Quenching Urban Thirst: Growing Cities and Their Impacts on Freshwater Ecosystems

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The development of water resources to satisfy urban water needs has had serious impacts on freshwater ecosystem integrity and on valuable ecosystem services, but positive trends are emerging that point the way toward a solution. We demonstrate this through case studies of water resource development in and around five large urban areas: Los Angeles, Phoenix, New York, San Antonio, and Atlanta. Providing freshwater ecosystems with the water flows necessary to sustain their health, while meeting the other challenges of urban water management, will require greatly increased water productivity in conjunction with improvements in the degree to which planning and management take ecosystem needs into account. There is great potential for improvement in both these areas, but ultimately water planners will also need to set limits on human alterations to river flows in many basins in order to spur greater water productivity and protect ecosystem water allocations before water supplies become overtaxed.

Keywords: biodiversity, freshwater ecosystems, hydrologic alteration, sustainable water development, water resources development

As we enter the 21st century, the management of water resources is emerging as one of humanity's most significant challenges. Population growth, economic development, and expansion of irrigated agriculture led to dramatic increases in water use during the 20th century (Gleick 2000), and the populations of most countries will continue to grow in coming decades. Between 2000 and 2050, the US population is projected to increase from 275 million to 403 million (USCB 2000), while the global population is expected to increase from 6 billion to almost 9 billion (UN 2003). This population growth has serious implications for food and energy production and urban expansion, all of which will place increasing pressure on available fresh water supplies.

The seriousness of the water management challenge is underscored by the fact that signs of water scarcity are already appearing around the globe. It is estimated that 41% of the world's population lives in river basins where the per capita water supply is so low that disruptive shortages could occur frequently. If current consumption patterns continue, that percentage will grow as the global population increases further (Revenga et al. 2000). In the United States, water managers in 36 states anticipate that they will face local, regional, or statewide water shortages some time during the next 10 years. Some of the nation's highest population growth rates are projected for western states where water is already in short supply (GAO 2003).

The difficulties in satisfying human demands for water are compounded by the implications of water development for the planet's freshwater ecosystems. Human impacts on freshwater ecosystems intensified in the past century because of modifications such as dams and water diversion structures, which have become ubiquitous around the world (Revenga et al. 2000). The available scientific evidence suggests that freshwater biodiversity is in crisis (Postel and Richter 2003). Aquatic species extinctions and population declines associated with water development have been well documented in the United States and many other developed countries, a story that is likely to be repeated in less-developed countries undergoing rapid changes in land and water use in association with growing populations (Postel and Richter 2003). As freshwater species and ecosystems are degraded, human society is losing a wealth of ecosystem services provided by healthy freshwater and estuarine ecosystems worldwide (Daily 1997).

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Figure 1. Water sources for Los Angeles, California.

Current trends indicate that urban water demand is likely to place increasing pressure on available water supplies in the future. Withdrawals for public water supply in the United States increased steadily from 1980 to 2000 at about the same rate as population growth, even as water withdrawals in the irrigation and industrial sectors leveled off (Hutson et al. 2004). Worldwide, the percentage of the population living in cities is projected to grow from 47% to 60% by 2030 (UN 2003). In this article, we focus on the ecological impacts associated with the development of water resources to support urban growth in five large American cities, using these impacts as a means of describing the potential ecological degradation that could result from future development of additional urban water supplies. We conclude with a discussion of ways to minimize or prevent this ecological damage through more efficient use of water and through the implementation of less damaging strategies for storing and diverting water for cities.

Our case studies describe the history of water resource development in and around five urban areas: Los Angeles, Phoenix, New York, San Antonio, and Atlanta. We feel justified in making some general points from our five case studies because water development in these areas has been typical of the 20th-century water development paradigm, which has focused on construction of physical infrastructure such as dams and aqueducts, relied on unsustainable use of groundwater resources, and included little attention to ecological values (Gleick 2000, Revenga et al. 2000).

Although some hopeful trends in water use have emerged in the past few decades, the probabilities of lessening the ecological impacts of future water development remain uncertain at best. One notable trend is a general slowing in the rate of increase in water withdrawals, including a leveling of total water demand in the United States since 1980 (Gleick 2000). The global rate of large dam construction has also decreased in recent decades (Gleick 2003). Perhaps most important, managers and policymakers appear to be placing an increasing emphasis on water conservation and ecosystem protection (Gleick 2000).

Our case studies reflect these trends. But water projects with considerable potential to damage freshwater ecosystems are still being proposed, such as India's efforts to link all 37 of its major rivers (Thakkar 2003), Spain's plan to build 120 dams in the Ebro River basin (Aguirre 2003), and China's South-North Water Transfer project, which proposes to transfer water from the Yangtze River to three other river basins (Shen 2003). An alternative approach would take advantage of the fact that there is much room for improvement in terms of reducing future water demands, reducing losses in conveying and storing water, shifting water from lower-value and more wasteful uses to higher-value and more efficient uses, and considering ecosystem needs in water planning and management.

Case studies

We selected five large metropolitan areas in the United States to characterize the recent history of water resource development and its impacts on natural ecosystem functions and native biota. The case studies span the country and encompass different types of natural systems (rivers, aquifers, and estuaries). All five case studies discuss historical and planned future water resource development.

Los Angeles. Los Angeles' pursuit of water supplies during the 20th century is a classic case of a growing urban area outstripping its local supplies and going outside its river basin to acquire more. In 1900, Los Angeles obtained all of its water from the Los Angeles River. But population growth, much of it as in-migration from other parts of the country, caused the city's needs to exceed this local water supply early in the 20th century. The import of water from other river basins enabled the Los Angeles area to grow at a phenomenal rate throughout much of the 20th century (Hundley 2001).

The city's search for water led it first to the Owens Valley and then to the Mono Basin, each located a few hundred kilometers (km) northeast of the city (figure 1). An aqueduct from the Owens Valley to the city was completed in 1913. This system was augmented in later years with a series of storage reservoirs, an extension of the aqueduct northward to the Mono Basin in 1940, and an expansion of aqueduct capacity in 1970. During this same period, the southern California region also began to import water through the Metropolitan

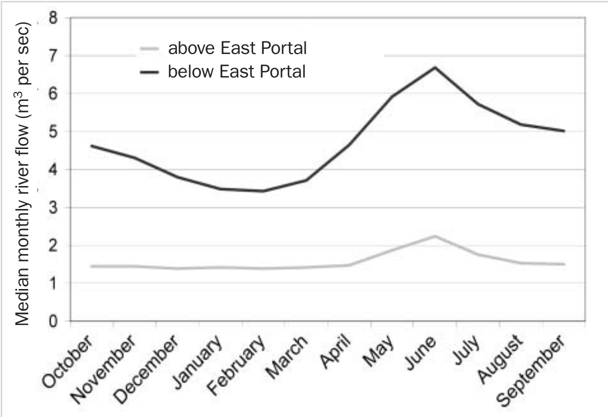

Figure 2. Median monthly flows in the upper Owens River, California, above and below the East Portal, where water being transferred from the Mono Basin enters the Owens River, 1940–1989. Flows have been drastically increased during all months of the year. Data are from CSWRCB (1994).

Figure 3. Water sources for Phoenix, Arizona.

Water District (MWD) of California, whose main sources were the Colorado River and the rivers of the Central Valley (figure 1). Water from these sources began to arrive in 1941 and 1971, respectively.

The ecological impacts of this expanding water-supply system have been serious and widespread. By the mid-20th century, the natural Los Angeles River ecosystem had become severely degraded by a combination of agricultural and municipal water use, water pollution, and flood control structures that included concrete lining of the river channel (Gumprecht 1999). While Los Angeles is only one of many users of MWD water, the sources of that water are also heavily impacted by water supply infrastructure. The great decline in native fish populations that has occurred in the lower Colorado River is due in part to the altered flow conditions created by dams and reservoirs (Mueller and Marsh 2002). Reduced freshwater inflows have seriously degraded the wetlands and once-productive fisheries in the Colorado River delta and estuary (Postel et al. 1998). Water-use projects in the Central Valley have severely affected the Sacramento–San Joaquin delta and upstream freshwater ecosystems, contributing to a great decline in salmon populations (CALFED Bay–Delta Program 2000).

