

Watersheds and Urbanization: Compelling and Emerging Challenges for the 21st Century

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Introduction

The United Nations estimates that by the year 2025, 60% of the world's population will live in cities ([UNFPA 1999](#)), highlighting the increasing importance of urban areas in shaping local and global environments. Alteration of local atmospheric and hydrologic processes by urbanization may yield a host of effects, of which altered urban climate and degraded stream water quality are two important examples, both closely related to the increased impervious surface area associated with urban systems. Presently, the impervious surface area of the conterminous U.S. is approximately equal to the area of the state of Ohio ([Elvidge et al., 2004](#)), demonstrating the magnitude of alteration by urbanization of the surface of the United States.

An urban system casts a direct footprint that is easily ascertained by a view from above. However, there are few (if any) urban landscapes whose water needs can be supported based on the direct footprint of the urban system; water demand is met by appropriation of supply from nearby and possibly distant watersheds, and underlying aquifers. As a result, the urban hydrologic footprint typically greatly exceeds the direct footprint. The boundary of the urban hydrologic footprint may not necessarily coincide with topographic boundaries (in the case of watersheds) or lithologic boundaries (in the case of aquifers).

Although traditional urban hydrology typically concerns the direct urban footprint in terms of its effects on the quantity and quality of runoff to receiving water bodies, our goal is to broaden the examination to the larger hydrologic footprint that is governed by the decisions of urban water managers and other urban policy makers. Although the effects of watershed and aquifer exploitation by urban demand are indistinguishable from exploitation of these same resources by agriculture and other non-urban industries, a critical difference between these two types of exploitation are the tradeoffs

that define management practices. The tradeoffs governing urban system management can be expected to be considerably more complex than those governing industry, since non-monetary costs and benefits such as standard of living, quality of environment, recreation, and public health hold higher priority relative to monetary costs and benefits in the urban system relative to agricultural and industrial settings. Furthermore, the development of management decisions in the urban setting is greatly influenced by public input.

To make intelligent decisions, urban water managers must rely on guidance provided by hydrologists, who must understand the fluxes, residence times, storage volumes, and flow pathways of water and its associated dissolved and suspended constituents in the various reservoirs of the urban hydrologic system. However, our understanding of these properties is highly limited for the important reservoirs and fluxes that cannot be observed directly (e.g. groundwater, plants, etc.), and measurements of the dissolved and suspended constituents in the system are also highly limited. Furthermore, the urban hydrologic footprint is not restricted to the terrestrial liquid phase, since growing evidence suggests that urban centers exert their own influences on local temperature, moisture, as well as on rates of evapotranspiration and precipitation.

Science Needed to Address Emerging Issues

This section describes compelling and emerging challenges to be addressed during the next decades by urban water managers and urban policy makers with respect to optimal management of urban hydrologic systems. To place these issues in the context of knowledge gaps regarding storage reservoirs, fluxes, residence times, and pathways in the urban hydrologic system, we divide the system into three compartments: atmosphere, surface, and subsurface.

Atmosphere

Urbanization has a profound effect on the surface radiation budget and exchange of heat, moisture, and momentum between the land surface and the atmosphere. Altered surface cover properties (e.g., albedo) and enhanced turbulent kinetic energy and mechanical mixing by buildings act to modify surface fluxes. The modified fluxes combined with an augmented atmospheric composition creates greater variability of regional meteorology and climate including temperature distribution and wind flow (Bornstein 1968; Bornstein 1987), clouds (Inoue and Kimura 2004), precipitation (e.g., Changnon 1992), and air quality (Quattrochi et al. 1998). In general, large urban areas have lighter winds, less humidity, sunshine and snow and more cloud, fog, thunderstorms and rain compared to surrounding rural areas (Oke 1987; Cotton and Pielke 1995).

Changes in vegetation structure associated with urbanization can have a marked influence on evaporative fluxes. For example, urban and especially suburban development in much of the western U.S. has replaced vegetation in semi-arid and arid regions (defined as precipitation less than 250 mm yr⁻¹ with available energy to

evaporate all of it) with grass and tree species that are native to humid temperate environments with annual rainfall above 1000 mm. Typical evapotranspiration (ET) rates of well-watered grasses and deciduous forests are $> 3\text{-}5 \text{ mm d}^{-1}$ (Meyers 2001, Wilson and Baldocchi, 2000), while those of semi-arid and arid ecosystems are an order of magnitude smaller. Bowen ratios (ratio of sensible to latent heat) of deciduous forests are 0.4 on average, and biomes in Mediterranean climates can have Bowen ratios as high as 5 or more (Wilson et al. 2002). Ecosystem-scale evapotranspiration in arid regions has not been extensively studied, but Bowen ratios should be even higher than those in Mediterranean climates, especially under limiting water conditions.

Western urban regions are often surrounded by irrigated crops, including water-inefficient species with high transpiration rates such as alfalfa and cereals. Latent heat fluxes in irrigated arid situations can exceed incoming radiation as energy is advected horizontally from surrounding hot dry lands (Rosenberg and Verma 1978). The same pattern has been observed in urban parks (Spronken-Smith et al. 2000), and it is reasonable to assume that ET rates from irrigated lawns in hot climates will also exceed available energy.

Due to the large evaporative demand in arid regions, and since mesic plant species in arid climates necessarily require large amounts of irrigation water, urbanization in the West has the potential to substantially increase evapotranspiration fluxes relative to native vegetation. Changes in ET and other factors associated with agriculture and urbanization have been implicated in regional climate change in semi-arid and mountainous regions of Colorado (Stohlgren et al. 1998). Further, similar changes in surface moisture and heat fluxes may alter convective rainfall patterns (Pielke 2001).

Urban development in eastern U.S. watersheds generally involves loss of vegetation and replacement of deciduous forests with built landscapes and impermeable surfaces. Removal of deciduous forests results in a net decrease of transpiration, causing decreased latent heat flux, increased sensible heat flux, and increased surface flows (Dow and Dewalle, 2000).

