Unintended Consequences of Urbanization for Aquatic Ecosystems: A Case Study from the Arizona Desert

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Many changes wrought during the construction of "designer ecosystems" are intended to ensure—and often succeed in ensuring—that a city can provide ecosystem goods and services; but other changes have unintended impacts on the ecology of the city, impairing its ability to provide these critical functions. Indian Bend Wash, an urbanizing watershed in the Central Arizona–Phoenix (CAP) ecosystem, provides an excellent case study of how human alteration of land cover, stream channel structure, and hydrology affect ecosystem processes, both intentionally and unintentionally. The construction of canals created new flowpaths that cut across historic stream channels, and the creation of artificial lakes produced sinks for fine sediments and hotspots for nitrogen processing. Further hydrologic manipulations, such as groundwater pumping, linked surface flows to the aquifer and replaced ephemeral washes with perennial waters. These alterations of hydrologic structure are typical by-products of urban growth in arid and semiarid regions and create distinct spatial and temporal patterns of nitrogen availability.

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Hydrology and geomorphology are important determinants of the structure and function of fluvial ecosystems. Streamflow quantity and timing are often viewed as "master variables" limiting the distribution and abundance of riverine species (Poff et al. 1997) and the flux of nutrients into streams (Grimm 1987, Welter et al. 2005). Geomorphic structure—particularly channel form—is known to influence community composition (Vannote et al. 1980) and trophic interactions (Doyle 2006), as well as ecosystem processes (Alexander et al. 2000). Nutrient dynamics are sensitive to a stream's fluvial geomorphology (Valett et al. 1994, Doyle and Stanley 2006), because as materials move across a landscape or through a stream, certain locations or "hotspots" account for a disproportionate amount of nutrient removal or processing, and these hotspots are linked to geomorphic structure (Peterjohn and Correll 1984, McClain et al. 2003).

The spatial arrangement of and connections among hotspots influence how materials move between patches, the relative availability of materials to different patches, and the export of materials to adjacent ecosystems. The origin, development, and persistence of these patches as well as their interactions are largely under geomorphic control (McAuliffe 1994). At large scales, work on lake districts (Webster et al. 2000) and lake chains (Kling et al. 2000) has demonstrated that lake chemistry is affected both by the strength of hydrologic linkages and by the type of flow paths between lakes (i.e., whether they are connected via surface streams or subsurface flows). At smaller scales, research on desert streams has shown that nitrogen (N) cycling is affected by vertical hydrologic exchange between interstitial hyporheic flows and surface waters, which link subsurface patches with high nitrification rates to surface patches with high algal growth and high N-uptake rates (Holmes et al. 1994, Valett et al. 1994). Indeed, it is becoming increasingly clear that stream ecosystems can be understood only in light of their geomorphology and hydrology (Doyle and Stanley 2006).

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The link between fluvial geomorphology and stream biogeochemistry is important to researchers' understanding of not only comparatively pristine streams but anthropogenically altered watersheds as well. Humans have been altering stream ecosystems at least since the invention of irrigated agriculture. Modern distribution systems rely on dams to store water during wet periods so that water can be delivered during drier times. Dams alter the hydrologic connectivity of the watershed, starving lower watersheds of the sediments needed to maintain deltas and estuaries (Vörösmarty and Sahagian 2000), preventing flood pulses that link floodplains and wetlands to riverine systems, and isolating upper portions of watersheds by establishing barriers to the migration of fish and other aquatic organisms (Pringle 2001). As the hydrologic connectivity of river networks declines, so too does their ability to provide ecosystem goods and services such as abundant clean water and historic patterns of biodiversity (Wilson and Carpenter 1999). The emerging science of urban ecology (Grimm and Redman 2004, Grimm et al. 2008) grapples with the impacts—many of which are unintended—of humans on fluvial systems. In this article, we focus on how altering the hydrologic connectivity of urban streams influences their biogeochemistry.

As cities grow, land is recontoured, vegetation is planted or removed, road networks are built, and buildings are erected. Landscape architects create novel landscapes and plant assemblages that are intended to influence ecosystem conditions; in effect, they produce "designer ecosystems" that may or may not be based on historical views of the location's ecology (Palmer et al. 2004). In arid and semiarid ecosystems, these designed, engineered ecosystems are often characterized by a mesic horticulture with elevated rates of primary production (Kaye et al. 2006) and an accompanying high water demand. To meet the demand for water and other resources, urban ecosystems import materials from ecosystems well beyond their physical boundaries (Baker et al. 2001, Jenerette et al. 2006). Streams also are modified to deal with excess flows during storms.

Storm-water management traditionally meant simplifying and straightening stream channels to improve their ability to convey flows from the city while armoring their banks to reduce erosion (Walsh et al. 2005). More recently, some "soft" storm-water management strategies have been implemented. For example, many communities now use retention basins to hold floodwater on site and promote groundwater recharge (Larson et al. 2005). How these changes in geomorphology and hydrology affect stream ecosystems has been the subject of much recent research (e.g., Groffman and Crawford 2003, Grimm et al. 2005). The term "urban stream syndrome" has been coined to describe the consistent suite of symptoms—including a flashier hydrograph, elevated nutrient and contaminant concentrations, altered channel morphology and stability, and reduced biotic richness—that characterize urban streams (Meyer et al. 2005). Less thought, however, has been given to the unintended impacts of alterations in hydrologic connectivity resulting from the infrastructure (e.g., Postel 2000) required to bring water to and move it through cities. Thus, three critical ecological questions are (1) How do water distribution systems affect the spatial and temporal connectivity of urban streams? (2) How do the changes in hydrologic connectivity resulting from these systems interact with other modifications resulting from urbanization to affect the ecological function of urban streams? and (3) Which of these changes result from deliberate human action, and which are unintended consequences of hydrologic alterations?

Here we take a case-study approach to answering these questions, exploring how urbanization and the associated changes in hydrology and geomorphology have affected a desert stream. We focus on Indian Bend Wash (IBW), an urbanizing watershed located within the study area of the Central Arizona–Phoenix Long-Term Ecological Research (CAP LTER) program (hereafter, the CAP ecosystem) (Grimm and Redman 2004). Although this example is particular to the Sonoran Desert, it highlights how modifications of fluvial systems can be especially pronounced in semiarid regions. In desert regions of the United States, the human population—consisting largely of immigrants from other US regions and Mexico—has perceptions of streams that often do not match the desert stream phenomenon. Even in arid regions where people historically were more cognizant of desert stream structure and function, modern water scarcity may dictate massive modifications of these ecosystems (Postel 2000). Thus, we argue that this case study holds lessons about the likely effects of urbanization on streams and rivers throughout the arid and semiarid world, regions that may well experience disproportionate urbanization in the future (UNEP 2006).