The severe impacts of Los Angeles's transbasin water diversions on freshwater ecosystems in the Owens Valley and Mono Basin are also well documented. The natural systems in both of these areas were subjected to some impacts before the arrival of Los Angeles' water supply infrastructure, including grazing, stocking of nonnative trout, diversions for irrigation, and small hydroelectric power facilities (CSWRCB 1994). Dewatering (drying) of streams due to water use did occur in some areas during droughts (CSWRCB 1994). Before 1913, the impacts of dewatering were probably most severe in the lower Owens River, where irrigation diversions reduced and altered the seasonal patterns of flow (Brothers 1984). But despite these impacts, aquatic habitat and riparian vegetation throughout the Owens River and Mono Basin were generally intact and similar to natural conditions before the impact of Los Angeles (CSWRCB 1994). The city's water-supply system greatly exacerbated any existing ecological problems, to the point of devastating some areas.

The historical Owens River was sustained by snowmelt from the Sierra Nevada, terminating in the saline Owens Lake. In 1913, the Los Angeles Aqueduct began pulling water from the river 100 km upstream of the lake. Below the aqueduct, all flow was eliminated except during extremely wet years (Brothers 1984). By the late 1920s, Owens Lake was dry (LADWP 2002). Beginning in the 1940s, the Owens River north of the aqueduct was used to transport water from the Mono Basin, and this led to increased median and peak flows in most of the river's length above the aqueduct (figure 2; Brothers 1984, CSWRCB 1994). The operations of the water-supply system also altered the seasonal patterns of flow, with the natural summer flow peak replaced by a longer period of high flows from spring through fall (Brothers 1984). The greatly augmented flows caused severe alterations in river habitat, including changes in temperature and sediment regimes and accelerated rates of bank erosion below the dams (CSWRCB 1994).

Native fish populations and riparian vegetation have been severely affected in the Owens River. Of the four native fish species, two are now federally listed endangered species, and another is a California species of special concern. Flow alteration from water diversions and impoundments, in combination with competition from nonnative species, has led to major declines in the historical ranges and abundances of these three species (USFWS 1998). The Owens pupfish was believed to be extinct from 1942 to 1964, when a small population was rediscovered (USFWS 1998). Almost all native riparian

Table 1. Water-supply sources of Los Angeles, 1980 and 1990–2000, in millions of cubic meters.

Source	1980 water supply[a]	1990–2000 water supply
Los Angeles Aqueduct (from Owens River and Mono Basin)	620	370
Metropolitan Water District (from Colorado River, Central Valley rivers, and other sources)	25	260
Local groundwater	110	110
Total	755	740

a. The 1980 measurements are representative of the distribution of water supplies before 1987.
Source: LADWP (2000).

vegetation disappeared from the lower Owens River, and flow alteration also adversely affected vegetation in the upper section of the river above the aqueduct (Brothers 1984, Stromberg and Patten 1992).

The Mono Basin has been similarly affected. This closed basin consists of a saline lake fed by creeks draining the Sierra Nevada, and the basin is one of California's richest natural areas. Mono Lake has no fish, but it is vital habitat for hundreds of thousands of migrating and nesting birds, which feed on the brine shrimp and flies that inhabit the lake (Wiens et al. 1993). Since 1941, water diversions have subjected the downstream portions of the creeks flowing into Mono Lake to prolonged dewatering, resulting in the elimination of riparian vegetation and the widening and incising of channels during floods (CSWRCB 1994). By the 1980s, decreases in water inflows had cut the volume of the lake in half and doubled its salinity. Lowered lake levels provided predators with greater access to bird nests and may have significantly affected the reproduction and survival of the birds' primary food sources (Wiens et al. 1993).

The epilogue to the stories of Owens River and Mono Lake is that environmental litigation has led to a reversal of at least some of the damage (Hundley 2001). When the Los Angeles Aqueduct's capacity was expanded in 1970, it necessitated increased water withdrawals from the Mono Basin and groundwater pumping in the lower Owens Valley, changes that accentuated the environmental impacts described above. This triggered a series of lawsuits throughout the 1970s and 1980s, which forced Los Angeles to agree to restore flows and wildlife habitat in the lower Owens River, mitigate dust storms from the lake bed of the dry Owens Lake, and limit exports from the Mono Basin so that the lake's elevation could return to more natural levels.

These events marked the beginning of a transition in Los Angeles's water-supply sources and water demand (table 1). The lost supplies from the Owens River and Mono Basin were largely replaced by water from MWD sources, and the city also began to emphasize water conservation. Following a lengthy drought from 1987 through 1992, Los Angeles began to invest seriously in reducing water demand; as a result, per capita water usage decreased by 15% between 1985 and 2000. The city's population is projected to continue to grow from 3.8 million to 4.8 million people between 2000 and 2020. Future increases in demand will be met by combining increased MWD water use with water conservation and recycling (LADWP 2000).

Phoenix, Arizona. Water resource development in and around Phoenix during the first half of the 20th century was driven by the needs of the agricultural sector, but since then, population growth and urban development have also become important factors (Kupel 2003). Table 2 shows current water supplies and demand in the Phoenix area. This area has a long history of water development, starting with the Hohokam Indians, who irrigated extensive parts of the Salt River Valley from AD 300 to 1450. However, the impacts of humans on the desert rivers of Arizona were far greater in the 19th and especially the 20th century (Whittlesey 1997).

Construction of irrigation canals in the Phoenix area began in the 1860s, but the highly variable flow of the Salt and Verde Rivers made irrigation difficult and led to damaging floods. To meet the needs of the growing agricultural sector, Roosevelt Dam was completed on the Salt River in 1911. This spurred further agricultural development in the valley. By 1930 three additional dams were built on the Salt River downstream of the Roosevelt Dam, and by 1946 two more dams were built on the Verde River, which joins the Salt River just upstream of Phoenix (figure 3). These dams were all constructed under the auspices of the Salt River Project (SRP), and water for irrigation was withdrawn from the Salt River at the Granite Reef Diversion Dam, just downstream of the confluence of the Salt and Verde Rivers.

Through this period, the city of Phoenix was a minor user of water compared with the agricultural areas around it. But droughts and a growing population during the 1940s caused the city's water needs to exceed its available supply, and in 1952 an agreement was reached to transfer water rights for parcels converted from agricultural to municipal use, using the SRP's existing canals and ditches to distribute the water. Transfer of water rights in this manner satisfied the city's needs until the 1980s, when the Central Arizona Project (CAP) was completed to bring water from the Colorado River (figure 3). This project involved an aqueduct and a storage reservoir on the Aqua

Table 2. Water sources for the Phoenix, Arizona, active management area, 1985 and 1995, in millions of cubic meters.

Water supply	Agriculture	Municipal	Industrial	Total
1985				
Local surface water	1008	456	12	1476
Local groundwater	953	342	78	1372
Central Arizona Project (water from Colorado River)	0	0	0	0
Effluent	39	14	1	54
Total	2001	812	90	2902
1995				
Local surface water	744	486	9	1240
Local groundwater	707	313	88	1108
Central Arizona Project (water from Colorado River)	149	187	2	339
Effluent	44	86	4	136
Total	1645	1073	102	2821

Note: Totals are not exact because of rounding.
Source: ADWR (1999).

Fria River (northwest of Phoenix), built on the site of an earlier small irrigation storage dam.

This extensive dam building and water use had devastating impacts on the water flow regime, geomorphology, vegetation, and fauna of the Salt River ecosystem. Below Granite Reef Dam, the river's flow dwindled until the mid-1940s, at which time the river was almost totally desiccated (Miller 1961). Dam operations and changes in sediment transport have led to down-cutting in the channel in areas where sandbars and islands were common before 1940 (Graf 2000). The dams on the Salt River have substantially modified the river's natural flow regime (figure 4; Fenner et al. 1985, Tellman et al. 1997). Native riparian vegetation has been eliminated or reduced by dewatering, by alteration of the natural flooding regime, and by a decline in the water table caused by groundwater pumping (Fenner et al. 1985, Tellman et al. 1997, Graf 2000).

