM [*R.W. Sheibley et al.*, unpublished data]. Uptake length is the distance in a stream traveled by a nutrient ion between its release into the water column until it is taken up again (by biotic or abiotic mechanisms). Longer uptake lengths indicate that the stream ecosystem is less efficient at retaining nutrients than a stream with short uptake lengths Uptake lengths for Sycamore Creek are given for comparison [from *Marti et al.,* 1997; *N.B. Grimm*, unpublished data].

Channel Type	Limits of uptake lengths (S_w) observed for NO_3
Urban, earthen lined (n=2)	300-500 m
Urban, concrete lined (n=2)	600-800 m
Native, Sycamore Cr (n=10)	10-200 m

retention basins show high denitrification potentials [*Zhu et al.*, 2004]; thus, they may represent alternative lowland sites of high N retention. This may occur on a smaller scale as well, as even small grassy areas between impervious surfaces, with their high concentrations of stored N, and a catchment's outlet may be hot spots of N retention [*Hope et al.*, 2004]. Such localized sites of high N retention result in lower export of N from urban catchments, whereas no similar effects for P or C are evident from these small-scale nutrient budgets.

The retentive capacity of streams and landscapes integrates the other four aspects of desert streams we have discussed, in that this capacity may be dependent on successional stage, nutrient inputs and limitation, and the spatial and temporal dynamics of SW–GW and terrestrial–aquatic linkages [*Grimm et al.*, 2003]. As a result, system context (e.g., imperviousness of soils, high P content of basin geology), intrinsic ecosystem features (e.g., dependence on SW–GW interactions, presence of rapidly growing biota), nature of modifications (e.g., degree of urbanization, prevalence of hydrologic engineering structures) and feedbacks (e.g., between nutrient availability, primary production, and algicide use) are all relevant to how urbanization affects the retentive capacity of streams (Figure 1).

SUMMARY OF COMPARISONS AND CONCLUSIONS

Role of Domain and Intrinsic Ecosystem Features in Governing Response to Urbanization

Our analysis of five key characteristics comparing desert streams (e.g., Sycamore Creek) and urban streams in the desert (e.g., Indian Bend Wash, Cave Creek, and other aquatic ecosystems in the Phoenix metropolitan area) shows similarities in individual disturbance events and, indeed, the role of episodic events in moving nutrients from the terrestrial landscape to recipient aquatic ecosystems. Succession, though not studied as thoroughly, is likely to exhibit similar characteristics to desert streams except when interrupted by human management activities (e.g., herbicide treatments, dredging, landscape planting). Locations of GW recharge in the arid Southwest are primarily in stream channels, and this is similar in urban systems as ephemeral channels increasingly are exploited to recharge groundwater—although the perched water tables of many of these streams results in reduced SW–GW interactions. Taken together, the similarities between aridland streams and their urban counterparts owe largely to constraints imposed by the context (Figure 1). For example, the arid climate, high flash-flood magnitude, and episodic land–water interactions are not obliterated by land-use change. In some cases studied, human modification has been insufficient to eliminate similarities of urban to native streams that arise from intrinsic ecosystem features (nature of the ecosystem), such as geomorphic structure and sediment content of channels that affect SW–GW interactions, or abundant P derived from P-rich minerals.

Nature of Land-Use Change and Human-Ecosystem Feedbacks

There are clearly many differences that can help inform future research on the functioning of streams draining landscapes with a high degree of land-cover change, such as urban areas. Most of these differences can be attributed to the litany of modifications and manipulations of streams and land cover in urban areas (i.e., nature of the land-use change; Figure 1). Owing to higher nutrient loading and altered hydrology, streams are likely not N limited, as are their desert counterparts. Although individual disturbances may be similar in character, regimes of disturbance clearly are changed. Connectivity within the landscape is interrupted by introduction of wholly new flowpaths (such as canals, pipes, storm drains, etc.), and this, of course, leads to dramatic alteration in SW–GW interactions, which play such a key role in desert stream biogeochemistry. Because of this loss of SW–GW connection and associated complexity of stream-riparian corridors, streams and riparian zones in urban areas make poor filters for N and other nutrients. However, alternate sites of nutrient retention have been formed because of attempts to retard loss of water to rivers. In effect, water is moved from streams into more convenient and manageable conveyance systems, thus probably the most significant urbanization effect is the complete loss of streams, and possibly their associated sediment flux. Given that desert streams tend to transport predominantly bedload and suspended load under natural flow conditions, this is not desired, and most engineering and urbanization surface changes decrease sediment supply (rather than transport capacity). Many high-quality datasets available from monitoring agencies can be exploited to explore these relationships.