Because urbanization is a spatially structured process, supplementing aggregate measures of land-use change with a more detailed description of the pattern of urban growth can yield new insights into the effects of urbanization on ecosystem function (Alberti 2005). Because N is often limiting in Sonoran Desert streams (Grimm and Fisher 1986), we focused on how urbanization has affected N dynamics in the IBW. This article presents (a) a spatially explicit description of how urban development proceeded in the IBW watershed; (b) a narrative description of the important changes in geomorphology and hydrology resulting from urbanization; and (c) a discussion of the unintended consequences these modifications have had on the fluvial ecosystems of the IBW, paying particular attention to changes in the N cycle.

Phoenix rising: Growth of a desert city

Cities of the CAP ecosystem are located in the northern Sonoran Desert's Valley of the Sun, near the confluence of the Salt and Gila rivers (figure 1) in the Basin and Range Lowlands hydrogeologic province of Arizona (Montgomery and Harshbarger 1989). This province is characterized by deep, gently sloping basins filled with alluvial material eroded from the surrounding mountains (Arrowsmith and Pewe 1999). The IBW is an ephemeral tributary of the Salt River (figure 1) that drains 520 square kilometers (km^2) of Scottsdale (founded in

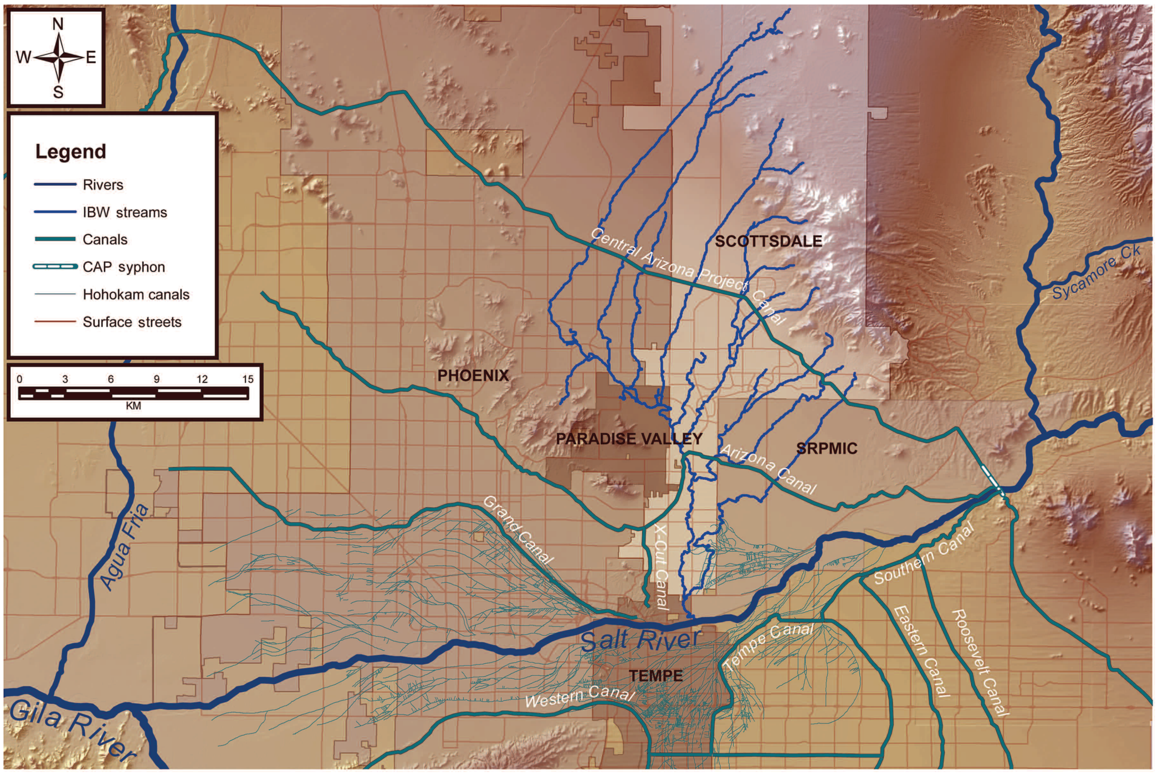


Figure 1. Shaded relief map of the Central Arizona–Phoenix (CAP) ecosystem and its key rivers and canals. Fine blue lines running through Scottsdale indicate the historic channel network of Indian Bend Wash (IBW). The major canals of the modern canal network are indicated with thick green lines; the fine green lines branching out to the west from the Salt River indicate the extensive historic Hohokam canal network. Arterial streets are provided to suggest the current extent of urbanization. Abbreviation: SRPMIC, Salt River Pima Maricopa Indian Community.

1888, incorporated in 1951), Paradise Valley (incorporated in 1961), and a corner of the Salt River Pima Maricopa Indian Community (established in 1879). Most of the watershed is atop unconsolidated sediments derived from the McDowell Mountains on its northeastern border (Arrowsmith and Pewe 1999). These sediments are the principal component of the regional aquifer system and store an estimated 200,000 cubic meters (m^3) of water in the top 360 m of the Salt River Valley aquifer (Montgomery and Harshbarger 1989).

Bringing water to the valley of the sun. Historically, storm runoff flowed across the watershed in nearly parallel channels that originated in the McDowell Mountains and trended southwest toward the Phoenix Mountains (figure 1). These rills emptied into the main stem, which curved around the Phoenix Mountains and continued to its confluence with the Salt River (figure 1). The sediments of the low-gradient valley floor were reworked during episodic floods, producing narrow low-flow channels subsumed within broader active channels, a configuration typical of southwestern streams (e.g., Fisher 1986). Because these sediments were coarse, hydrologic exchange between surface and subsurface would have been rapid and frequent. Early observations (Lee 1905) and more recent data (ADWR 1999) show that the water table was always below the channel, indicating that the IBW was a losing stream that flowed only during floods.

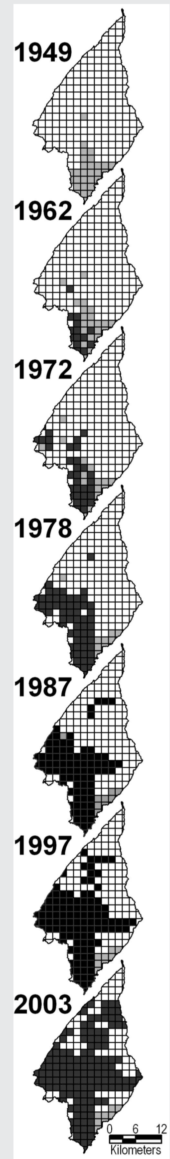
The Sonoran Desert experiences hot summers and mild winters, with two distinct rainy seasons. Because the growing season is so dry, agriculture in the region relies on irrigation. The first irrigation canals in the CAP ecosystem were built by the Hohokam, who inhabited the region for nearly 1800 years, beginning around 300 BC (Bayman 2001). Between approximately AD 600 and AD 1450, they constructed extensive canal systems on several rivers in Arizona, including the Salt and Gila rivers (figure 1; Bayman 2001). Farming returned to the valley in the 1860s with construction of Swilling's ditch, the region's first modern canal (Luckingham 1989). The Arizona Canal was completed in 1885 and was followed by numerous additional canals that brought nearly 101,000 hectares under irrigation by 1910 (Luckingham 1989). Between 1911 and 1946, six dams were constructed along the hydrologically variable Salt and Verde rivers to ensure a more reliable supply of irrigation water. The most ambitious water project was the Central Arizona Project canal, which was constructed between 1973 and 1993 and now brings an additional 1.7×10^9 m^3 per year of surface water into the valley (USBR 2005). Groundwater has been an important but declining source of water in the region, dropping from 47% of water use in 1985 to 39% in 1995 (ADWR 1999).