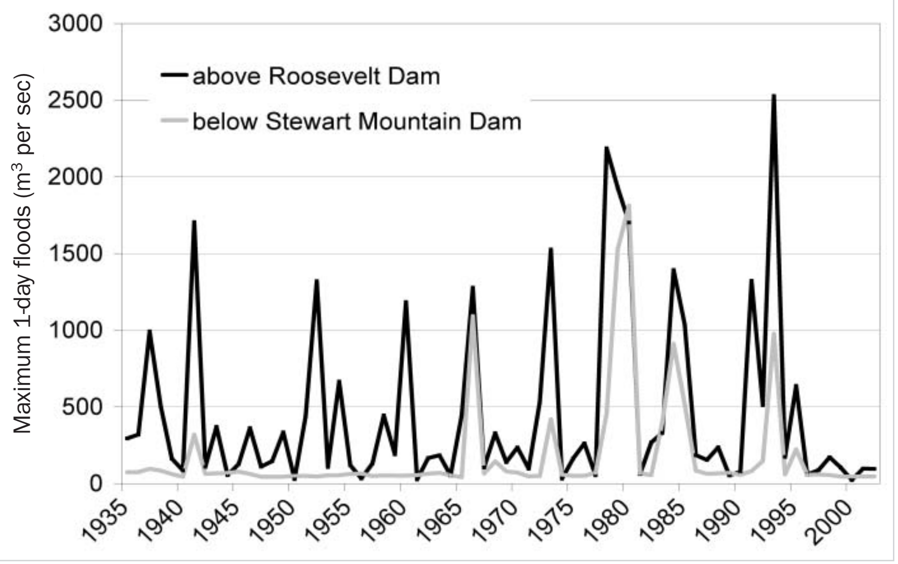

Figure 4. Maximum 1-day floods on the Salt River, above the uppermost dam (Roosevelt Dam) and below the lowest storage reservoir (Stewart Mountain Dam) near Phoenix, Arizona. Reduction or elimination of natural floods has created serious problems for native cottonwood forests in the Salt River Valley. Data are from US Geological Survey stream gauging stations 09498500 and 09502000.

These changes in ecosystem processes and conditions also contributed to the extirpation of the native fish fauna. By the mid-1920s, half of the 14 species native to the river were gone below the Granite Reef Dam, and by the mid-1940s, three at most were left (Miller 1961, Minckley and Deacon 1968). In the vicinity of the Roosevelt Dam, no native species remained in 1950 (Miller 1961). A review of US Fish and Wildlife Service Recovery Plans (USFWS-AEFSO) indicates that eight federally endangered or threatened fish species have been extirpated from these sections of the Salt River. Impoundments, altered flow regime, and dewatering are identified as major causes of this decline in native fish populations (Miller 1961, Minckley and Deacon 1968).

Rivers other than the Salt River have also been affected. Although the Verde River still flows perennially, the dams have altered the natural hydrology of the river (Whittlesey 1997). Since 1927, the Agua Fria River below the irrigation dam has flowed only in wet years or in response to flood events (Tellman et al. 1997). The flow of the Gila River below its con-

Table 3. Water sources for New York City, in millions of cubic meters per year.

System	Safe water yield during drought	Maximum water withdrawal
Croton River watershed	264	331
Catskill creeks	518	661
Upper Delaware River tributaries	639	826
Total	1421	1818

Source: Endreny (2002).

fluence with the Salt River is dominated by effluent from wastewater treatment plants (Tellman et al. 1997). A substantial proportion of the native fish fauna has been extirpated from all of these rivers, and dams and associated water use are one of the major causes (USFWS-AEFSO). Finally, water use in the Phoenix area contributes to the poor condition of the Colorado River ecosystem, which was described in the Los Angeles case study.

Future water-supply planning in the Phoenix area will need to provide water for a population that is projected to grow by 50% from 2005 to 2025, at which time municipal water demand is projected to surpass agricultural demand. Of great concern to water planners is the fact that groundwater use is significantly overdrafting the region's aquifers. Future needs will be met with a combination of increases in CAP water, recycling of effluent, and water conservation (ADWR 1999).

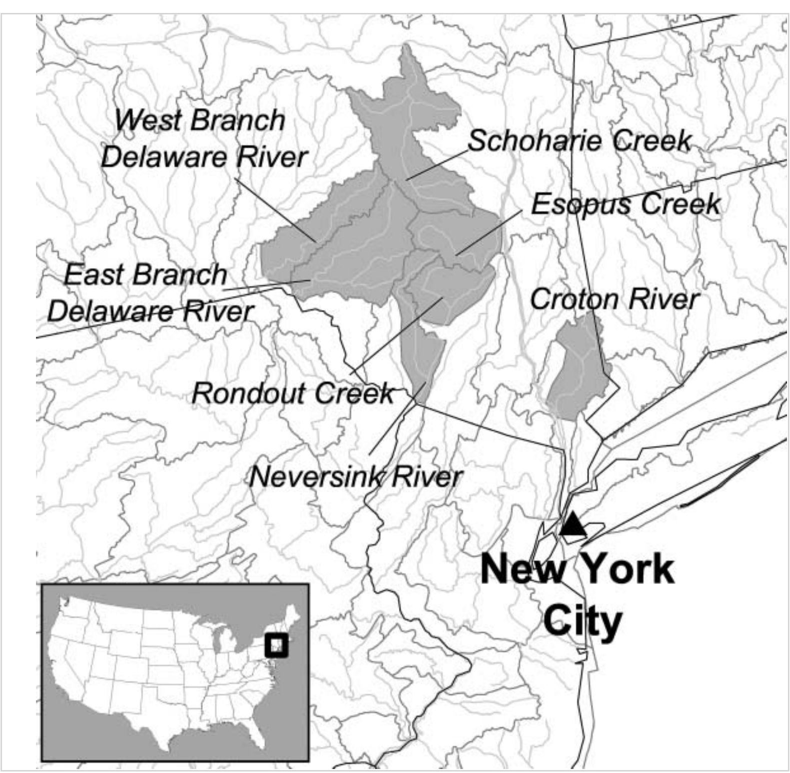

Figure 5. Water sources for New York City. The upper Delaware system includes the three upper Delaware tributaries (East and West Branches and Neversink River) and Rondout Creek, which is a tributary to the Hudson River. Rondout Creek is used to transfer water from the upper Delaware on its way to New York.

New York City. The city of New York has the most extensive water-supply system in the eastern United States, extending more than 150 km to the northwest into the Catskill Mountains and the upper tributaries of the Delaware River (table 3, figure 5). Expansion of this water-supply system was essential to accommodate the large population increases and accompanying economic growth that occurred during the 19th to mid-20th centuries (Galusha 1999). New York relied on local supplies of water until 1842, when an aqueduct was completed to bring water from the Croton River, a watershed some 75 km north of Manhattan (figure 5).

The Croton system, continually expanded through 1911, eventually consisted of 12 reservoirs and 3 controlled lakes. When this supply proved inadequate, two large reservoirs and an aqueduct were constructed between 1907 and 1927 to transfer water from two creeks in the Catskill Mountains, in the Hudson River basin. The final (and largest) piece of the New York system involved the construction between 1937 and 1965 of four more large reservoirs and aqueducts to transfer water from the three upper tributaries of the Delaware River. There are no plans to expand the system further, because the local population has stabilized and because the city's water conservation programs have been effective. Per capita water use in New York City dropped by 29% from 1990 to 2002 (NYCDEP 2004).

Water projects in New York have had serious impacts on river flows and biota, although the documentation of these impacts is not as detailed or complete as in some of our other case studies. The Croton River is a case in point. Little documentation of ecological impacts in the Croton River is available, but given the density of dams in this relatively small watershed, it seems likely that the flows have been altered greatly. Impacts are better documented in the Catskill creeks (Schoharie and Esopus Creeks). Water is transferred from a reservoir on Schoharie Creek south to New York City (the creek flows north); this has cut the creek in half, changed its flow pattern, and altered it from a cold-water to a warm-water fishery. The creek is dewatered for 15 to 25 km downstream of the reservoir (SRC 2002). As for Esopus Creek, a federal court recently fined New York City for violating the Clean Water Act by discharging excess

Table 4. Alteration of mean, high, low, and summer flows in the upper Delaware tributaries, in cubic meters per second.