We have identified several feedbacks between the socioecosystem and the agents of change that are especially applicable to streams. People do enjoy water amenities, and therefore some of the more recent modifications in the Phoenix metro area are meant to improve recreational access to streams or "greenways." In Indian Basin Wash, lowered GW levels coupled to establishment of man-made, perennial lakes has resulted in a feedback wherein more water is pumped from GW to fill the lakes, with implications discussed earlier for SW–GW interaction, biogeochemistry, and riparian vegetation. Finally, the causal chain leading from elevated nutrient loading to high aquatic primary productivity to use of herbicide and algicide to control growth is a clear example of a feedback.

Trade-Offs in Urbanization of Aridland Stream Ecosystems

Returning to the issue of trade-offs, we identified several trade-offs that affect urban stream ecosystem function. The interaction of society and human activities with stream ecosystems is more than just a series of impacts; rather, extraction of ecosystem services is a central feature of this interaction. Specifically, water supply, flood protection, and recreational amenities are chief among the benefits or services that society expects to gain from streams. Not surprisingly in the arid urban environment of central Arizona/Phoenix, water delivery has been the primary driver of modifications to the hydrology and geomorphic structure of its streams, and these modifications in turn have led to a loss of streams. In the case of Indian Bend Wash, however, floodplain modifications meant to provide both flood management and recreational/aesthetic amenities have created a completely new kind of urban stream—consisting of a chain of lakes filled by groundwater and connected by slow-moving and occasionally dry stream segments. Recent interest in creation of more such "restored riparian areas" may result in multiple new stream ecosystem configurations in this urban environment.

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REFERENCES

Ackerly, N. W., J. B. Howard, and R. H. McGuire, *La Ciudad Canals: A Study of Hohokam Irrigation Systems at the Community Level,* Volume 2 of 7 in Ciudad Monograph Series, Arizona Department of Transportation, 1987.

Anderson, T. W., *Summary of the Southwest Alluvial Basins, Regional Aquifer-System Analysis, South–Central Arizona and Parts of Adjacent States,* U.S. Geological Survey Professional Paper 1406-A, USGS, Washington, DC, 1995.

Arnold, C. L., and C. J. Gibbons, Impervious Surface Coverage: The Emergence of a Key Environmental Indicator, *Journal of the American Planning Association,* 62, 243–258, 1996.

Baker, L. A., A. J. Brazel, N. Selovar, C. A. Martin, F. Steiner, N. E. McIntyre, A. Nelson, and L. Musacchio, Local Warming: Feedbacks from the Urban Heat Island, *Urban Ecosystems*, 6, 83–203, 2002.

Baker, L. A., D. Hope, Y. Xu, J. Edmonds, and L. Lauver, Nitrogen Balance for the Central Arizona–Phoenix (CAP) Ecosystem, *Ecosystems*, 4, 582–602, 2001.

Baker, M. A., C. N. Dahm, and H. M. Valett, Anoxia, Anaerobic Metabolism, and Biogeochemistry of the Stream Water–Groundwater Interface, *Streams and Groundwaters*, edited by J. B. Jones and P. J. Mulholland, pp. 259–283, Academic Press, San Diego, CA, 2000.

Baker, V. R., Stream Channel Response to Floods, With Examples from Central Texas, *Geological Society of America Bulletin*, 88, 1057–1071, 1977.

Belnap, J., J. Welter, N. B. Grimm, N. Barger, and J. Ludwig, Linkages Between Microbial and Hydrologic Processes in Arid and Semi-Arid Watersheds, *Ecology*, in press, 2004.

Cerveny, R. S., and R. C. Balling, Weekly Cycles of Air Pollutants, Precipitation and Tropical Cyclones in the Coastal NW Atlantic Region, *Nature,* 394, 561–563, 1998.

City of Scottsdale, *Indian Bend Wash*, Communications and Public Affairs, Scottsdale, AZ, 1985.

City of Tempe, *Tempe Town Lake on the Rio Salado*, http://www.tempe.gov/rio/, 2003.