Evapotranspiration is also influenced by urban temperature and roughness effects since they affect the potential evaporation and boundary layer turbulence. Knowledge of such urban biosphere-atmosphere interactions, however, is limited and represents a major scientific challenge for the coming years.

Although phenomena like the so-called urban heat island (UHI) were identified in the 19th century (Howard 1833; Landsberg 1956), comprehensive measurement and understanding of how urbanization alters land-surface processes, weather, climate, and hydrology remains elusive. Knowledge in these areas is critical since land-surface modification is as important for past and future climate change as greenhouse gas concentrations and other global-scale anthropogenic effects (e.g., Pielke et al. 1998).

Although the UHI is commonly generalized as "cities are hotter", in reality, urban effects on temperature are highly complex. For example, Brazel et al. (2000) show that in Washington/Baltimore, a large, humid, east coast metropolitan area, average minimum temperatures are 4-5 °C higher than in surrounding rural areas, while maximum temperatures are only 1-2 °C higher. In Phoenix, located in the arid southwest, minimum temperatures are also several degrees warmer than at a nearby

desert site, but maximum temperatures are actually cooler. This urban "oasis effect" appears to be the result of the irrigation of the urban and surrounding agricultural landscapes, which increases evapotranspiration and reduces the amount of solar radiation available for sensible heating and may in part be responsible for altered precipitation patterns (Diem and Brown, 2003). More generalized studies support the hypothesis that the UHI is most pronounced at night (e.g., Cayan and Douglas, 1984), but also suggest that averaged across the United States the urban effect may be one of slight cooling during the day (Kalnay and Cai, 2003). Such cooling may arise from the aforementioned increase in evapotranspiration over arid or semi-arid regions, although increased turbulent mixing due to the larger urban roughness length (e.g., Atkinson, 2003) and a larger solar albedo from urban aerosols (Ramanathan et al., 2001) may also play roles. The urban temperature effect also varies by season and the size and composition of the urban area (Oke, 1973; Landsberg, 1981; Karl et al., 1988).

The UHI is one of four factors typically identified as contributing to urban-induced precipitation modification. As summarized by Huff and Changnon (1973) urbanization affects precipitation by:

1. Altering the atmospheric stability through urban temperature effects (UHI).
2. Modifying cloud microphysical processes through the addition of cloud condensation nuclei (CCN) from automobiles and industry.
3. Increasing the surface roughness length, turbulence, and mechanical mixing due to the presence of tall buildings.
4. Modifying the low-level moisture content.

The effect on precipitation of these four factors acting synergistically with other local forcing (e.g., terrain, coastline curvature) and global forcing (e.g., climate change) processes is difficult to isolate for a given city. In fact, decades of data collection, analysis, and numerical simulations have produced varying and uncertain conclusions (Lowry, 1998). If present, the urban effect is likely to be observed downwind, rather than within, the central urban district because time is needed for the formation of droplets and their growth to sufficient size to precipitate to the ground surface (Oke 1987). Examples of urban precipitation modification are abundant, and first appeared in the early 20th century (e.g., Horton 1921, see also Landsberg 1956). More recent studies have documented increases in warm-season precipitation (typically related to deep convection) ranging from 5%-25% over and up to 50-75 km downstream of medium (e.g., Tulsa, OK) to large (e.g., St. Louis, MO) U.S. cities (e.g., Huff and Changnon 1973, Braham et al. 1981, Changnon et al. 1991). The areal coverage and magnitude of the enhancement is directly related to the size of the city (Changnon 1992). More recently, Shepherd et al. (2002) used the precipitation radar aboard the Tropical Rainfall Measuring Mission (TRMM) satellite to illustrate that for several southeast US cities, the average percentage increase in conditional mean rainfall rate in the identified urban impact zone over an upwind control area was 28%. Evidence has suggested that the increased precipitation amounts may be produced by urban-induced invigoration of existing storms making them more intense (Huff and Vogel 1978; Changnon 1978). This possibility has implications for surface flooding and stream geomorphology in urban environments.

In contrast, there are examples of urban suppression of precipitation. [Changnon et al. \(1991\)](#) report a slight decrease in precipitation downstream of St. Louis for some cool-season events that were likely dominated by stratiform precipitation. Observations by the TRMM satellite also suggest the possibility of precipitation suppression downstream of urban areas, with pollution acting to inhibit cloud droplet coalescence and both primary and secondary ice generation (e.g., [Rosenfeld 2000](#)). [Givati and Rosenfeld \(2004\)](#) suggest that such effects may reduce precipitation over portions of California and Israel by as much as 25%. Suppression of precipitation in watersheds feeding surface water resources or recharge zones of ground water resources have significant implications for areas experiencing severe water shortages. Finally, in mid and high latitudes, the warming accompanying the UHI may alter the fraction of precipitation falling as snow ([Changnon, 2003](#)), which has significant implications for cool-season urban runoff and water-borne pollution.

The above discussion illustrates the complex ways in which land-surface change alters the weather, climate, and land surface-atmosphere components of the hydrologic cycle of urban regions. Unfortunately, comprehensive understanding of these effects across the complex mosaic of urban types, climates, and local forcings is lacking. In particular, knowledge is needed to better understand and predict:

1. how urbanization, in its many variants, climatological regions, and scales, alters the distribution, intensity, and variability of precipitation.
2. the implications of urban precipitation modification for hydrologic and ecological processes, particularly those with relevance to society such as water management, water-borne pollution, and disease dynamics.
3. how urban-scale and global/regional climate change interact to dampen or amplify urban temperature, evapotranspiration, and precipitation effects.