	Mean annual flow	7-day maximum flow	7-day minimum flow	Mean flow June–October
East Branch, Delaware River				
1942–1953	20.9	115.4	1.55	7.98
1954–2002	5.97	51.7	0.20	5.03
Percentage change	–72	–55	–87	–37
West Branch, Delaware River				
1953–1963	21.9	126.8	1.20	8.88
1964–2002	16.2	84.8	0.49	16.3
Percentage change	–26	–33	–59	+84
Neversink River				
1942–1952	7.53	39.3	0.97	3.77
1953–2002	1.51	9.2	0.16	1.89
Percentage change	–80	–77	–83	–50

Note: Flows in all three rivers have been drastically reduced, with the exception of summer flows in the West Branch, which have increased because water is released from the reservoir to meet minimum low-flow requirements farther downstream in the Delaware River. Dates of dam construction are 1953 (Neversink), 1954 (East Branch), and 1964 (West Branch). Data are from USGS stream-gauging stations 01417000 (East Branch), 01425000 (West Branch), and 01436000 (Neversink).

sediment into the creek through its water-supply tunnel (NYWEA 2003).

The most widespread impacts are found in the upper Delaware tributaries, the East and West Branches of the Delaware River and the Neversink River. The natural flow regime of all these rivers has been significantly altered (table 4, figure 6). Researchers have also attributed some specific biological impacts to the dams, although a lack of preimpoundment data makes some of these conclusions necessarily speculative. The remnant American shad run on the Delaware River, once the largest shad run in the country, uses spawning areas near the confluence of the East and West Branches, not too far below the dams. The dam on the West Branch had an immediate adverse impact on shad spawning in the 1960s, due to the intolerance of shad for cold-water dam releases, so these important spawning areas may be of precarious suitability (Chittenden 1976). Efforts to meet downstream flow targets have resulted in rapid flow alterations in the West Branch that cause shad strandings and may also affect shad spawning and out-migration (Colin Apse, The Nature Conservancy, New Paltz, NY, personal communication, September 2003).

Donnelly (1993) hypothesized that populations of benthic invertebrates on the Delaware River are scarce because of a coarsening of the river bed below the dams and increased summer flows on the West Branch. Harman (1974) found the Delaware River to be almost devoid of mussels; he attributed this to fluctuations in water levels and reduced temperature below the dams. Data collected by Strayer and Ralley (1991) and Baldigo and colleagues (2002) indicate that mussels were less abundant and diverse below the dam on the Neversink River, although Baldigo and colleagues (2002) admit that it is uncertain whether the net effect of the dam on mussels is positive or negative. They do speculate that although stabilization of flows from Neversink Dam may have helped a few mussel species flourish, this has probably occurred at the expense of native biodiversity throughout the river system, because the dams disrupt native fish species' movements and alter natural ecosystem processes.

San Antonio, Texas. The San Antonio area's sole source of water has historically been the Edwards Aquifer (figure 7). Withdrawals from the aquifer quadrupled from the early 1930s to the 1980s (Hamilton et al. 2003), with municipal water use accounting for more than half of current usage

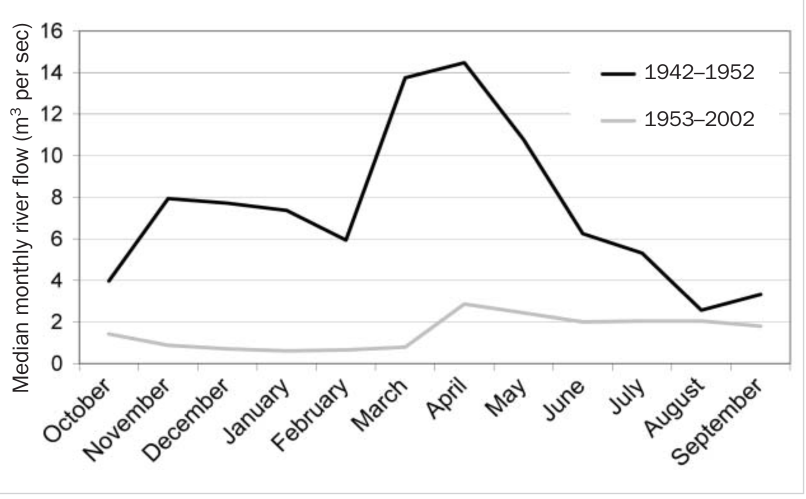

Figure 6. Median monthly flows on the Neversink River before and after construction of the Neversink Reservoir (1953). Water withdrawals and dam operations have drastically reduced and altered the monthly pattern of flows. Data are from USGS stream-gauging station 01436000.

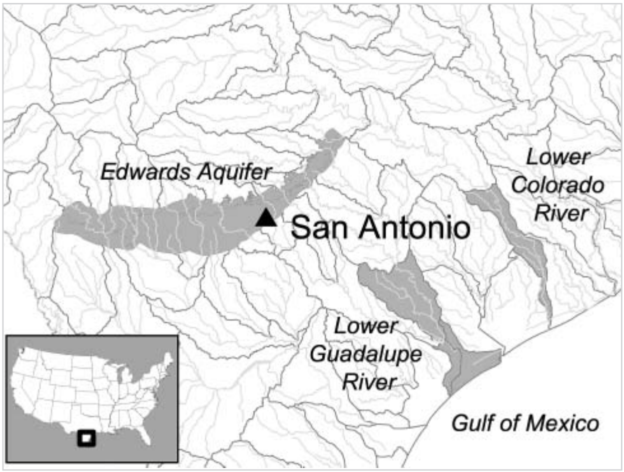

Figure 7. The current water source (Edwards Aquifer) and two planned water sources (the Guadalupe and Colorado Rivers) for San Antonio, Texas. The map shows the recharge and artesian zones of the San Antonio segment of the Edwards Aquifer.

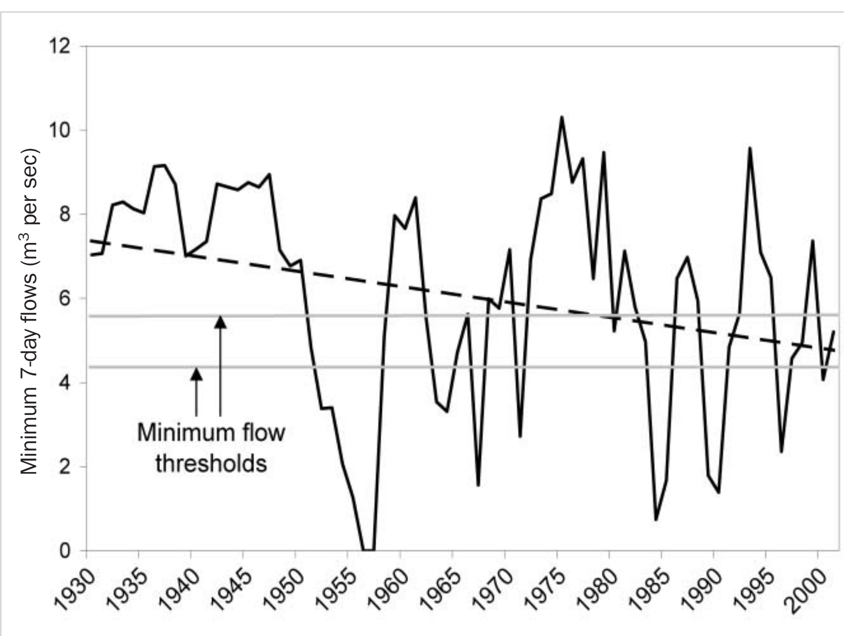

Figure 8. Annual minimum 7-day flows at Comal Springs, near San Antonio, Texas, appear to be declining over time (see dashed linear trend line). Before the severe drought of the early 1950s, recorded flows at Comal Springs never dropped below the minimum flow thresholds (the two horizontal lines) suggested by the US Fish and Wildlife Service (USFWS 1995) as necessary to protect the endangered fountain darter. However, in more than half of the years in recent decades, minimum flows have dropped below these thresholds. The two minimum flow thresholds specify the flows below which further water withdrawals would constitute activities that "take" (5.66 cubic meters per second) or "jeopardize" (4.25 cubic meters per second) this listed species under the Endangered Species Act. Data are from USGS stream-gauging station 08169000.

and agricultural withdrawals for the rest (USFWS 1995). Over the last 20 years, well withdrawals have averaged about half of aquifer recharge (Hamilton et al. 2003), with severe environmental consequences for native species that depend on the aquifer. There are currently eight federally endangered and threatened aquatic species for which a primary threat is loss of spring flows caused by groundwater pumping (USFWS 1995).