Nevertheless, between 1900 and 1990, some portions of the watershed saw the groundwater table drop nearly 90 m as a result of extensive pumping (ADWR 1999).

Box 1. Tracking land-use change in the desert.

The series of simplified, classified watershed maps in the figure details the rapid urbanization of Indian Bend Wash (IBW), Arizona. Pristine desert dominated the watershed through the 1960s, but in the early 1970s, urban areas began to expand rapidly, first replacing farms and later expanding into virgin desert. As with other portions of Phoenix (Gober and Burns 2002), expansion was rapid, with much of the construction occurring on the urban fringe. Farms west of the Salt River Pima Maricopa Indian Community, or SRPMIC (the remaining agrarian parcel in the Southeast), were quickly converted to urban land use, resulting in a sharp boundary between the dense housing developments of Scottsdale and the remaining agricultural fields. Urban land use migrated east along the northern border of the SRPMIC and north to the Central Arizona Project canal. Then, in the late 1980s, urbanization jumped beyond the canal and moved rapidly north through the watershed.

Watershed-scale changes in land use were tracked using historic aerial photography from seven years (1949, 1962, 1972, 1978, 1987, 1997, and 2003). With the exception of the 1949 layer, which only covered the southern two-thirds of the watershed, each layer covered 95% to 100% of the watershed. Arterial roads in the IBW watershed run north-south or east-west and are spaced approximately one mile (1.6 kilometers [km]) apart. As urbanization of the watershed proceeded, development filled in square-mile (2.6-km^2) plots bordered by these roads. We documented the progress of this development by overlaying a grid of 1 × 1 mile (1.6 × 1.6 km) cells atop the watershed with cell borders generally aligned with the main roads and clipped to the watershed's extent. Land cover in each cell for each year was visually examined and designated as desert, agriculture, or urban, depending on which cover class covered the greatest area. Desert areas showed no sign of modern development. Agricultural areas comprised cultivated or fallow fields and associated buildings. Urban areas contained any of the following: residential developments, commercial properties, irrigated urban parks, golf courses, or other built structures, including roads. We assumed that if a cell was predominantly desert in one year, it was also desert in previous years. Likewise, once a cell was converted to urban use, it was assumed to remain urban. Using this logic, we were able to classify all the unphotographed cells, including those from 1949.



Land-use change in Indian Bend Wash. Diagrams show land-use patterns from 1949 through 2003. The dominant land cover in each cell is indicated by the cell's hue: white (desert), light gray (agricultural fields), or dark gray (urban land use).

Urban expansion. Like the mythical bird, the cities of greater Phoenix rose from the ashes of the Hohokam villages and farmsteads that preceded them. Indeed, many modern irrigation canals were re-excavated from old Hohokam structures. Unlike the structures created by that earlier civilization, however, these new urban centers were built with transportation in mind. When the city of Scottsdale was founded in 1888, it was platted on a square-mile (2.6-km^2) grid (Gammage 1999), and as the region grew, this initial pattern was repeated. This choice was made in part because building on the grid was well suited to the automobile, and in part because the expense of bringing services—especially water—to the surrounding desert meant that entire, uniform subdivisions were cheaper and more profitable to build than less-structured urban forms (Gammage 1999). Nonetheless, the urbanization of the IBW began slowly. As late as 1962, 84% of the IBW remained undeveloped desert, with a few farms south of the Arizona Canal the only significant land use (figure 2). With the arrival of cheap central air conditioning in the late 1950s, however, the regional population began to explode (Gammage 1999). By the early 1970s, subdivisions had begun proliferating between the Arizona Canal and the future Central Arizona Project canal (box 1). Urbanization followed a wavelike pattern during periods of peak development (Gober and Burns 2002), with subdivisions replacing first agricultural fields and later pristine desert as the fringe migrated northward (box 1). The result was a relatively uniform urban expanse, which covered 65% of the watershed by 2003 (box 1, figure 2).

The altered hydrology and geomorphology of Indian Bend Wash

The extensive canal network of the CAP ecosystem not only enabled its explosive growth but also dramatically altered the hydrology and geomorphology of fluvial ecosystems.

Rerouting water through the desert. Unlike the rivers that feed them, canals tend to run parallel to topographic contour lines rather than cutting across them. Because of this, canals disrupt historic flowpaths by severing surface flows, thus reducing the hydrologic connectivity (Pringle 2001) of the underlying stream network. Completion of the Arizona canal isolated the lower 48 km^2 of the IBW watershed from the northern 471 km^2 (figures 1, 2), restricting surface flows into