Addressing these issues will require comprehensive measurement and modeling capabilities. In particular, the following are needed:

1. Long term and intensive (i.e., field program) measurements of surface energy budgets and fluxes in a variety of urban forms and climate regions, including complex terrain. Essential to the success of these efforts are
 - a) improvements in surface-layer mass conservation (e.g. eddy covariance) methodologies,
 - b) improvements in convective boundary layer budgeting techniques (e.g., [Raupach et al. 1992](#); [Lloyd et al 2001](#), [Helliker et al. in review](#)),
 - c) advances in instrumentation to measure water vapor and its stable isotopes (e.g., tunable diode laser ([Lee et al. in review](#)) and other infrared absorption techniques ([Kerstel et al. 1999](#))),
 - d) extensive application of sapflow techniques ([Granier 1987](#)) combined with radio telemetry innovation and extensive vegetation surveys, and
 - e) large-scale ET flux measurements with scanning Raman lidar ([Eichinger et al., 2000](#)) or other technologies.
2. The development of microscale meteorological networks to describe the urban heat island, urban moisture island, and urban-scale thermally driven circulations

and their impact on convective initiation and precipitation. Such networks should combine multilevel tower observations with remotely sensed data from precipitation radars, wind profilers, and RASS (does this need to be defined? Bowling: yes, I don't know what it is) systems.

3. Understanding the effect of aerosols on precipitation, including suppression (Rosenfeld 2000; Givati and Rosenfeld 2004), enhancement (Hobbs et al. 1970; Cotton and Pielke 1995), and increased lightning activity (Orville et al. 2001). Additional coordinated field studies building upon the Houston Environmental Aerosol Thunderstorm (HEAT) project (R. Orville, personal communication) and substantial enhancements to predictive modeling are needed to improve the understanding of the effect of aerosols under specific meteorological conditions for different cities.
4. The development of improved multiscale (i.e., global, mesoscale, and urban scale) hydrometeorological models for improved analysis and prediction of how urbanization alters urban hydrologic processes.

Surface

Physical System

An urban surface water system is an intricate, highly regulated network of rivers, streams, lakes, ponds, pipes, channels, junctions, diversions, bypasses, pumps, regulators, storage and treatment facilities and other hydraulic complexities that collect, convey, store, and treat water. Streams, rivers, and other natural water bodies in an urban environment have been highly modified from their original state by channelization, flow diversions, and other hydraulic modifications. Furthermore, traditional riparian corridors and buffer zones are typically replaced with housing, parking lots, roadways, and other urban structures built directly adjacent to the waterway.

Stormwater surface runoff generated in urban areas during rainfall events may be conveyed in combination with sanitary wastewater to a treatment facility (a so-called combined-sewer system). Combined-sewer systems originally were created during the nineteenth century when connections from failing cesspools and privies were made to existing stormwater drainage pipes (Melosi 2000). Alternatively, the system may be designed with separate conduit systems to convey wastewater and stormwater (i.e., separate-sewer system). In the U.S., most drainage systems constructed recently have been separate, but approximately 800 cities and 40 million people continue to be served by combined-sewer systems. Both combined- and separate-sewer systems contribute discharges to the urban hydrologic system and impact the receiving water body hydrology and ecosystems.

Impacts

The nature of the impacts of channel modification, diversions, and contaminant inputs depends on the characteristics of the urban area, the size, intensity, and frequency of precipitation events, the size and characteristics of the receiving water (e.g., degree of mixing and dilution and assimilative capacity), and the beneficial uses of

the receiving water (e.g., water supply, recreation, ecological habitat). As the physical structure of the hydrologic system is modified by urbanization, so are the magnitudes and qualities of associated fluxes and reservoirs. Prior to urbanization, dry weather streamflow is sustained by surrounding groundwater aquifers. However as urbanization progresses, anthropogenic inputs to and diversions from streamflow increase, and in ultra-urban environments the dry-weather surface flow may be reduced significantly and/or dominated by point source discharges including effluent from municipal and industrial treatment facilities and nuisance flows from the stormwater drainage system (e.g., excess irrigation water, inappropriate discharges, cross-connection flows).

Increased flashiness of flow is characteristic of urban hydrologic systems. The removal of trees and vegetation during urbanization reduces interception capacity, thereby increasing the amount of rainfall reaching the ground surface. The compaction of soil during construction (e.g., Pitt et al. 2003) and the introduction of impervious surfaces (Scheuler, 1994) reduce depression storage, infiltration and groundwater recharge. These altered surface characteristics combined with the introduction of efficient drainage systems, increase the drainage density of urban catchments, which decreases the travel time of overland flow to the drainage system and thus augments the frequency of higher magnitudes of stormwater runoff rates and streamflow (e.g., Booth and Jackson 1997).

The concern with stormwater discharges has traditionally been downstream flooding, channel erosion, and physical damages to drainage infrastructure. Flood control strategies historically involved channelization of streams and rivers in urbanizing environments. The elevated high flows during wet weather are a concern not only from a flood control and geomorphology perspective, but also for habitat alterations and aquatic life impacts. Increased frequency and duration of elevated discharge rates in urban streams (e.g., Hollis 1975) contribute to accelerated stream bank and bed erosion (e.g., Hammer 1972), resulting in physical habitat destruction (e.g., EPA1997).

Increased erosion increases turbidity in the water column, thereby reducing light penetration, visibility, fish gill clogging, burial of bottom-dwelling flora and fauna, and a resulting shift of macroinvertebrate and fish species and density (EPA 1977). In addition, sediment deposition fills impoundments, complicates water treatment, and reduces aesthetic quality of water bodies all of which have tremendous economic ramifications. Removal of the riparian corridor also eliminates tree and vegetation canopy coverage over waterways resulting in increased in-stream temperatures (Brown and Krygier, 1970) and effects on aquatic life. Klein (1979) documented the reduction in macroinvertebrate diversity in urban streams in Maryland and noted the impact to be present when watershed imperviousness exceeded 10-15%. Numerous studies have since represented urbanization by percent imperviousness and corroborated the 10% threshold of imperviousness as the critical condition for stream impacts (e.g., Booth and Jackson 1997).