The situation in Comal Springs is indicative of the effects of groundwater pumping on Edwards Aquifer fauna. Comal Springs, the largest spring fed by the aquifer, is home to the endangered fountain darter and many other endemic species (USFWS 1995). Most of its flow moves through an aquifer zone directly under Bexar County, where San Antonio is located. Although the Comal Springs flow remains perennial, cycles of significantly decreased flow have increased in frequency over the past 30 to 40 years, and the effects of droughts have become more severe (Crowe and Sharp 1997), threatening the fountain darter (figure 8). Reduced flows affect the darter by reducing available habitat and encouraging the proliferation of invasive snails, which consume aquatic plants that provide cover for the darter. In 1990 it was estimated that if pumping continued to increase at historic rates and another multiyear drought occurred like the one in the 1950s, the spring would go dry for years (USFWS 1995).

Fortunately, recent events indicate that these trends are reversing. In the early 1990s, a number of lawsuits forced Texas to pass a bill establishing the Edwards Aquifer Authority to manage the aquifer. The bill mandated that well withdrawals be restricted to 493 million cubic meters (m^3) per year after 2007, well below the 1989 peak of 669 million m^3 (EAA 1998, Hamilton et al. 2003). A drought-management plan has been established that will further restrict water use when aquifer levels drop below critical levels (EAA 1998). Another positive development is San Antonio's strong commitment to water conservation. The city reduced per capita water use by 24% between 1984 and 2000 (Eckhardt 2004).

While management of the Edwards Aquifer seems to be headed in the right direction, San Antonio has plans for a number of future projects to meet the needs of

Table 5. Source and amount of water used in Atlanta, Georgia, and 10 surrounding counties, 2000.

Source	Water use (millions of cubic meters)
Chattahoochee River (including Lake Lanier) and tributaries	427
Etowah River (including Lake Allatoona) and tributaries	78
Ocmulgee River and tributaries	31
Flint River and tributaries	21
Groundwater	9
Total	565

Note: The total is not exact because of rounding.
Source: ARC (2000).

a population that is projected to double by 2050 (SAWS 2003). Two of these projects involve transferring water 200 km or more from the lower reaches of the Colorado and the Guadalupe Rivers (figure 7). These two projects could potentially harm the health of the downstream Matagorda Bay and Guadalupe estuaries, both of which support productive fisheries. Flows from the Guadalupe are also necessary to maintain the food sources of the endangered whooping crane, whose only self-sustaining population winters in this area (USFWS 1994).

Although we cannot point to definitive studies on the impacts of these projects, existing analyses indicate that their ecological consequences could be severe. The local regional water-planning group modeled the impacts of the diversions to off-channel reservoirs that would be used for the Colorado River project, using projected 2050 water demands. During a repeat of the drought period of 1947 to 1956, the project would reduce annual average flows by 28%. Average monthly flows in July through September would be below the critical inflow levels necessary to provide sanctuary habitat for fishery species in the estuary during the most severe droughts (LCRWPG 2000). Modeled flows show even more severe declines during certain annual and seasonal periods of the same drought (Norman Johns, National Wildlife Federation, Austin, TX, personal communication, March 2004). The Guadalupe River project will use about one-sixth of the unused water rights in that basin. Modeling indicates that if all existing water rights in the basin were exercised during a repeat of the 1950–1956 period, estuary inflows would be reduced by 17% to 43% below current levels and by 36% to 72% below historic levels, depending on the year (Norman Johns, National Wildlife Federation, Austin, TX, personal communication, March 2004). Future long-term studies are planned on the impacts of both these projects (SAWS 2003).

Atlanta, Georgia. The fast rate of population growth in Atlanta threatens to expand the environmental impacts of its current water management system. Population in the 10-county metropolitan area is projected to increase more than 50% by 2030 (MNGWPD 2003). Most of Atlanta's water comes from

Figure 9. Primary water sources for Atlanta, Georgia, and the downstream Apalachicola Bay estuary.

the Chattahoochee River and especially from the Lake Lanier reservoir, which was completed in 1958 for the purposes of hydropower, water supply, and flood control (table 5, figure 9). The city's other main water source is the Etowah River, which has a similar multipurpose reservoir (Lake Allatoona) that was constructed in 1950.

The two Atlanta water-supply reservoirs have affected natural conditions in these rivers. The flow of the Chattahoochee through Atlanta is controlled by the management of Lake Lanier, which decreases the frequency of high- and low-flow events and causes large daily fluctuations in flows because of hydropower generation (Couch et al. 1996). Releases of cold water from the dam have altered the natural thermal regime of the river, and colder water temperatures have enabled development of a nonnative trout fishery downstream from Lake Lanier to Atlanta (Couch et al. 1996). For the Etowah River, Burkhead and colleagues (1997) have detailed the impacts of the Allatoona dam (in combination with other stressors) on the extremely diverse native fauna, much of which is now extirpated. The depauperate fish fauna below the dam becomes increasingly diverse for some 60 km downstream, suggesting that the impoundment is having serious impacts.

Ecological impacts in these and other rivers could increase substantially because of future water-supply needs. Current

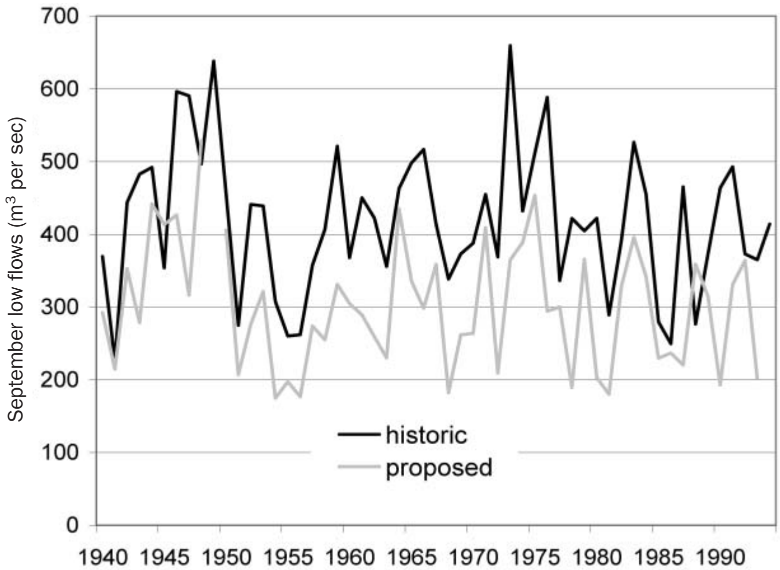

Figure 10. Simulated impacts of future water use on flows in the lower Apalachicola River. The 1939–1993 measured September low flows are compared with model simulated low flows based on projected 2030 levels of water use in the basin, as submitted by the state of Florida during interstate water compact negotiations in February 2002.

plans are to increase withdrawals from the Chattahoochee, the Etowah, and other rivers; to reallocate reservoir storage from other functions to water supply; to construct five new small reservoirs; and to increase water conservation (ARC 2000, MNGWPD 2003). Atlanta's initial proposal in 1990 to increase withdrawals from the Chattahoochee and Flint Rivers was immediately controversial, in great part because of potential downstream impacts on the Apalachicola Bay estuary (figure 9). Apalachicola Bay is one of the most important fishery resources in North America, accounting for 90% of Florida's oyster production (ACOE 1998). Conflicts over water use in the Apalachicola–Chattahoochee–Flint (ACF) river basin led to the 1997 signing of a three-state water compact between Alabama, Florida, and Georgia (Richter et al. 2003). After 5 years of negotiation, the states failed to reach agreement on a formula for allocating water in the ACF basin.