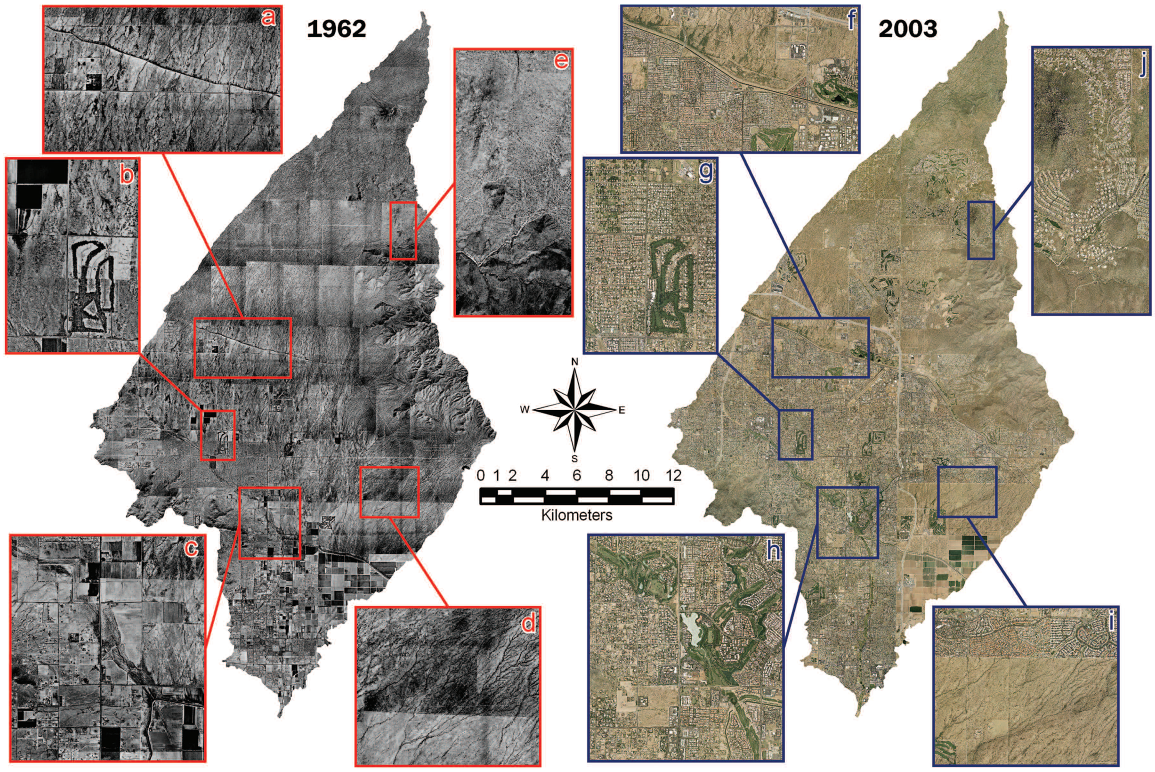


Figure 2. Aerial photographs comparing development in the Indian Bend Wash (IBW) in 1962 (black-and-white) and 2003 (color), with exploded views of key features. In 1962, with the exception of the remains of the failed Old Verde canal (a) and two golf courses (e.g., b), one of which can still be seen in 2003 (g), virtually the entire region north of the Arizona Canal remained desert (d, e). The IBW's main stem was still clearly visible as a thin strip of vegetation (c). By 2003, urban development had migrated past the Central Arizona Project canal (f) and into the northern portion of the watershed (j); the IBW flood-control greenbelt encased the main channel of the wash (h); and most subparallel channels of the upper portion of the wash, which were so conspicuous in 1962 (d), were restricted to a narrow strip of desert just north of the Salt River Pima Maricopa Indian Community (i). Dark bands running north-south and east-west in the 1962 image are artifacts of the original photographs. Photographs: US Geological Survey (1962) and Landiscor (2003).

the southern basin to the intersection of the IBW with the Arizona Canal. Because floods periodically overtopped the canal at this juncture, a siphon was later constructed that allowed the canal flow to safely cross beneath the wash (figure 3). Completion of the Central Arizona Project canal further subdivided the watershed. In part to ensure the integrity of the canal and in part to provide further flood control, this canal was engineered so that overland flows from upslope areas collect in large retention basins along its northern bank (Matthews 1985), severing the northern 241 km^2 from the remainder of the watershed (figures 1, 2). Together, these two canals have subdivided the watershed into three hydrologically distinct basins, reducing the land area that contributes to runoff.

As urban development continued, so did the region's mercurial relationship with water (Larson et al. 2005). As subdivisions replaced farms and desert, new construction began to encroach on the floodplain of the IBW (figure 3). Riparian vegetation was removed, floodplain risers recontoured, and the active floodplain subdivided (figure 3). One of the most dramatic geomorphic modifications of the systems geomorphology was the radical reworking of the confluence of the IBW and the Salt River. These changes resulted in a pronounced increase in flood risk. Although nearly dry most of the year, the IBW remained prone to periodic flash floods, and many residents suffered greatly during these rare events (Matthews 1985). Scottsdale attempted to reduce the flood risk, by passing a 1964 ordinance limiting development on floodplain land (Matthews 1985) and subsequently by working with the Maricopa County Flood Control District and the US Army Corps of Engineers to design a storm-water system to convey the 850-m^3-per-second 100-year flood through the city to the Salt River (figure 3). Rather than build a cement-lined chute, engineers developed a solution that relied on a 12-km greenbelt, with streams and artificial lakes sitting within a broad, protected floodplain, to shunt water through the city (figures 2, 3). Thus was built one of the United States' first "nonstructural" flood-management systems.

Creating novel geomorphic features. Historically, because flow through the IBW was ephemeral, there was little to no standing water in either the main stem or the larger watershed. This changed as numerous lakes were built throughout the