The stress that increasing populations impart to the quality of surrounding water resources is demonstrated by the findings of Beer (1997), who found that along the Thames river in England, water passes at least six times through people before reaching the estuary. Increases in bacterial and toxic pollutant loading occur in response to urbanization. The transport of wastes carrying disease-causing pathogens

during dry and wet weather to receiving water bodies has the potential to create severe nuisance conditions with dramatic public health ramifications. For example, a study of stormwater discharges to Santa Monica Bay found strong evidence of an increased risk of a broad range of adverse health effects (e.g., fever, ear discharge, gastrointestinal illness) caused by swimming in ocean water close to stormwater outfalls when indicator coliform densities were high (Haile et al. 1996). A range of urban sources of toxic pollutants contribute to receiving waters, including those emanating from urban landscape maintenance (e.g. yards, golf courses, and parks), which has been highlighted in recent years as a significant cause of pesticide contamination of urban waterways and shallow aquifers. The urban use of insecticides and pesticides in the U.S. grew from 86-million kg in 1964 to 300-million kg in 1993 (Barbash and Resek, 1996). Also, numerous emerging contaminants have recently been detected at trace levels in urban-influenced waterways. Drugs used in human medical care may enter the environment via discharges of municipal sewage, hospital effluents, sewage sludges, landfill leachates, domestic septic tanks, and production residues (Verstraeten et al., 2002). Over the counter sales and hospital use of common pharmaceuticals such as ibuprofen and aspirin have been estimated to yield consumption rates of up to 1000 tons/yr in Germany, a figure that rivals the annual use of the most important herbicides that are widely found in surface waters (Verstraeten et al., 2002).

Input of oxygen-demanding substances and nutrients associated with sanitary and stormwater discharges can lower the dissolved oxygen (DO) content in the receiving water to levels harmful to aquatic life, resulting in limited biological diversity, degraded aesthetic quality, and toxicity impacts. There have been numerous studies of the effect of dry- and wet-weather discharges on low DO levels in receiving water bodies (e.g., Keefer et al. 1979), which have found that the response is complicated by downstream processes and the effects of the modified DO on aquatic life is difficult to isolate from the effects of other waste discharges (Field and Pitt 1990). Excessive inputs of nutrients such as bio-available nitrogen and phosphorus can accelerate eutrophication causing increased algae and aquatic plant growth and potentially harmful algal blooms (De Jonge et al. 2002). This may in turn lead to reduced DO levels, toxic conditions, and fish kills. Nutrient pollution has been identified as a likely cause of hypoxia (reduced dissolved oxygen levels) in coastal areas (e.g., dead zone in Gulf of Mexico, Chesapeake Bay) and the dry- and wet-weather contribution of nutrients from urban areas through surface and atmospheric pathways may be significant (Boesch et al. 2001).

State of Remedy

Historical remedies of urban water pollution relied on re-direction of the waste stream. For example, as a remedy to direct sanitary discharges, interceptor sewers were constructed to divert the dry- and wet-weather flows to a strategically located discharge point or a treatment facility. Re-direction solutions merely transferred the problem downstream. In the case of interceptor sewers, they lacked the design capacity to transport the combined sewage during large rainfall events; hence relief structures called combined-sewer overflows (CSOs) were required to allow overflows from the system to prevent damages and disruption of downstream treatment facilities

(Moffa et al. 1997). CSOs have been a source of water quality and ecological problems for decades due to the elevated temperatures of discharges, the high velocities, and the input of bacteria, floatable material, solids, toxic organic compounds and trace metals, oxygen-demanding substances, and nutrients discharged from CSO when in operation (Field and Turkeltaub 1981; EPA 1996). Most of the larger U.S. cities served by combined systems have developed long-term control plans to address their CSO problems. For example, the Metropolitan Sewer District of Greater Cincinnati is expected to spend more than \$1 billion on CSO control (Landers 2004), while improvements to New York City's and Atlanta's wastewater infrastructure have been even more costly.

Most modern sewer systems in the U.S. are designed to convey sanitary flow separately from stormwater (separate-sewer systems). However, groundwater infiltration and rainfall-derived infiltration and inflow (RDII) to the sanitary line during wet weather requires the use of overflow controls called sanitary-sewer overflows (SSOs). SSOs are less of a problem for receiving waters than CSO for flow rate impacts because discharges are not as large. However, SSOs discharge a range of contaminants associated with raw municipal and industrial sewage including disease-causing pathogens, floatable material, solids, toxics, and nutrients (Golden 1995). Remedies for CSO and SSO problems are still needed.

In the U.S., point source dry-weather discharges were the first to be targeted by federal regulations with the passage of the Clean Water Act in 1972. During the 1970s, regulations combined with federal funding initiated much needed improvements to urban wastewater collection and treatment systems. The focus on point source discharges improved the quality of several water bodies that previously had notorious water quality problems (e.g., Cuyahoga River, Lake Erie). However, targeting individual point sources did not eliminate the cumulative effect of myriad controlled and uncontrolled urban discharges. In addition, as control of point source discharges increased, nonpoint source discharges emerged as a significant source of contaminants to water bodies. Recently, watershed-wide planning strategies, the Total Maximum Daily Load (TMDL) program, and the National Pollutant Discharge Elimination Systems (NPDES) Stormwater Program have all been introduced and have potential to protect urban waterways from potentially harmful dry- and wet-weather and point and non-point source discharges. Currently, nonpoint sources of pollution, especially those associated with wet-weather flows, are identified as one of the primary causes of degraded water quality and ecosystems in urban areas of the U.S. (EPA, 1983; EPA, 2002).

The USGS National Water Quality Assessment (NAWQA) Program is the primary source for long-term, nationwide monitoring data characterizing the quality of groundwater, surface water, and aquatic ecosystems with more than 35 urban sites included in the study (USGS, 2004). Results from a decade of NAWQA observations has noted concentrations of phosphorus exceeded desired EPA goal to control plant growth in more than 70% of urban streams sampled, pesticide concentrations exceeded at least one EPA guideline for protecting aquatic life in 93% of urban streams sampled, fecal coliform bacteria commonly exceeded recommended standards for water recreation (Hamilton et al. 2004; Coles et al. 2004). Decades of research has shown

that the quality of stormwater discharges even without sanitary input is sufficient to degrade receiving water quality and ecosystems (e.g. [Burton and Pitt, 2004](#)). The most recent national water quality report to Congress ([EPA, 2002](#)) identified urban runoff-storm sewers as a leading source of impairment of lakes, ponds, reservoirs and estuaries. This attention to stormwater has led to the proliferation of regulatory actions and best management practices as remedies. Realization of the impact of channelization has initiated efforts to naturalize fluvial systems in urban areas to enable them to once again support healthy, biologically-diverse aquatic ecosystems ([Wade et al. 2002](#)). Construction sites represent one of the more significant sources of suspended sediment discharges to water bodies ([Wolman and Schick 1967](#)) and are currently being targeted by the U.S. Environmental Protection Agency for possible effluent guidelines ([Pritts et al. 2001](#)).