A major concern is that increased water withdrawals in the Atlanta area will markedly decrease low flows in the downstream Apalachicola River, thereby affecting the estuary (figure 10). The exact water-reduction threshold that would trigger permanent negative impacts is uncertain, but research indicates that artificial reduction in flows during periods of drought or near-drought could cause the ecosystem to switch permanently to a far less productive state (Livingston et al. 1997).

The most comprehensive study of the impacts of Atlanta's future water withdrawals on these rivers is the Army Corps of Engineers' draft environmental impact statement (EIS) on the proposed ACF compact (ACOE 1998). The general conclusion of the EIS is that current levels of flow can be maintained in the ACF rivers, but only through some combination of water conservation and reductions in reservoir functions other than water supply. However, projections of population growth used in the EIS have proved to be underestimates when compared with actual population growth, because of faster-than-expected growth around Atlanta (ACOE 1998). Furthermore, estimates of water demand were based on long-term averages, not on the worst-case demands that may occur during droughts (ACOE 1998). These considerations affect the degree to which measures such as water conservation and changes in reservoir operations would have to be implemented.

Loss of ecosystem services

In addition to the impacts on ecosystem processes and native biota described for each of our case studies, the development of water resources has destroyed, compromised, or threatened a wide range of ecosystem services. Natural freshwater and estuarine ecosystems provide many beneficial services, including commercial and sport fisheries, flood attenuation, groundwater recharge, wildlife habitat, pollution dilution, soil fertilization, recreation, and aesthetics (Daily 1997, Postel and Richter 2003). Although a full description of impacts to ecosystem services in our case-study areas is beyond the scope of this article, we cite a few examples below.

Many impacts on fishery resources and wildlife habitat have already been mentioned. With regard to recreation benefits, the economic impact of trout fishing in the upper Delaware tributaries was estimated at $30 million in 1996 (Maharaj et al. 1998). Once restored, the lower Owens River is expected to provide greater recreational opportunities (LADWP 2000). Restoration activities along the Salt River are expected to provide benefits in terms of recreation, flood control, and habitat (McKinnon 2003). Similar restoration is under study for the Los Angeles River (Western Water 2003). Restoration projects such as these are likely to cost tens of millions of dollars or more, indicating the value to society of healthy freshwater ecosystems.

Natural functioning of many of these systems also reduces or prevents water and air pollution. The dams in the upper Delaware River tributaries must now be carefully managed to control the salt front in the Delaware estuary (NYCDEP 1998), because of the potential impacts of increased salinity on downstream water supplies. Water withdrawals and reservoir evaporation increase the salinity of the lower Colorado River, a problem whose costs and repair expenses are in the hundreds of millions of dollars (Miller et al. 1986). Water pollution is a serious problem in the Chattahoochee River near Atlanta (Couch et al. 1996) and could become worse if increased consumption of water from the river reduces its dilution capacity. Dry lake beds at Owens Lake and Mono Lake have led to dust storms and serious air pollution (Wiens et al. 1993, Hundley 2001).

To our knowledge, a full cost accounting of the value of lost ecosystem services compared with the benefits gained has not been conducted for any of these areas, so it is impossible for us to rigorously assess the net effects of these projects. Our purpose here is merely to point out that, while the human benefits associated with water development have clearly been considerable, they have been accompanied by serious consequences for the health of river ecosystems, leading to a significant diminishment of many environmental qualities and services that are valuable to society in both monetary and nonmonetary terms. As we discuss below, improving the assessment of trade-offs between ecological integrity, ecosystem services, and human needs for water would be an important step forward for water resource management.

Analysis and recommendations

A problem that is common to each of our case studies, and in fact is typical of the vast majority of water projects completed to date, is a lack of consideration of the water needs of freshwater ecosystems. Water resource development needs to be reoriented under the rubric of ecologically sustainable water management, which has been defined as "protecting the ecological integrity of affected ecosystems while meeting intergenerational human needs for water and sustaining the full array of other products and services provided by natural freshwater ecosystems" (Richter et al. 2003). In advocating this new paradigm for water management, we are not proposing that water withdrawals be reduced everywhere to restore freshwater ecosystems to their natural state. Clearly this would be both undesirable and unrealistic. What we are proposing is that by using existing technologies and management tools, and by spurring further innovations in their use, water managers can do a much better job of protecting freshwater ecosystems while meeting human needs.

A first principle of ecologically sustainable water management is the need to provide freshwater ecosystems with the water flows necessary to sustain their health. This includes both assessing those needs scientifically and implementing water policies and laws that protect adequate water flows for these ecosystems. Postel and Richter (2003) have advanced the concept of creating an "ecosystem allocation" of water in each river basin as a foundation for water management. They discuss the scientific approaches available for defining ecosystem water needs, provide examples of laws and policies that are being used around the world for protecting ecosystem water allocations, and describe many innovative practices being applied by water managers to restore the balance between human and ecosystem needs. It is much easier to integrate the allocation of water for ecosystems into water development plans before water supplies are overallocated to human uses than it is to restore ecosystem flows. Thus, it is important that water planners move swiftly to implement ecosystem water allocations before water supplies become overtaxed.

We believe it is possible to attain a better balance between human and ecosystem needs, for two primary reasons. First, huge gains can be made in increasing water productivity. Many cost-effective methods for water conservation already exist, and new technologies are constantly evolving that will allow even greater efficiencies (Gleick 2000, Vickers 2001, Gleick et al. 2003). Second, through improved planning and management, it should be possible to store and extract the water necessary for human uses in much less ecologically damaging ways than those used in the past.

The gains from urban water conservation in San Antonio, New York, and Los Angeles have already been discussed. Conservation in Los Angeles and San Antonio has helped these cities deal with the reductions in water supply that accompanied ecosystem restoration in the Owens Valley and Mono Basin and better management of the Edwards Aquifer. Elsewhere, there appears to be great untapped potential for even more urban water conservation. A recent study in California found that 30% of urban water use could be saved using existing technologies at costs below what it would cost to tap into new sources of supply (Gleick et al. 2003).

Evidence from our other two cities confirms this potential. While water-use efficiency in the Atlanta area has increased only slightly since 1990 (ARC 2000), an analysis of future conservation options indicates that by 2030 per capita water use can be reduced by another 15%, while saving money because of reduced water treatment costs (MNGWPD 2003). Some progress has been made in Phoenix, but there appears to be potential for much more, to judge from the example of Tucson, another Arizona city with a similar climate that began conserving water in the 1970s (Kupel 2003). Systemwide and single-family residential per capita water use were 28% and 26% higher, respectively, in Phoenix than in Tucson in 2001 (WRA 2003).

Water management during droughts is of particular importance. Cities need drought-management plans that require more aggressive conservation as drought severity increases and that are responsive to ecosystem needs, like the plan being implemented for the Edwards Aquifer. Water planning is typically done using projections of "safe yield," which is the maximum quantity of water that can be withdrawn during the drought of record. But estimates of safe yield need to take into account the flow needs of ecosystems during droughts and the ability of cities to reduce demands during those same periods. Los Angeles's responses to the droughts of 1976–1977 and 1987–1992 showed that cities can reduce water demand when necessary (LADWP 2002).

There is considerable potential for meeting future urban water-supply needs through reductions in agricultural water use as well. It is widely acknowledged that irrigation practices in many regions of the world, such as the western United States, are highly inefficient when compared with those in other regions that are using state-of-the-art technologies and management practices (Gleick 2000, Vickers 2001, USBR 2003). The MWD in southern California has invested considerable sums of money to improve agricultural water

Table 6. Future water-demand and water-supply projections for San Antonio, Texas, and Atlanta, Georgia.