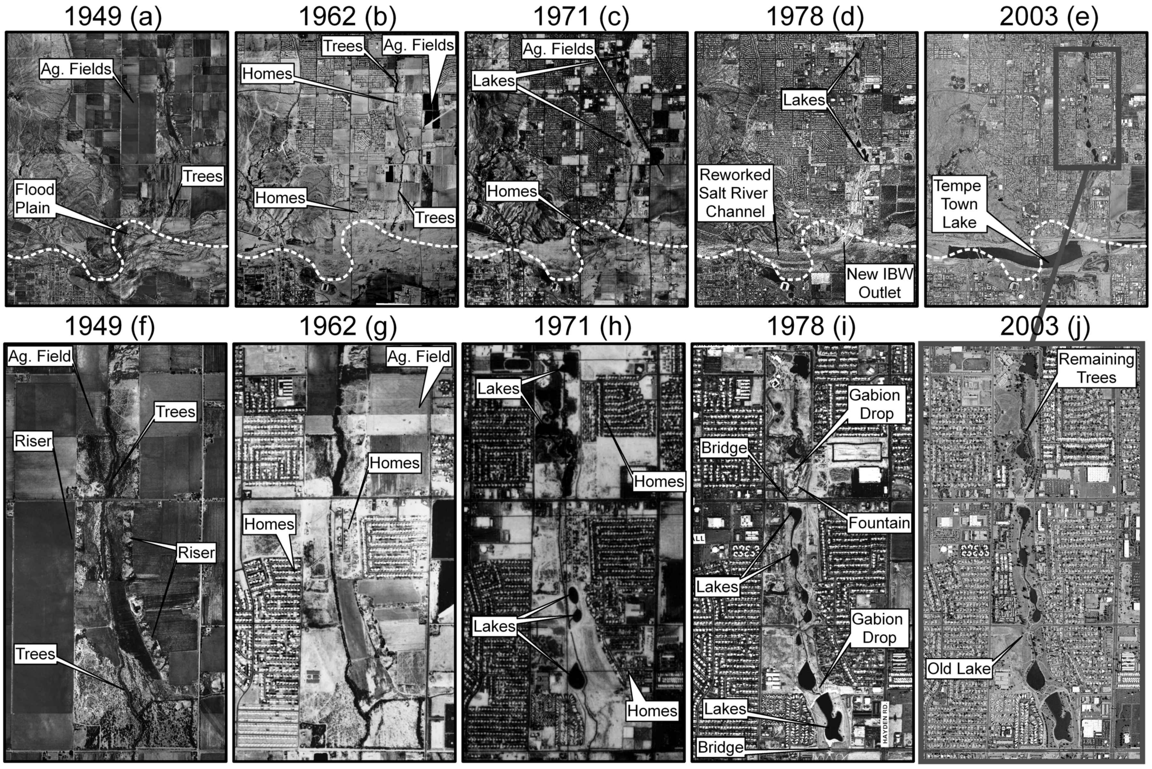


Figure 3. Aerial photographs documenting broad (top panels) and detailed (lower panels) changes in the lower Indian Bend Wash (IBW). The top panels (a–e) show changes in the confluence of the IBW and the Salt River resulting from development. In 1949, a large meander (dotted line) was a prominent feature downstream from the confluence, while numerous trees dotted the broad, braided floodplain of the Salt River. This bend was lost as new housing developments replaced farmland and encroached on the floodplains of the two rivers (b, c). As development proceeded, the Salt River channel was further straightened and deepened to contain large floods (d) and then reworked again to accommodate Tempe Town Lake (e). Lower panels (f–j) provide a detailed view of modifications made during the construction of the greenbelt. In 1949, the IBW's channel consisted of a broad floodplain bounded by steep risers, which graded into the adjacent agricultural fields (f). By 1962, the conversion of farmland had begun (g). When floodplain construction was banned in 1964, numerous homes already sat in the floodplain (g, h). Lake construction began when El Dorado Park was established in 1966 in the northern end of the reach. The park included two of the first artificial lakes (h). Seven more lakes were added to the reach between 1966 and 1978 (i), while one was removed (j). During construction of the greenbelt, numerous flood-prone homes were removed (h–i), but not before a sizable flood caused extensive damage and displaced numerous residents. In 1977, the lower 3.2 kilometers of the wash was reworked to create an outlet channel (d). The greenbelt stabilized sediments with irrigated turf, protected traffic with bridges, and reduced stream power through a variety of energy-dissipation structures, including riprap and gabion drops as well as the artificial lakes (i). Photographs: Flood Control District of Maricopa County (1949), US Geological Survey (1962, 1971), and Landiscor (1978, 2003).

watershed as amenities to urban parks, as hazards on golf courses, as accent features for new subdivisions, and as features of the IBW greenbelt (figure 3). Indeed, the number of lakes grew steadily as urban land use expanded. Thus, although there were only 5 lakes found in the IBW in 1949, and only 30 lakes in 1973, 176 lakes with a combined surface area of more than 1.4 km^2 dotted the watershed by 2003. More than 40 of these lakes were established along the main stem of the IBW alone, where, in addition to their value as recreational amenities, they store irrigation water and add surface roughness to the channel (USACE 1975). Lake water levels are maintained with canal water, groundwater, or reclaimed wastewater. For example, the Salt River Project diverts a mixture of groundwater and surface water from the Arizona Canal through a series of lateral canals (figure 4) to maintain lake levels in the lower IBW greenbelt (figure 4, inset). The relative contributions of these various water sources can significantly affect surface water chemistry, perhaps most importantly by increasing inorganic N concentration (box 2).

Urbanization's intended and unintended consequences

People are continuously altering the urban environment, making cities the quintessential designer ecosystems (Palmer et al. 2004). Although some of these changes help to ensure that the city can provide ecosystem goods and services (An-

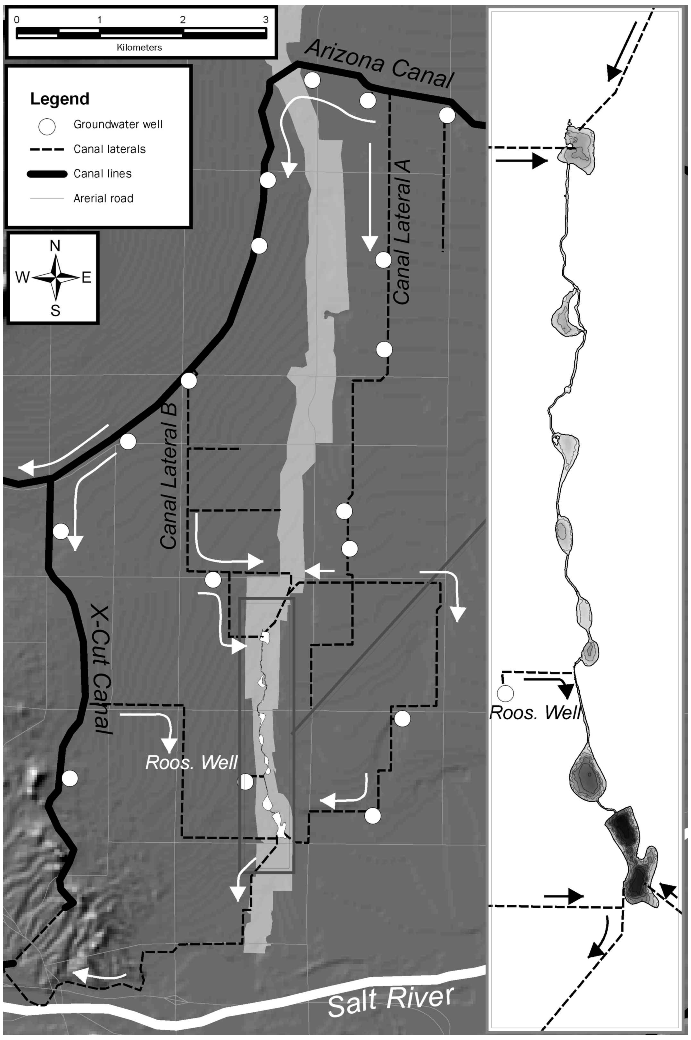


Figure 4. Simplified map of canals, canal laterals, and groundwater wells feeding eight lakes in the lower Indian Bend Wash watershed (inset). Surface water from the Salt, Verde, and Colorado rivers flows through the Arizona Canal and is supplemented with groundwater from the wells along its southern edge. Two canal laterals (A and B) deliver water to the stream above the reach as well as directly to lakes in the lower portion of the wash (inset). The Roosevelt (Roos.) well discharges directly to the stream. During baseflow, water is returned to the canal system through a drain in the bottom lake. Arrows indicate dominant flowpaths. Locations of lateral canals and wells are taken from the Salt River Project Zanjero Map.

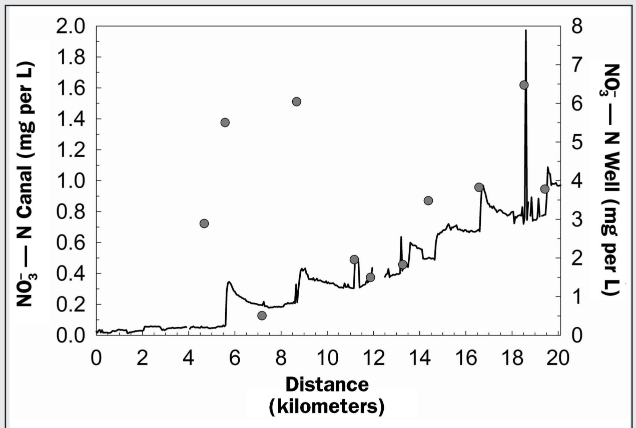
dersson 2006), others have unintended impacts on the ecology of the city and thus on its ability to provide these critical functions. In the cities of the CAP ecosystem, many of these modifications redistribute water for the purposes of flood control and irrigation. These changes allow farmers to grow crops and homeowners to cultivate landscapes that previously were not seen in the desert. Large floods are now efficiently shunted through the system, and smaller floods are diverted into retention basins to promote groundwater recharge. Artificial lakes provide a variety of goods and services, including temporary water storage, nutrient removal, fishing, and boating. The resulting urban oasis is as much the fruit of human imagination and ingenuity as it is of the desert's underlying ecology. Yet while many of the changes in ecosystem structure and function were the intended result of these efforts, many others were the unintended consequences of radically altering the hydrology of the desert.