Areas of Need

Runoff prediction has been a concern for centuries. However, existing approaches use approximations to represent evapotranspiration and infiltration in urban environments, and furthermore do not account for anthropogenic inputs (e.g. landscape irrigation, leaky water systems) that affect antecedent moisture conditions. There is a lack of physically-based process descriptions derived from experiments in urban-influenced environments. For example, infiltration through pavement (e.g., [Ragab et al. 2003](#)) and infiltration through disturbed urban soils ([Pitt and others](#)) requires that infiltration processes be represented with empirical or physically-based methods rooted in experiments performed on these media. Accurate representation of these components of the urban hydrologic system accounting for the heterogeneity of urban environments is needed to allow development of runoff prediction methodologies that are less specific to a certain system and less dependant on calibration to a particular site. This is especially the case for the future integration of surface hydrologic and hydraulic models with atmospheric and subsurface models to investigate the complete hydrologic cycle in urban environments.

More investigations are needed of combinations of climate, development pattern and age, topography, and geography from an urban water perspective before, during, and after urbanization. Observation networks spanning the urban-rural interface are needed to collect short-term intensive and long-term hydrologic information during periods of urbanization to improve process description in models and provide critical information to guide decisions of urban water planners, designers, managers, and policy makers. In particular contexts, many of the public health, aquatic life, aesthetic, and socioeconomic impacts caused by dry- and wet-weather surface discharges are fairly well understood scientifically, and further improvements to correct these problems is in many ways dependent on overcoming institutional challenges by linking science and policy more effectively. There is a need to build from knowledge based on particular contexts to overall understanding of the urban water cycle integrating climate, topography, geology, ecology, and hydrology for various development types.

Long-term, continuous monitoring to track the flows through the urban environment is needed. This continuous monitoring can provide feedbacks to real-time control

systems for optimal management of the urban drainage system in addition to collecting valuable data for scientific studies. Continuous monitoring of relatively small isolated stream reaches with single or limited outfalls must be expanded by imbedding single outfall observations with downstream observations in a nested arrangement to characterize the cumulative effect of continuous stretches of hydrologic modification and contaminant inputs. Even the densest presently existing network of rain and flow gauges provides only a partial view of the hydrologic response to precipitation forcing in urban systems. Radar and other remote monitoring systems provide spatially and temporally continuous remote sensing observational tools that are needed for specific real-time urban water management. These tools remain in need of improvements in order to account for the heterogeneity of the system. Additional development and utilization of remote sensing and advanced portable sensor systems is required to monitor movement and storage of water and constituents within the urban land surface water system.

The uncertainty of urban hydrologic modeling efforts is compounded by the need to represent the surface with high-spatial-resolution and high-fidelity on a seasonal basis. Further enhancement to urban hydrologic modeling will rely on improved techniques to collect and process topographic and structural data necessary to describe flow pathways. One data source currently being explored is airborne Light Detection and Ranging (LIDAR) data. LIDAR-derived digital elevation models (DEMs) have successfully been used to describe the bare-earth topography in hydrologic models to produce more accurate hydraulic simulations (e.g., [Bates et al. 2003](#)). Another source of data being explored for urban hydrologic modeling applications is multi-spectral data collected from satellite platforms. Processing remotely sensed multi-spectral data has been successfully used to derive surface cover properties for urban hydrologic applications, e.g., impervious surfaces and land cover type ([Ha et al. 2003](#)). Further exploration of remote sensing instruments and data processing approaches are needed to produce continuous near real-time spatial data coverages of soil moisture, streamflow stages, vegetation coverage and roughness, and other hydrologic properties at the resolution necessary for urban hydrologic modeling (~10 m) for ingestion into urban hydrologic modeling or to set initial and boundary conditions.

Simulating surface hydrologic processes using physically-based, conceptual, or empirical models usually is based on model parameters selected to represent one or a combination of physical watershed characteristics, e.g., land use/land cover, soil type, slope. However, the relationship between physical watershed characteristics and hydrologic model parameters is uncertain. This uncertainty is currently being addressed in the Model Parameter Experiment (MOPEX), which was specifically designed to improve the techniques used to prepare *a priori* estimates of parameters used in basin-scale hydrologic models and land surface parameterization schemes in atmospheric models ([Hogue et al., 2004](#)). However, there remains a need for development of improved relationships between surface characteristics and model parameters for urban conditions (e.g., compacted soils, landscaped areas, impervious areas). A coordinated observational-modeling effort amongst numerous case study locations is needed to strengthen the knowledge of relationships between surface characteristics and hydrologic model parameters in urban environments.

If the goal is to forecast future outcomes in urban systems, one must understand the basis for land use decisions made by industries and households. Currently, economic forecasting is limited by difficulties in projecting supply and demand into the future, particularly in the face of new public policies (e.g., minimum lot sizes, provision of public services, road construction, inner city restoration, tax incentives for home ownership, and gasoline taxes), and by difficulties in producing spatially explicit predictions at scales relevant to water and ecosystems scientists. Models developed to produce probabilistic forecasts at a spatially explicit scale are still in their infancy (Nilsson et al., 2003).

Urban streams are known to have poorer water quality and more degraded aquatic life than non-urban streams. However, the cause-effect relationship has not been adequately defined. Ecological effects may be due to a single stressor or the synergistic effects of multiple stressors. Some studies have concluded that the physical modification and hydrologic changes of the water body may be more important than chemical contaminant inputs for determining ecological impact for specific water bodies (e.g., Pitt and Bissonette 1984). There is a need to establish the relative importance of physical modifications to water bodies, hydrologic modification by diversion and input, chemical contaminant and sediment inputs, temperature changes, and other stressors that are suspected of degrading the biological health of receiving waters.