Cooper, J. R., and G. C. Cone, Geologic Map and Cross-Section of Arizona/Generalized from Geologic Map of Arizona (1969) by the Arizona Bureau of Mines and U.S. Geological Survey, and from Arizona Highway Geologic Map by the Arizona Geological Society (1967), 1969.

Dahlen, P. R., *The Effects of Engineered Recharge of Untreated Surface Water in Southwestern Ephemeral Stream Channels*, M.S. thesis, Arizona State University, Tempe, 2000.

Dahm, C. N., N. B. Grimm, P. Marmonier, H. M. Valett, and P. Vervier, Nutrient Dynamics at the Interface Between Surface Waters and Groundwaters, *Freshwater Biology*, 40, 427–451, 1998.

Dent, C. L., and N. B. Grimm, Spatial Heterogeneity of Stream Water Nutrient Concentrations over Successional Time, *Ecology*, 80, 2283–2298, 1999.

Dent, C. L., N. B. Grimm, and S. G. Fisher, Multiscale Effects of Surface–Subsurface Exchange on Stream Water Nutrient Concentrations, *Journal of the North American Benthological Society*, 20, 162–181, 2001.

Edmonds, J. W., *Dissolved Organic Carbon Dynamics in Urban and Desert Stream Ecosystems*, Ph.D. dissertation, Arizona State University, Tempe, 2004.

Fergason, K.C., *Investigation of Changes in Water Table Elevation Associated with Tempe Town Lake*, M.S. thesis, Arizona State University, Tempe, 2001.

Fisher, S. G., Structure and Dynamics of Desert Streams, *in Pattern and Process in Desert Ecosystems,* edited by W. G. Whitford, pp.119–139, University of New Mexico Press, Albuquerque, NM, 1986.

Fisher, S. G., L. J. Gray, N. B. Grimm, and D. E. Busch, Temporal Succession in a Desert Stream Ecosystem Following Flash Flooding, *Ecological Monographs*, 52, 93–110, 1982.

Fisher, S. G., N. B. Grimm, E. Marti, R. M. Holmes, and J. B. Jones, Material Spiraling in Stream Corridors: A Telescoping Ecosystem Model, *Ecosystems*, 1, 19–34, 1998.

Fisher, S. G., J. R. Welter, J. D. Schade, and J. C. Henry, Landscape Challenges to Ecosystem Thinking: Creative Flood and Drought in the American Southwest, *Sciencia Marina*, 65, 181–192, 2001.

Folke, C., A. Jansson, J. Larsson, and R. Costanza, Ecosystem Appropriation by Cities, *AMBIO,* 26, 167–172, 1997.

Gold, A. J., P. A. Jacinthe, W. R. Groffman, W. R. Wright, and R. H. Puffer, Patchiness in Groundwater Nitrate Removal in a Riparian Forest, *Journal of Environmental Quality*, 27, 146–155, 1998.

Graf, W. L., Impact of Suburbanization on Fluvial Geomorphology*,* *Water Resources Research*, 11, 690–692, 1975.

Graf, W., editor, *Recent Channel Changes in the Salt River, Phoenix, Arizona: A Field Guide*, Arizona State University, Tempe, 1998.

Grimm, N. B., Nitrogen Dynamics During Succession in a Desert Stream, *Ecology*, 68, 1157–1170, 1987.

Grimm, N. B., Disturbance, Succession, and Ecosystem Processes in Streams: A Case Study from the Desert, *in Aquatic Ecology: Scale, Pattern and Process,* edited by P. S. Giller, A. G. Hildrew, and D. G. Raffaelli, pp. 93–112, Blackwell Scientific, London, 1994.

Grimm, N. B., and S. G. Fisher, Nitrogen Limitation in a Sonoran Desert Stream, *Journal of the North American Benthological Society*, 5, 2–15, 1986a.

Grimm, N. B., and S. G. Fisher, Nitrogen Limitation Potential of Arizona Streams and Rivers, *Journal of the Arizona–Nevada Academy of Science*, 21, 31–43, 1986b.

Grimm, N. B., and S. G. Fisher, Stability of Periphyton and Macroinvertebrates to Disturbance by Flash Floods in a Desert Stream, *Journal of the North American Benthological Society*, 8, 293–307, 1989.