City	Water demand (millions of cubic meters per year)	Water supply (millions of cubic meters per year)	Difference (millions of cubic meters per year)	Planned increases in supply from selected projects
San Antonio	382	604	222	185 from Colorado River; 86 to 117 from Guadalupe River
Atlanta (moderate population projection)	1035	1396	361	478 from increased withdrawals from Chattahoochee, Etowah, and other rivers; 126 from five new reservoirs
Atlanta (high population projection)	1191	1396	205	478 from increased withdrawals from Chattahoochee, Etowah, and other rivers; 126 from five new reservoirs

Note: San Antonio projections are estimated dry year supply and demand for 2050. Projections for the Atlanta area (16 north Georgia counties) are estimated "safe yield" supply and demand under two different population projections (assuming implementation of recommended aggressive conservation program) for 2030. The difference between projected demand and supply and the planned increases in supply indicates the potential for scaling back or postponing these planned projects.
Source: SAWS (2003) and MNGWPD (2003).

practices so that the water saved can be used for urban water supplies (Hundley 2001). In Texas, Keplinger and colleagues (1998) found that paying farmers to suspend irrigation in dry years was a relatively cost-effective way to maintain the flows of Comal Springs.

Even where increases in water supply are needed, options are available to increase the supply in a far less ecologically damaging manner than has been commonly practiced. Existing reservoir storage can be reallocated to urban water supply from other purposes, such as irrigation supply, hydroelectric power generation, reservoir recreation, and even flood control. Groundwater aquifers can be used in conjunction with surface reservoirs to increase total storage capacity, and many aquifers can be better recharged by enhancing natural infiltration or by providing artificial recharge. When additional surface storage is absolutely necessary, off-channel storage facilities can be constructed, instead of building in-stream reservoirs that block the movements of aquatic organisms and alter the natural flows of water, sediment, and organic materials. Considerable potential also exists for restoring more natural patterns of river flow below storage reservoirs. With increased attention to the flow regimes needed to support downstream ecosystems, dam operations can be modified to more closely mimic the natural flow regime of rivers (Postel and Richter 2003, Richter et al. 2003).

Achieving ecologically sustainable water management will require a proactive planning process that examines the economic and ecological trade-offs between water conservation, options for increasing supply, and the integrity of freshwater and estuarine ecosystems. A review of water-planning documents for San Antonio (SAWS 2003) and the Atlanta area (MNGWPD 2003) indicates that despite projections of rapid population growth, both cities have leeway to drastically scale down or postpone development of their planned sources of water supply (table 6). Given future possible technological innovations in water conservation, we might expect even greater reductions in their future water demands. While it is clearly prudent to have a small buffer of supplies in case of unforeseen circumstances or emergencies, it is also important not to open up new sources of supply until absolutely necessary, so as to protect ecosystem health and to create as great an incentive as possible for conservation.

Ultimately, water planners will need to set limits on human alterations to river flows in many basins in order to protect ecosystem water allocations (Postel and Richter 2003). Such an approach is embodied in the precedent-setting South Africa National Water Act of 1998. The act states that "the quantity, quality and reliability of water required to maintain the ecological functions on which humans depend shall be reserved so that the human use of water does not individually or cumulatively compromise the long term sustainability of aquatic and associated ecosystems." Similar approaches are being applied in the Murray-Darling River basin in Australia, where a cap has been placed on future water withdrawals in order to arrest the deterioration of the river's ecological health. Under the assumption that there is great potential for future conservation and that water markets will be implemented, a 1999 study projected that the Murray-Darling basin economy will double over 25 years, even with the cap in place (AATSE–IEA 1999).

Although there are many positive trends leading toward a new paradigm for water development, the transition to this new paradigm has really only just begun. While some cities such as San Antonio are well into making the transition, there are many others, such as Atlanta and Phoenix, that have barely started. While we can point to some good examples of improved practices, meeting the challenge of water management in the 21st century will require a much broader application of existing and emerging methods for increased water productivity and better ecosystem management, in combination with innovative solutions such as those that are being tried in the Murray-Darling Basin. There is clearly an immense task ahead, but if the principles outlined here are wholeheartedly adopted, we believe this endeavor can ultimately be successful.

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References cited

[AATSE–IEA] Australian Academy of Technological Sciences and Engineering and Institution of Engineers, Australia. 1999. Water and the Australian Economy: A Joint Study of the AATSE and IEA. Victoria (Australia): AATSE.

[ACOE] US Army Corps of Engineers. 1998. Water allocation for the Apalachicola–Chattahoochee–Flint (ACF) River Basin, main report of the Draft Environmental Impact Statement. Federal Register 63: 53023–53024.

[ADWR] Arizona Department of Water Resources. 1999. Third Management Plan for the Phoenix Active Management Area, 2000–2010. Phoenix (AZ): Arizona Department of Water Resources.

Aguirre M. 2003. Defending Spain's Ebro River. World Rivers Review 18 (3): 8–9.

[ARC] Atlanta Regional Commission. 2000. Atlanta Regional Water Supply Plan Update, August 2000. Atlanta (GA): ARC.

Baldigo BP, Schuler GE, Riva-Murray K. 2002. Mussel Community Composition in Relation to Macrohabitat, Water Quality, and Impoundments in the Neversink River, New York. US Geological Survey Open-File Report 02-104.

Brothers TS. 1984. Historical vegetation change in the Owens River riparian woodland. Pages 75–84 in Warner RE, Hendrix KM, eds. California Riparian Systems: Ecology, Conservation, and Productive Management. Berkeley: University of California Press.

Burkhead NM, Walsh SJ, Freeman BJ, Williams JD. 1997. Status and restoration of the Etowah River, an imperiled southern Appalachian ecosystem. Pages 375–444 in Benz GA, Collins DE, eds. Aquatic Fauna in Peril: The Southeastern Perspective. Decatur (GA): Lenz Design and Communications.

CALFED Bay–Delta Program. 2000. Final programmatic environmental impact statement/environmental impact report. Federal Register 65: 45104–45105.

Chittenden ME Jr. 1976. Present and historical spawning grounds and nurseries of American shad, *Alosa sapidissima*, in the Delaware River. Fishery Bulletin 74: 343–352.

Couch CA, Hopkins EH, Hardy PS. 1996. Influences of Environmental Settings on Aquatic Ecosystems in the Apalachicola–Chattahoochee–Flint River Basin. Atlanta (GA): US Geological Survey National Water-Quality Assessment Program. Water-Resources Investigation Report 95-4278.

Crowe JC, Sharp JM Jr. 1997. Hydrogeologic delineation of habitats for endangered species: The Comal Springs/River system. Environmental Geology 30 (1–2): 17–28.

[CSWRCB] California State Water Resources Control Board. 1994. Final Environmental Impact Report for the Review of Mono Basin Water Rights of the City of Los Angeles. (12 July 2004; *www.monobasinresearch.org/images/mbeir/fititlepage.pdf*)

Daily GC, ed. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Washington (DC): Island Press.

Donnelly TW. 1993. Impoundment of rivers: Sediment regime and its effects on benthos. Aquatic Conservation: Marine and Freshwater Systems 3: 331–342.

[EAA] Edwards Aquifer Authority. 1998. Groundwater Management Plan 1998–2008. San Antonio (TX): EAA.

Eckhardt GA. 2004. Edwards Aquifer Homepage: Conservation. (18 June 2004; *www.edwardsaquifer.net/conservation.html*)

Endreny T. 2002. Sustainability for New York's Drinking Water: Source Expansion, User Conservation, and Climate Prediction. (19 June 2004; *www.nywea.org/clearwaters/313060.html*)

Fenner P, Brady WW, Patton DR. 1985. Effects of regulated water flows on regeneration of Fremont cottonwood. Journal of Range Management 38: 135–138.

Galusha D. 1999. Liquid Assets: The Story of the New York City Water Supply System. Fleischmanns (NY): Purple Mountain Press.