The dry weather or “low flow” regime is a major governor of the structure of aquatic communities (Nilsson et al., 2003). Estimating the magnitude, duration, and future changes in flows is required in order to make accurate forecasts of the future of flowing water ecosystems. However, existing models that are used to determine the in-stream flows needed to support populations of aquatic organisms are mostly focused on fish, and they are designed for use in streams regulated by dams, not for streams dominated by diffuse, multiple source inputs of water throughout the watershed, as occurs in urbanizing watersheds (Nilsson et al., 2003). Furthermore, relating the magnitude and duration of low flow events to urbanization is difficult, since existing data collection is geared toward protection of engineered structures from high flows (Nilsson et al., 2003).

The recent surge in urban channel restoration activities in response to the understanding that historical channelization reduced the quality of the aquatic ecosystems requires much improved understanding of the relationship between geomorphology, ecology, and hydrology in various fluvial-urban-climate systems. The lack of this understanding limits planning and design of restoration strategies. In order to develop scientifically-based guidance for restoration strategies larger numbers of long-duration surveys are required of habitat and aquatic life for different geomorphologic and development states.

Related to the above is the fact that ecosystem health is related to extreme events (flow magnitude and quality). Extreme events in receiving water bodies are generated by efficient centralized stormwater collection systems. There currently is an ongoing shift from the centralized urban water management paradigm where water demands for all uses are met from the urban distribution system and wastewater and stormwater are concentrated into larger flows to be handled at downstream locations toward a decentralized approach utilizing hydrologically-functional landscapes in which drainage, collection, and possibly treatment are each addressed at the neighborhood or lot scale.

Past demonstration projects have shown the value of innovative lot-scale urban water conservation and reuse (Foster et al. 1988). The use of on-site infiltration approaches to stormwater control successfully demonstrated in Europe and Japan have recently been introduced as part of alternative urban development approaches in the U.S. (e.g., low-impact development, smart growth) (Prince George's County, 1999). The potential impacts from proliferation of alternative development practices implementing decentralized stormwater (and possibly wastewater) management need to be investigated to better understand the hydrologic budget at this local scale and to determine the aggregate effects of these practices at larger scales. Continuous monitoring of on-site water management at the lot scale is needed to determine the unintended effects of wastewater recycling, water reuse, and on-site wet weather flow management. Beyond the water budget, there is a need to consider the relative economic and environmental performance of separate-sewer systems versus combined-sewer systems in new developments that practice on-site strategies to manage non-extreme flows.

Subsurface

Groundwater Recharge

Urban water managers make decisions regarding water use in a complex environment influenced by water providers, regulators, non-governmental organizations, and the public. Further complication results from poor understanding of the volumes, fluxes, residence times, and pathways of water and its dissolved and suspended constituents. This is especially true for the subsurface component of the overall water budget, which is of course a reservoir that is not directly observable. Groundwater is becoming an increasingly important component of urban water budgets nationwide (Showstack, 2004). Increased groundwater withdrawals are likely in the future as surface water sources are nearly fully developed.

Groundwater use is not strictly an urban effect, however, continued conversion of agricultural to municipal water rights consequent to population growth leads to the reasonable expectation that groundwater management will increasingly emerge as an urban issue. Urban water managers require the benefit of knowledge of rates of water recharge to the subsurface in order to effectively manage this resource. Water managers lack the information necessary to evaluate the rate at which groundwater resources are being utilized relative to their replenishment. Estimates of recharge rates of water to the subsurface have been shown to decline as a function of the amount of information available. For example, in the Salt Lake Valley, total recharge was estimated to be 369,000 acre-ft/yr by Hely in 1971, 346,000 acre-ft/yr by Waddell et al. in 1987, and 317,000 acre-ft/yr by Lambert in 1995. A similar decline has been noted for the Rio Grande Basin near Albuquerque.

In the western U.S. much of the recharge to the subsurface is ultimately derived from the surrounding mountains, yet the subsurface volumes, fluxes, residence times, and flowpaths of water in the mountain-basin system are poorly understood. Hydrologic studies of mountainous terrain have traditionally focused on surface water, and typically

assume that groundwater storage and flow in the bedrock underlying mountain catchments is negligible. Basin-scale modeling studies often treat groundwater as a residual term, on the basis that it is a negligible component of the water budget. However, recent studies suggest that bedrock groundwater may be an important component of mountain hydrologic systems. Because of rapid development in many mountain areas, greater utilization of mountain water resources, and increasing pressures on basin aquifers potentially recharged by mountain groundwater, there is a growing need to improve our understanding of the occurrence, storage and flow of groundwater in mountainous terrain.

Urbanization affects the groundwater component of the water budget not only through utilization of water, but also through alteration of the quantity and quality of water that is recharged to the subsurface, and this is an important issue for urbanizing areas across the nation. Hence, urban water managers must manage a system that is itself altered by urbanization, and is therefore a moving target. Unfortunately, the magnitude of these alterations is very poorly constrained.

Quantity of recharge to subsurface

Studies have shown that in most urban environments, recharge rates to the subsurface increase due importation of water to the urban footprint, and leakage to the subsurface underlying the urban footprint. Underneath the urban hydrologic footprint, the effects of urbanization are similar to the effects of karstification (Sharp et al. 2003). Water mains and sewer systems may develop leaks after construction and provide pathways for enhanced recharge (Foster 1996; Yang et al., 1999), and irrigation in wealthy urban areas may contribute significantly to recharge to the subsurface (Al-Rashed and Sherif, 2001). Determination of these pathways is prerequisite to determination of groundwater utilization rates, and forecasting of low surface flows based on land use change (Nilsson et al., 2002). However, quantifying recharge in the urban footprint is problematic with very few reliable estimates currently available in the literature (Lerner, 2002).