Unintended consequences of redistributing water. Unlike river networks, which continually concentrate flows into higher-order channels, canals produce hydrologic systems that resemble trees. Water is collected from the uplands through the dendritic river network (roots), is diverted across vast expanses of land via arterial canals (trunks), and finally is redistributed across the landscape via dendritic lateral canals (branches). Thus, the rivers of central Arizona no longer continuously concentrate flows and deliver their dissolved and suspended loads to the Colorado River, but instead deliver them to the farms, parks, lakes, and homes of the CAP ecosystem (the metaphorical tree's leaves). Although such distributary flowpaths are common features of deltas and, in some cases, of floodplains, the extent, stability, and inland location of these flowpaths in the CAP LTER represent a profound change in the hydrologic connections that characterize this arid ecosystem.These changes in the branching pattern of the drainage network may have important consequences for ecosystem function. Recent studies have argued that the shape and size of drainage basins as well as the structure and distribution of confluences within them can affect ecosystem processes such as nutrient retention (see Fisher et al. 2007). Similarly, the branching structure of distributary canal networks will influence their ecosystem function.

Because aquatic habitats (Seitzinger 1988) and their interfaces with terrestrial habitats (Peterjohn and Correll 1984) are frequently hotspots for N transformations, the redistribution of water through canals for irrigation has important impacts on nutrient cycling. By making water widely available, irrigation systems increase contact between river water and sediment (and soils) and thus tend to increase nutrient processing rates. Moreover, in typical desert stream ecosystems, terrestrial-aquatic interactions, while particularly important for determining stream-water chemistry, are episodic and restricted primarily to storm events (Grimm et al. 2005, Welter et al. 2005). Because processes like nitrification and denitrification frequently are stimulated when soils are wetted, the CAP ecosystem's canal system, by increasing the temporal availability of water, increases N-processing rates (e.g., Zhu et al. 2004, Roach 2005). Even in mesic ecosystems where flows are more predictable and interactions between stream water and stream sediments are less temporally variable, irrigation is still likely to increase contact between stream water and upland soils, thus ensuring that hotspots are provided a steady supply of potentially rate-limiting materials. Further, because irrigation can and often does continue year-round, nutrient processing in the uplands of the IBW is not as tightly coupled

Box 2. Canal nitrogen availability.

The relative proportions of Salt River water, Verde River water, Central Arizona Project water, and groundwater flowing through the Central Arizona–Phoenix ecosystem's network of canals change on both a seasonal and an annual basis, producing spatial and temporal variation in solute concentrations (Sullivan 1996, 1998). One of the most pervasive changes in surface water–groundwater interactions in Indian Bend Wash (IBW) is linkage of the Salt River aquifer with surface flows via groundwater pumping. We examined how this modification affects nitrate (NO_3^-) concentration in surface flows by conducting an extensive longitudinal survey along a 20-kilometer (km) section of the Southern/Tempe canal. Although not in the IBW watershed, this section is managed in much the same fashion as the Arizona Canal. Water from the Salt River, the Gila River, and the Central Arizona Project canal is diverted into the Arizona and Southern/Tempe canals at Granite Reef Dam, 6.4 km downstream of the Salt and Verde rivers confluence (figure 1). Canals are cement lined to prevent seepage, and their flows are augmented by groundwater pumping during periods of high demand.



Downstream changes in nitrate (NO_3^-) in canal water collected from the Southern and Tempe canals on 21 June 1999. Solid line represents the NO_3^- concentration in canal water collected every 50 meters. Circles indicate the concentration of NO_3^- in water from actively pumping groundwater wells. Note that groundwater and canal water NO_3^- concentrations are plotted on different axes in both graphs. Abbreviations: L, liter; mg, milligram; N, nitrogen.

A synoptic survey conducted on 21 June 1999 showed that NO_3^- in the canals increased tenfold, from less than 0.1 milligram per liter to nearly 1.0 mg per L as water flowed downstream (see the figure). The increase in NO_3^- concentrations was not linear; instead, a series of abrupt spikes was associated with the input of NO_3^--rich groundwater, followed by gradual declines (see the figure). During the survey, 13 sampling teams collected 803 water samples during a three-hour period. Additional grab samples were taken from outflows of 11 groundwater wells pumping into the canal. Water samples were transferred to acid-washed, prerinsed, 60-milliliter bottles and placed on ice for transport back to the laboratory, where they were immediately filtered through Whatman GF/F filters. Within 24 hours of collection, samples were analyzed for NO_3^--N by colorimetric analysis after reduction to nitrite on a Bran-Luebbe TrAAcs 800 autoanalyzer.

to precipitation events as it is in the uplands of the surrounding desert (Welter et al. 2005).

The creation of the numerous artificial lakes in the IBW has also had important unintended impacts by increasing the abundance of hotspots and thus changing rates of nutrient cycling. Shallow lakes have characteristics that favor denitrification (Seitzinger 1988), a major process associated with N removal. Research conducted over the past four years in IBW lakes (figure 4) shows that these urban lakes are hotspots of N removal, as indicated by high potential denitrification rates of sediments (4.7 ± 0.4 milligrams [mg] N per kilogram [kg] per hour; mean ± 1 standard error; Roach 2005). The mean mass-specific potential denitrification rates observed in the surrounding floodplain soils (1.5 ± 0.1 mg N_2O-N per kg per hour; mean ± 1 standard error; Roach 2005), although significantly lower than those observed in the lakes, were similar to values from riparian soils in the more mesic Baltimore ecosystem (Groffman and Crawford 2003) and substantially higher than previously published rates of urban soils in the CAP ecosystem (Zhu et al. 2004), indicating that floodplain soils are seasonally important N sinks.

Unintended consequences of altered sediment dynamics. Imperviousness typically increases as mesic landscapes are urbanized, which, in turn, produces flashier hydrographs and increasing discharges (Leopold 1968). In the arid and semiarid West, however, the effects of urbanization are not so predictable. Roofs, parking lots, and roads are impervious surfaces that increase runoff, but some changes may have the opposite effect. When compared with hardpan desert soils, the abundant irrigated vegetation of urban landscapes tends to increase soil permeability and decrease runoff (Abrahams et al. 1995). The urbanization of the IBW has been accompanied by a simultaneous proliferation in impervious surfaces, such as rooftops and parking lots, and by the expansion of grassy landscapes through the establishment of irrigated turf. Thus, although average permeability may not have decreased as the IBW urbanized, we nevertheless hypothesize that by establishing new patches that differ markedly from desert soils, urbanization has increased spatial variation in permeability and thus has altered how water flows across the landscape.