[GAO] General Accounting Office. 2003. Freshwater Supply: States' Views of How Federal Agencies Could Help Them Meet the Challenges of Expected Shortages. Washington (DC): GAO.

Gleick PH. 2000. The changing water paradigm: A look at twenty-first century water resources development. Water International 25: 127–138.

———. 2003. Global freshwater resources: Soft-path solutions for the 21st century. Science 302: 1524–1528.

Gleick PH, Haasz D, Henges-Jeck C, Srinivisan V, Wolff G, Cushing KK, Mann A. 2003. Waste Not, Want Not: The Potential for Urban Water Conservation in California. Oakland (CA): Pacific Institute for Studies in Development, Environment, and Security.

Graf WL. 2000. Locational probability for a dammed, urbanizing stream: Salt River, Arizona, USA. Environmental Management 25: 321–335.

Gumprecht B. 1999. The Los Angeles River: Its Life, Death, and Possible Rebirth. Baltimore: Johns Hopkins University Press.

Hamilton JM, et al. 2003. Edwards Aquifer Authority Hydrogeologic Data Report for 2002. San Antonio (TX): Edwards Aquifer Authority.

Harman WN. 1974. The effects of reservoir construction and canalization on the mollusks of the Upper Delaware watershed. Bulletin of the American Malacological Union (May): 12 –14.

Hundley N Jr. 2001. The Great Thirst: Californians and Water—A History. Berkeley: University of California Press.

Hutson SS, Barber NL, Kenny JF, Linsey KS, Lumia DS, Maupin MA. 2004. US Geological Survey Estimated Use of Water in the United States in 2000. USGS Circular 1268. (30 June 2004; *water.usgs.gov/pubs/circ/2004/circ1268/*)

Keplinger KO, McCarl BA, Chowdhury ME, Lacewell RD. 1998. Economic and hydrologic implications of suspending irrigation in dry years. Journal of Agricultural and Resource Economics 23: 191–205.

Kupel D. 2003. Fuel for Growth: Water and Arizona's Urban Environment. Tucson: University of Arizona Press.

[LADWP] Los Angeles Department of Water and Power. 2000. Fiscal Year 2000 Urban Water Management Plan. Los Angeles: LADWP.

———. 2002. Urban Water Management Plan Fiscal Year 2001–2002 Update. Los Angeles: LADWP.

[LCRWPG] Lower Colorado River Water Planning Group. 2000. Lower Colorado River Water Planning Group Adopted Plan. Austin (TX): LCRWPG.

Livingston RJ, Xufeng N, Lewis FG III, Woodsum GC. 1997. Freshwater input to a Gulf estuary: Long-term control of trophic organization. Ecological Applications 7: 277–299.

Maharaj V, McGurrin J, Carpenter J. 1998. The Economic Impact of Trout Fishing on the Delaware River Tailwaters in New York. Alexandria (VA): American Sportfishing Association and Trout Unlimited.

McKinnon S. 2003. $25 mil set aside for river projects plan would restore habitats, stop floods. Arizona Republic, 6 November 2003, p. B4.

Miller RR. 1961. Man and the changing fish fauna of the American Southwest. Papers of the Michigan Academy of Science, Arts, and Letters 46: 365–404.

Miller TO, Weatherford GD, Thorson JE. 1986. The Salty Colorado. Washington (DC): Conservation Foundation.

Minckley WL, Deacon JE. 1968. Southwestern fishes and the enigma of endangered species. Science 159: 1424–1432.

[MNGWPD] Metropolitan North Georgia Water Planning District. 2003. Water Supply and Water Conservation Management Plan—Final Report. Atlanta (GA): MNGWPD.

Mueller GA, Marsh PC. 2002. Lost, a Desert River and Its Native Fishes: A Historical Perspective of the Lower Colorado River. Fort Collins (CO): US Geological Survey. (30 June 2004; *www.fort.usgs.gov/products/publications/10026/10026*)

[NYCDEP] New York City Department of Environmental Protection. 1998. Drought Management Plan and Rules. (21 June 2004; *www.nyc.gov/html/dep/pdf/droughtp.pdf*)

———. 2004. New York City's Water Supply System. (18 June 2004; *www.nyc.gov/html/dep/html/droughthist.html*)

[NYWEA] New York Water Environment Association. 2003. Government Affairs Newsletter for February 1 2003 to April 25 2003. (18 June 2004; *www.nywea.org/gac/gac03_02-04.cfm*)

Postel S, Richter B. 2003. Rivers for Life: Managing Water for People and Nature. Washington (DC): Island Press.

Postel S, Morrison J, Gleick P. 1998. Allocating fresh water to aquatic ecosystems: The case of the Colorado River delta. Water International 23: 119–125.

Revenga C, Brunner J, Henninger N, Kassem K, Payne R. 2000. Pilot Analysis of Global Ecosystems: Freshwater Systems. Washington (DC): World Resources Institute.

Richter BD, Mathews R, Harrison DL, Wigington R. 2003. Ecologically sustainable water management: Managing river flows for ecological integrity. Ecological Applications 13: 206–224.

[SAWS] San Antonio Water System. 2003. Quarterly Report on Water Resources, July–September 2003. San Antonio (TX): SAWS.

Shen D. 2003. China's water crisis. World Rivers Review 18 (3): 6–7.

[SRC] Schoharie River Center, Environmental Study Team. 2002. A Rapid Bioassessment on the Schoharie Creek, Schenectady, Montgomery, and Schoharie County, NY. (30 June 2004; *http://home.att.net/~esteam/SchoharieCreek.pdf*)

Strayer DL, Ralley J. 1991. The freshwater mussels (Bivalvia: Unionoidea) of the upper Delaware River drainage. American Malacological Bulletin 9: 21–25.

Stromberg JC, Patten DT. 1992. Response of *Salix lasiolepis* to augmented stream flows in the upper Owens River. Madrono 39: 224–235.

Tellman B, Yarde R, Wallace MG. 1997. Arizona's Changing Rivers: How People Have Affected the Rivers. Tucson: University of Arizona. Water Resources Research Center Issue Paper 19.

Thakkar H. 2003. Huge water transfer scheme would be death warrant for India's rivers. World Rivers Review 18 (3): 1, 7.

[UN] United Nations. 2002. World Urbanization Prospects: The 2001 Revision. Data Tables and Highlights. New York: UN.

———. 2003. World Population Prospects: The 2002 Revision. Highlights. New York: UN.

[USBR] US Bureau of Reclamation. 2003. Water 2025: Preventing Crises and Conflict in the West. (18 June 2004; *www.doi.gov/water2025/)*

[USCB] US Census Bureau. 2000. National Population Projections. (18 June 2004; *www.census.gov/population/www/projections/natproj.html*)

[USFWS] US Fish and Wildlife Service. 1994. Whooping Crane Recovery Plan. Albuquerque (NM): USFWS.

———. 1995. San Marcos and Comal Springs and Associated Aquatic Ecosystems (Revised) Recovery Plan. Albuquerque (NM): USFWS.

———. 1998. Owens Basin Wetland and Aquatic Species Recovery Plan, Inyo and Mono Counties, California. Portland (OR): USFWS.

[USFWS-AEFSO] US Fish and Wildlife Service, Arizona Ecological Field Services Office Document Library. Various years.

Vickers A. 2001. Handbook of Water Use and Conservation. Amherst (MA): WaterPlow Press.

Western Water. 2003. The Los Angeles River. Sacramento (CA): Water Education Foundation.

Whittlesey SM. 1997. The vanished river. Pages 28–57 in Whittlesey SM, Ciolek-Torrello R, Altschul JH, eds. Vanishing River: Landscapes and Lives of the Lower Verde Valley: The Lower Verde Archaeological Project: Overview, Synthesis, and Conclusions. Tucson (AZ): SRI Press.

Wiens JA, Patten DT, Botkin DB. 1993. Assessing ecological impact assessment: Lessons from Mono Lake, California. Ecological Applications 3: 595–609.

[WRA] Western Resource Advocates. 2003. Smart Water: A Comparative Study of Urban Water Use Efficiency across the Southwest. Boulder (CO): WRA.