Roof runoff to soakaways and infiltration boreholes or basins designed to dispose of storm runoff are unique to the urban environment (Price 1994.) Although only documented in a limited number of studies, these sources may be 30% to 50% of total recharge to a basin, with leaking water mains typically being the largest of these sources (Lerner, 2002).

In urban areas, infrastructure associated with underground piping can be extensive and can induce significant influence on groundwater flow patterns and rates. Lerner (2002) reports that in Nottingham UK about 50 km of water carrying piping exists per square kilometer of urban area. If the average width of trenches constructed to accommodate this piping were 1 m, then 5% of the urban footprint has a highly modified permeability structure. Such complexity makes the application of traditional techniques (e.g. the application of Darcy's Law via a groundwater flow model) very difficult and prone to errors.

Quality of recharge to subsurface

Expansion of the urban footprint often precedes expansion of the municipal sewage collection system. Expanding subdivisions are often initially constructed with septic systems to reduce costs, leading to enhanced downward flux of anthropogenic pollutants, and resulting in water quality degradation in underlying shallow aquifers. In the intervening decades prior to expansion of sewage collection systems, many western cities have discovered elevated nitrates and pharmaceuticals in shallow groundwater (e.g., Seiler et al., 1999). In semi-arid and arid areas, septic systems may provide the vast majority of ground water recharge (Seiler, 1999).

Shallow groundwater will likely be increasingly viewed as an important resource as urban water managers seek to maximize supply. Although market forces are unlikely to eliminate threats to shallow groundwater quality, it will be important that water resource managers are provided with the understanding needed to effectively manage these impaired water resources. To date, little monitoring of these resources have been undertaken.

In an effort to maintain the quality of groundwater supplies in urban areas, well-head protection studies are commonly performed. Strategies for well-head protection have been based largely on the concept of capture zones that contain a specified travel time (e.g. 15 years). Delineating these capture zone is typically accomplished by flow-path modeling with very limited site-specific data. Furthermore very little calibration or validation of modeled capture zones is performed.

Riverbank Filtration

Occasional failures of water treatment systems occur even in industrialized nations, e.g. the 1993 outbreak of cryptosporidiosis in Milwaukee, Wisconsin infected 403,000 people and resulted in 70 fatalities (MacKenzie et al., 1994). . The problem is by no means strictly urban; as of the year 2000, half of U.S. drinking water wells tested had evidence of fecal contamination, and an estimated 750,000 to 5.9 million illnesses per year are estimated to result from contaminated groundwaters in the U.S. (Macler and Merkle, 2000). Although the issue of water supply contamination by pathogens is not restricted to urban settings, the issue is of great importance to cities, where the ability to increase water supply without appropriation of new sources will depend on the ability to treat impaired sources economically.

Riverbank filtration is an emerging water treatment technology that promises partial, or in some cases complete, treatment of impaired water (Ray et al., 2002). Riverbank filtration is fundamentally a hydrologic process, since it relies on naturally occurring contaminant removal mechanisms (e.g. adsorption of dissolved constituents, filtration and straining of suspended constituents) operating in natural riverbank materials during transport from a river to a receiving well. During riverbank filtration, it is crucial to ensure a sufficiently high rate of attachment (and low rate of detachment) of suspended constituents (e.g. protozoan, bacterial, or viral pathogens) during transport to water supply wells, as reviewed by Schijven et al. (2002).

Hydrologists are presently able to predict colloid/microbe attachment rates during transport in porous media under conditions where colloids/microbes experience attraction only to the porous media grain surfaces (so-called favorable deposition conditions). However, in natural porous media, a repulsive component always exists in colloid-surface interactions, confounding our ability to predict the transport of colloids/microbes in groundwater aquifers and riverbank materials. The results obtained from both laboratory and field studies simply do not fit theory. Theory predicts log-linear decreases in microbial concentrations with distance from source, whereas experiments show that such distributions mostly do not apply in environmental media (e.g. [Li et al., 2004](#)). Furthermore, very little is known about the controls on colloid/microbe release from sediment grain surfaces back into groundwater, other than that it does occur at relatively slow rates that are potentially significant over long periods (weeks to years or more) ([Zhang et al., 2001](#)). Attachment and release of biological contaminants is also complicated by the potential influence of biological activity on transport (e.g. via motility, changes in surface properties with metabolic activity, cell division mediated transport, etc., as reviewed in [Murphy et al. \(2000\)](#)). The situation is further and greatly complicated by the potential for water-rock interaction during transport, which affects not only the aqueous chemistry that governs physicochemical aspects of attachment and detachment, but also the redox and nutrient conditions that govern biological activity ([Murphy et al., 2000](#)).

Because riverbank filtration wells are typically placed in relatively coarse sediments in order to maximize rate of groundwater recovery, the riverbed itself (typically a mixture of coarse and fine materials) is considered to be an important first line for removal of dissolved and suspended contaminants in the river water ([Gollnitz, 2002](#)). However, the thickness of riverbeds is temporally variable, especially during floods, hence understanding changes in the permeability and thickness of riverbeds as a function of flood stage is important to maintaining water quality during riverbank filtration. As well, contaminant loading into rivers increases during flood, compounding the difficulty in maintaining quality of riverbank filtered water.

Aquifer Storage and Recovery

The increasing demand for water by urban centers will increasingly require storage of water during times of excess supply for subsequent utilization during times of need. The large evaporative losses associated with surface reservoirs, as well as the growing public unpopularity of dams, renders underground storage of water supplies appealing. Engineered infiltration or injection of water resources into the subsurface is termed aquifer storage and recovery (ASR). This technology is operating at hundreds of sites around the U.S., and is accepted as a proven, cost-effective means for seasonal or longer term storage of water that meets federal primary drinking water standards.

Initial development and conditioning of the storage zone around an ASR well is usually required in order to develop a buffer zone separating the stored water from the surrounding native groundwater ([Pyne, 2002](#)), which typically exceeds at least one of the federal primary drinking water standards. Once this initial development is

completed, the quality of the recovered water is usually suitable for drinking following disinfection.