As changes in imperviousness have occurred, low- and middle-order stream channels also have been lost through the direct conversion of small rills and tributaries to turf or impervious surfaces (see also figure 2d, 2i). Together with the increase in impervious area resulting from the construction of roofs and roads (Leopold 1968) and with the establishment of irrigated turf, these changes have combined to anchor sediments in place, reducing their flux into the main stem of the IBW. Reduced sediment loads alter reach-scale geomorphology because sediment-starved streams experience greater channel erosion and incision (Trimble 1997). Our observations of the IBW main stem suggest that downcutting has occurred along some stream segments (to more than 1 m in the lower reaches). Engineered features of the flood-control project, such as channel straightening, exacerbate the effects of reduced stream flows. Because irrigated turf has stabilized the coarse sediments of the greenbelt, it both reduces sediment flux and further limits channel migration. Consequently, the low-flow channel has become narrow, straight, and entrenched by downcutting, while artificial lakes have been filling with fine organic sediments (to a depth of more than 50 centimeters in some lakes). These changes reduce interactions between surface and subsurface water in the simplified channel and thus reduce nutrient processing rates within the fluvial portion of the system (Grimm et al. 2005); however, the creation of lakes and irrigated floodplains establishes new hotspots for N transformation.

The effects of land-use change on sediment dynamics have been exacerbated by the construction of the region's extensive canal infrastructure. As mentioned previously, the CAP and Arizona canals divide the IBW watershed into three distinct basins. The effects of the reduced hydrologic connectivity have been pronounced. The Army Corps of Engineers estimated that the CAP canal reduced peak flood discharges through the greenbelt by 37% (Lee 1988). This substantial loss of stream power has reduced the ability of floodwaters to transport and rework channel sediments. In addition, the Arizona Canal greatly restricts, and the CAP canal essentially eliminates, fluxes of materials (i.e., sediments and dissolved nutrients) between upper and lower subbasins. In some ways, this is akin to the effects of dams on large rivers, which dampen flood peaks and rob rivers of their sediment loads, causing predictable impacts on downstream ecosystems (Junk et al. 1989). Thus, the IBW now not only receives less sediment input from the uplands but also has a reduced capacity to move the sediments that do reach its main stem. Because geomorphology is a major determinant of the location of biogeochemical hotspots (McClain et al. 2003, Welter et al. 2005, Fisher et al. 2007), changes that alter the fluvial geomorphic processes responsible for channel maintenance can affect ecological processes (Poff et al. 2006) generally and nutrient cycling specifically (Orr et al. 2006). Indeed, unlike the distribution in Sycamore Creek, a typical and well-studied Sonoran Desert stream with a similar watershed area (505 km^2), the distribution of hotspots within the main stem of the IBW is no longer as much a function of its fluvial geomorphology (Valett et al. 1994, Fisher et al. 2007) as it is a function of where lakes have been excavated and how well the floodplain is connected via irrigation.

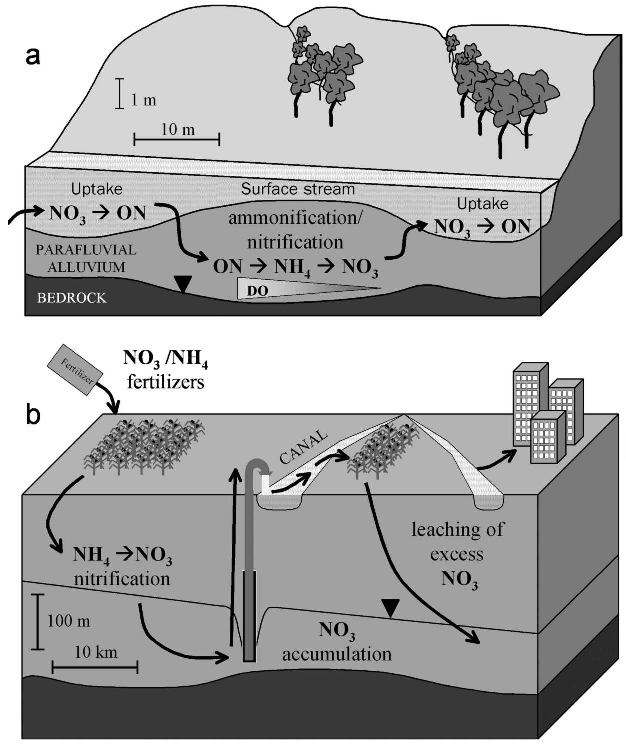


Figure 5. Conceptual model of how surface-subsurface interactions determine nitrate (NO_3) availability in Sycamore Creek (a) and the Indian Bend Wash (IBW) ecosystem (b). The processes of assimilation, ammonification-nitrification, and hydrologic exchange determine nitrogen (N) dynamics in both systems, but the scales at which these processes operate differ markedly. Cycling of N in Sycamore Creek is rapid and tightly coupled to biotic drivers, and it occurs over short spatial scales. Conversely, N cycling in the IBW is slow and dominated by human inputs, and it is characterized by long lag times between the leaching of N into groundwater and its return to surface ecosystems. Further, groundwater pumping and canals establish spatial connections between distant, formerly isolated patches in the IBW. Finally, drivers of hydrologic exchange in the two ecosystems are different, with physical processes forcing movement of water between surface and subsurface systems in Sycamore Creek, whereas human actions connect groundwater with surface flows in the IBW. Abbreviations: DO, dissolved oxygen; NH_4, ammonium; ON, organic N.

Unintended introduction of lags in time and space. Because the IBW was historically ephemeral, surface-subsurface exchanges were very likely limited to episodic discharge events. However, the relatively broad floodplain visible in 1945 suggests that when the wash flowed, surface-subsurface exchange was extensive. Although portions of the IBW are now perennial, surface-subsurface exchanges have been limited by simplifying the stream segments and lining the lakes with clay. To understand how this may affect N availability, we contrast the processes responsible for determining N concentrations in the IBW with those of Sycamore Creek.

In Sycamore Creek (figure 5a), nutrient cycling is strongly affected by interactions between surface and subsurface patches, which are largely a function of geomorphology (Dent et al. 2001). Sediments of variable thickness (i.e., the hyporheic zone) are underlain by a bedrock layer. Recharge zones occur where sediments are relatively deep, while discharge zones occur where shallow bedrock forces subsurface water to the surface. When surface waters enter the hyporheic zone, they carry high concentrations of dissolved oxygen and dissolved organic matter derived from primary production in the surface stream, creating ideal conditions for coupled mineralization and nitrification reactions (Holmes et al. 1994). When this nitrate (NO_3^-)-rich water returns to the surface in upwelling zones, primary producers use it to fuel further growth (Valett et al. 1994). Thus, although NO_3^- concentrations are often high in upwelling zones, algal uptake causes a longitudinal decline in surface water NO_3^- concentration as the water flows downstream (Valett et al. 1994).