Grimm, N. B., and S. G. Fisher, Responses of arid-land streams to changing climate, *in Global Climate Change and Freshwater Ecosystems,* edited by P. Firth and S. Fisher, pp. 211–233, Springer-Verlag, New York, 1992.

Grimm, N.B., S. E. Gergel, W. H. McDowell, E. W. Boyer, C. L. Dent, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, M. McClain, and G. Pinay, Merging Aquatic and Terrestrial Perspectives of Nutrient Biogeochemistry, *Oecologia,* 442:485–501, 2003.

Grimm, N. B., J. M. Grove, S. T. A. Pickett, and C. L. Redman, Integrated Approaches to Long-term Studies of Urban Ecological Systems, *BioScience*, 50, 7, 571–584, 2000.

Grimm, N. B., H. M. Valett, E. H. Stanley, and S. G. Fisher, Contribution of the Hyporheic Zone to Stability of an Arid-Land Stream, *Internationale Vereinigung fur Theoretische und Angewandt Limnologie, Verhandlungen,* 24, 1595–1599, 1991.

Helfield, J. M., and R. J. Naiman, Effects of Salmon-Derived Nitrogen on Riparian Forest Growth and Implications for Stream Productivity, *Ecology,* 82, 2403–2409, 2001.

Hill, A. R., Nitrate Removal in Stream Riparian Zones, *Journal of Environmental Quality*, 25, 743–755, 1996.

Holmes, R. M., S. G. Fisher, and N. B. Grimm, Parafluvial Nitrogen Dynamics in a Desert Stream Ecosystem, *Journal of the North American Benthological Society*, 13, 468–478, 1994.

Holmes, R. M., J. B. Jones, S. G. Fisher, and N. B. Grimm, Denitrification in a Nitrogen-Limited Stream Ecosystem, *Biogeochemistry*, 33, 125–146, 1996.

Hope, D., M. W. Naegeli, A. Chan, and N. B. Grimm, Nutrients on Asphalt Parking Surfaces in an Arid Urban Ecosystem, *Water, Air and Soil Pollution Focus*, 4, 371–390, 2004.

Jones, J. B., S. G. Fisher, and N. B. Grimm, Nitrification in the Hyporheic Zone of a Desert Stream Ecosystem, *Journal of the North American Benthological Society*, 14, 249–258, 1995a.

Jones, J. B., S. G. Fisher, and N. B. Grimm, Vertical Hydrologic Exchange and Ecosystem Metabolism in a Sonoran Desert Stream, *Ecology*, 76, 942–952, 1995b.

Jones, J. B., R. M. Holmes, S. G. Fisher, N. B. Grimm, and D. M. Greene, Methanogenesis in Arizona, USA Dryland Streams, *Biogeochemistry*, 31, 155–173, 1995c.

Jones, J. B., and P. J. Mulholland, *Streams and Ground Waters,* Academic Press, San Diego, 2000.

Lauver, L., and L. A. Baker, Wastewater Reuse: Implications for Nitrogen Export and Retention in the Phoenix Ecosystem, *Water Research*, 34, 2734–2760, 2000.

Leopold, L. B, *Hydrology for Urban Planning: A Guidebook on the Hydrologic Effects of Urban Land Use*, U.S. Geological Survey Circular 554, Washington, DC, 1968.

Luck, M. A., G. D. Jenerette, J. G. Wu, and N. B. Grimm, The Urban Funnel Model and the Spatially Heterogeneous Ecological Footprint, *Ecosystems*, 4, 782–796, 2001.

MacLeod, A., *Artificial Hydrologic Controls and the Geomorphology of the Greater Phoenix Area*, Honors thesis, Arizona State University, Tempe, 2003.

Marti, E., N. B. Grimm, and S. G. Fisher, Pre- and Post-flood Retention Efficiency of Nitrogen in a Sonoran Desert Stream, *Journal of the North American Benthological Society*, 16, 805–819, 1997.

McClain, M. E., E. W. Boyer, C. L. Dent, S. E. Gergel, N. B. Grimm, P. M. Groffman, S. C. Hart, J. W. Harvey, C. A. Johnston, E. Mayorga, W. H. McDowell, and G. Pinay, Biogeochemical Hot Spots and Hot Moments at the Interface of Terrestrial and Aquatic Ecosystems, *Ecosystems*, 6, 301–312, 2003.