Nitrogen cycling in the IBW is analogous to that in Sycamore Creek (figure 5b), but it operates at different spatial and temporal scales and responds to different mechanisms. Nitrate concentration in the groundwater below the IBW, like that below the Southern/Tempe canal, is substantially elevated (Roach 2005) because of leaching from fertilized agricultural fields (Xu et al. 2007). When groundwater is used to supplement canal flows or to maintain urban lake levels, the NO_3^- concentration in surface waters is increased (box 2). Although the effect of hydrologic exchange between groundwater and surface water in the IBW is analogous to the exchanges between parafluvial and surface flows in Sycamore Creek, the spatial and temporal scales at which the interactions occur are much greater. The parafluvial sediments of Sycamore Creek are shallow (approximately 1 to 2 m) compared with the depth to groundwater under the IBW (approximately 20 to 100 m). In addition, hydrologic exchange is relatively rapid in Sycamore Creek, with hyporheic residence time estimated to be on the order of hours to days (Dent et al. 2007). This contrasts sharply with the temporal scale of the interaction between surface and groundwater in the IBW ecosystem. Even ignoring the time originally required to fill the Salt River aquifers, the time required for irrigation water to percolate into the groundwater and be returned to the surface via groundwater pumping is considerable (years to decades). The time required to complete this cycle implies a lag between shifts in agricultural management practices and changes in groundwater chemistry, reducing the ability of managers to limit the flux of N into the IBW. Finally, the mechanisms producing groundwater–surface water exchange in Sycamore Creek are physical (e.g., positive vertical hydraulic gradients), whereas in the IBW, mixing of groundwater and surface streams can effectively occur only as a result of human action. Clearly, this large-scale redistribution of water across the desert creates new linkages between sources of N (groundwater) and hotspots of nutrient processing. Nevertheless, how these large-scale changes in surface–subsurface interactions interact with small-scale changes in the distribution and characteristics of hotspots remains largely an open question.

Independent of how altered hydrology affects nutrient processing, increases in N are potentially important in themselves. Nitrogen is the limiting nutrient in many streams of the Southwest (Grimm and Fisher 1986), and increased N loading to streams in many biomes is associated with reduced capacity to remove and process the nutrient (Mulholland et al. 2008). Further, because the hydrology of the CAP ecosystem has been engineered to minimize hydrologic loss, much of the N entering the system is retained (Baker et al. 2001) and, through groundwater pumping, is repeatedly made available to its aquatic and irrigated ecosystems.

Conclusions

Integrating humans into ecological studies remains a central challenge of ecology (Pickett et al. 1997, Grimm et al. 2000, Alberti et al. 2003). Evidence from the IBW supports the thesis that the ecological effects of urbanization on streams often are produced by land-use change and propagated by hydrologic alterations. In the desert, perhaps the most profound impact of urbanization is the reconfiguration of surface hydrology. Human decisions and actions produce engineered systems that differ dramatically from their pristine predecessors. In the IBW, we focused on how hydrologic changes such as artificial lakes, canal systems, and groundwater pumping have altered the fluvial geomorphic processes responsible for maintaining channel form and created unintended impacts on nutrient cycling. Although we do not emphasize them here, it is important to note that other ecological functions also change when the hydrologic connectivity of fluvial ecosystems is lost or changed.

Because people engineer their surroundings to provide the amenities they desire, urbanization often introduces novel features, such as perennial water bodies in arid and semiarid ecosystems. People enjoy living by water and, as Dubai's Palm Island and Tempe Town Lake (figure 3) make clear, will go to great lengths to create beachfront or lakefront property. These artificial water bodies may be designed for reasons as diverse as provision of recreational opportunities, water storage, or flood control, but their impact is rarely limited to their intended use. In the IBW, artificial lakes are sediment traps that have become hotspots of N cycling. In other cities, artificial lakes may help to spread exotic species or attract unwanted insects, such as mosquitoes. Clearly, the unintended consequences of establishing perennial water bodies must be carefully evaluated—especially in arid and semiarid cities, where these impacts may be striking—if managers are to have any hope of mitigating them.

Canals divert and transport water through and between river basins (Postel 2000). Invariably, many of these canals will cut across flowpaths and reduce hydrologic connectivity. By severing flowpaths, canals can limit the upstream migration of fish or the downstream drift of invertebrates. That dams and diversions sever a river's connection with the sea, potentially starving coastal ecosystems of important nutrients, is well recognized (Nilsson et al. 2005), but their impact on upland tributaries has been less well understood. And while canals may be considered archetypal features of desert landscapes, their impacts are not restricted to arid and semiarid ecosystems. Even in the boggy areas of Louisiana, canals have been shown to alter wetland hydrology by reducing sediment flux (Turner 1997). Unfortunately, while canals may be as important as dams in reducing hydrologic connectivity, the extent of their impact on fluvial systems is not known. Future assessments of their impact must account for their ability to affect the flow of water, materials, and biota across a landscape as well as for their ability to deliver water to points of scarcity.

Our work also underscores the importance of time lags in anthropogenically influenced ecosystems. Groundwater wells have established a new link between surface and subsurface flows. This link not only increases the spatial and temporal availability of water but has the unintended effect of increasing the flux of NO_3^- through urban waterways by returning N leached from historic fertilizer applications to surface flows. It is likely that other impacts of urbanization are lagged in time and may be felt only years or decades after the original event.

Many of the observed changes in hydrology were a byproduct of efforts to design an ecosystem that conforms to human residents' vision. Patterns of development in the Phoenix area broadly, and in Scottsdale specifically, resulted from a series of human decisions, both by individual developers choosing where to build and by larger institutions deciding how to provide ecosystem services such as flood control (e.g., the City of Scottsdale and the US Army Corps of Engineers) and water for cities and crops (e.g., the Salt River Project). Many of these decisions were guided by conflicting needs for flood control and a reliable water supply. Efforts to establish reliable irrigation were largely successful and have created the illusion of unlimited water in this desert ecosystem (Gammage 1999), a theme that can be seen repeated in many other arid and semiarid cities, including Denver, Las Vegas, and Los Angeles. Increased spatial and temporal availability of water in the IBW enhances primary production, while tending to anchor sediments in place and alter biogeochemical cycling. Lakes not only serve as hotspots for N processing but also have become attractors for waterfowl, changing the distribution of birds within the CAP ecosystem (Servoss et al. 2000). Clearly, deserts are ecosystems where fundamental processes are dramatically altered by the simple addition of water.

Previous urban researchers have noted that the consequences of decisions about how and where a city develops are often "phase lagged" by a decade or more (Alberti 2005). An example from our case study is that consequences of past fertilizer use in the IBW are just now being realized as groundwater pumping returns agricultural NO_3^- to the surface. Such delays have been observed in other ecosystems as well. For example, efforts to control N loading to the Baltic Sea from the Norrström drainage basin are hindered by contributions from slow groundwater pathways and by the accumulation of N in subsurface pools (Baresel and Destouni 2005, Xu et al. 2007). As more and more agricultural lands are converted to urban uses (e.g., del Mar López et al. 2001), unexpected consequences resulting from historical legacies are likely to become more common and will present challenges to managers as they attempt to maintain ecological functions in cities.

A key lesson from this and other studies of urban ecosystems is that as cities grow, the ability of native ecosystems to provide their historic goods and services is often bartered so that the city can provide a different set of ecosystem goods and services. While some of these trade-offs may be intended—for example, trading one set of species for another—others are unintended. These unintended consequences of urbanization extend beyond fluvial ecosystems and will remain a challenge for managers who hope to maintain the ecological integrity of native ecosystems or simply to ensure the continued delivery of important goods and services by these ecosystems.

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