Montgomery, E. L., and J. W. Harshbarger, Arizona Hydrogeology and Water Supply, *Geologic Evolution of Arizona*, edited by S. J. Reynolds, pp. 827–840, Arizona Geological Society Digest, Tucson, 1989.

Morrice, J. A., H. M. Valett, C. N. Dahm, and M. E. Campana, Alluvial Characteristics, Groundwater–Surface Water Exchange and Hydrological Retention in Headwater Streams, *Hydrological Processes*, 11, 253–267, 1997.

Mulholland, P. J., J. L. Tank, D. M. Sanzone, W. M. Wollheim, B. J. Peterson, J. R. Webster, and J. L. Meyer, Nitrogen Cycling in a Forest Stream Determined by a N-15 Tracer Addition, *Ecological Monographs*, 70, 471–493, 2000.

Mulholland, P. J., J. L. Tank, J. R. Webster, W. B. Bowden, W. K. Dodds, S. V. Gregory, N. B. Grimm, S. K. Hamilton, S. L. Johnson, E. Marti, W. H. McDowell, J. L. Merriam, J. L. Meyer, B. J. Peterson, H. M. Valett, and W. M. Wollheim, Can Uptake Length in Streams Be Determined by Nutrient Addition Experiments? Results from an Interbiome Comparison Study, *Journal of the North American Benthological Society*, 21, 544–560, 2002.

Paul, M. J., and J. L. Meyer, Streams in the Urban Landscape, *Annual Review of Ecology and Systematics*, 32, 333–365, 2001.

Richey, J. E., M. A. Perkins, and C. R. Goldman, Effects of Kokanee Salmon (*Oncorhynchus nerka*) Decomposition on the Ecology of a Subalpine Stream, *Journal of the Fisheries Research Board of Canada*, 32, 817–820, 1975.

Schade, J. D., S. G. Fisher, N. B. Grimm, and J. A. Seddon, The Influence of a Riparian Shrub on Nitrogen Cycling in a Sonoran Desert Stream, *Ecology*, 82, 3363–3376, 2001.

Schumann, H. H., and Associates, *The Potential for Land Subsidence in the North Indian Bend Wash (NIBW) Site,* Report prepared for NIBW participating companies, 4199 Campus Drive, Suite 550, Irvine, CA 92612, 1998.

Stefanov, W. L., M. S. Ramsey, and P. R. Christensen, Monitoring Urban Land Cover Change: An Expert System Approach to Land Cover Classification of Semiarid to Arid Urban Centers, *Remote Sensing of the Environment,* 77, 173–185, 2001.

Sterner, R., and J. Elser, *Ecological Stoichiometry: The Biology of Elements from Molecules to the Biosphere*, Princeton University Press, Princeton, NJ, 2002.

Triska, F. J., V. C. Kennedy, R. J. Avanzino, G. W. Zellweger, and K. E. Bencala, Retention and Transport of Nutrients in a 3rd-Order Stream in Northwestern California: Hyporheic Processes, *Ecology*, 70, 1893–1905, 1989.

U.S. Army Corps of Engineers, *Gila River Basin, Arizona, Indian Bend Wash, Design Memorandum No. 1, General Design Memorandum, Phase II, Project Design for Indian Bend Wash*, Los Angeles, 1975.

U.S. Census Bureau, *Census 2000* (http://www.census.gov/main/www/cen2000.html), 2000.

Valett, H. M., S. G. Fisher, N. B. Grimm, and P. Camill, Vertical Hydrologic Exchange and Ecological Stability of a Desert Stream Ecosystem, *Ecology*, 75, 548–560, 1994.

Webster, J. R., P. J. Mulholland, J. L. Tank, H. M. Valett, W. K. Dodds, B. J. Peterson, W. B. Bowden, C. N. Dahm, S. Findlay, S. V. Gregory, N. B. Grimm, S. K. Hamilton, S. L. Johnson, E. Marti, W. H. McDowell, J. L. Meyer, D. D. Morrall, S. A. Thomas, and W. M. Wollheim, Factors Affecting Nitrogen Retention in Streams—an Inter-Biome Perspective, *Freshwater Biology*, 48, 1329–1352, 2003.

Zhu, W., N. D. Dillard, and N. B. Grimm, Urban Nitrogen Biogeochemistry: Status and Processes in Green Retention Basins, *Biogeochemistry*, in press, 2004.

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