ASR is site specific since each location sits within a particular hydrogeologic location, and is subject to different water management opportunities, constraints, water laws, regulations, policies, and responsibilities and expertise of state agencies.

With the success of ASR has come consideration of whether this technology could be used to store reclaimed water (treated wastewater) (Pyne, 2002). A significant barrier to the recharge of reclaimed water for ASR purposes is the lack of credit or allowance for water quality changes that occur once the water reaches a saturated aquifer. As a result, regulatory agencies have generally taken the position that the quality of water recharged to the aquifer must meet all drinking water standards at the well head prior to recharge. This regulatory position is rooted in the Underground Injection Control (UIC) program, the regulations of which are aimed at controlling the disposal of waste materials into wells (Pyne, 2002).

Concerns with treated wastewater include the presence of viruses, bacteria, and protozoa. Viruses and bacteria are inactivated rapidly at temperatures above 20 °C, but can persist for more than a year at temperatures below 10 °C. Per distance rates of virus removal vary greatly with subsurface setting, and the rate of virus and bacteria removal has been shown in many cases to decrease with distance of transport (e.g. Li et al., 2004). Disinfection processes used for treating microbes may generate by-products such as trihalomethanes (THMs) and haloacetic acids (HAAs), some of which are considered carcinogenic (Pinholster, 1995). Furthermore, it is estimated that only 10-20% of the organic content of treated wastewater is identified (Pinholster, 1995), and so may not be properly evaluated in terms of regulatory standards that protect human health. Studies performed at ASR operations have indicated that HAAs degrade rapidly (days) in aerobic zones of the subsurface, whereas THMs degrade over periods of weeks in anaerobic zones if they are present (Pyne, 1996).

Mobilization of trace metals such as As, Hg, and U are concerns that arise from the geochemical equilibration that occurs upon injecting allochthonous water into the subsurface (Arthur et al., 2002). Water quality changes during ASR operations are significant, and are driven by geochemical and biological processes that need to be better understood in order to assess the degree of water quality improvement that can be expected during storage, in order to allow clear evaluation of the viability of recharging reclaimed water during ASR operation.

The surface re-use of treated waste water, e.g. grey water recycling and use of reclaimed water, e.g. for irrigation of residential lawns, golf courses, and parks in urban areas, also may slowly increase the concentrations of natural constituents, as has occurred in Israel with increases in boron concentrations in shallow ground waters. Research is needed to determine potential inadvertent effect of these conservation practices on soil and shallow groundwater quality.

Subsurface heterogeneity complicates our understanding of dissolved and suspended contaminant transport in subsurface materials, including riverbanks, as described above in the context of riverbank filtration. Riverbank filtration development relies on an estimated travel time from the river to the well that is based on average

behavior of contaminants in the system. That flow in the subsurface is not uniform, but rather follows ubiquitous preferential pathways is well demonstrated. Preferential flow paths are ubiquitous in the subsurface; and the magnitudes of these effects depend on scale, e.g., they result from the geologic framework at the basin scale, non-homogeneity of lithologic facies and anthropogenic alterations at the site scale, and grain packing irregularities or variation in fracture density at the core scale.

Areas of Need

Developing tools to quantify “natural” groundwater recharge is a critical research need, but of particular importance to the urban environment is quantifying changes in recharge due to urbanization. Utilizing a traditional water balance approach is particularly problematic because (1) storage volumes are inherently large relative to recharge fluxes, (2) the spatial distribution of the relevant storage parameter (e.g. specific yield or storativity) must be known but is typically uncertain, (3) spatial patterns in urban recharge can be highly variable due to specific subsurface sources (e.g. leaking water mains) and a highly altered permeability structure. Furthermore, decades are often required to reliably ascertain changes in storage using traditional water balance approaches, and yet effective urban planning requires a rapid evaluation.

The transient nature of the sources of urban recharge (especially leaking water mains) and their complex spatial distribution require hydrologic tools that are capable of large spatial coverage while maintaining a high resolution in both space and time. For example, satellite-based topographic surveys (to detect changes in land surface elevation resulting from enhanced groundwater recharge) or high-resolution microgravity methods (to detect local changes in the mass of the earth due to recharge) are potential tools that need further development and refinement. Such surveys may be particularly important in the urban environment because of the intense variability and transient nature of recharge.

Tools are needed that provide an effective integration of highly complex sources. For example, a solute mass balance between “natural” and “urban” sources has sometimes been capable of locating and quantifying increases in recharge due to urbanization. When water imported from outside the urban watershed has different chemical or isotopic signatures from sources within the watershed, mass balance can quantify each source. Further refinement and development of tracer techniques is needed.

Industrial and domestic activities within the urban footprint lead to a wide range of potential sources of groundwater contamination. In order to assess the impacts of these activities on groundwater quality, the distribution of sources must be overlain on the distribution of recharge rates and then evaluated in the context of biogeochemical reactions and physical transport mechanisms. Because all of these elements (distribution of both contaminant and water sources along with transport processes) are highly uncertain, forecasting changes in water quality in urban aquifers is fundamentally problematic with existing tools and data.

Improved strategies are needed that are based on measurable quantities. For example, rather than basing well-head protection on flow path modeling with limited site-specific data, it might be possible to base protection schemes around an understanding of groundwater travel times (as given by groundwater dating techniques).

In addition to refinement of new techniques, greater attention needs to be paid to long term monitoring of shallow water resources (e.g. effects of recycling, irrigation efficiency, nitrogen uptake, boron, selenium, and salinity), via networks of vadose zone flux meters and shallow groundwater wells to determine volumes, fluxes, residence times, and pathways of water and its dissolved and suspended constituents. Increased focus is needed on the proliferation of septic tanks in developing areas. Tracers of increased urban recharge need to be developed that can be used to date and track recent recharge and to assess its availability as a resource.

Research is needed on low cost treatment options for shallow ground water in urban areas. In the future we will need to develop methods to more effectively use this source of impaired (slightly contaminated) water. Examples, of treating shallow urban groundwater may include filtration, flocculation, or reverse osmosis or some combination of these methods plus others. Treatment methods will produce their own contamination or disposal problems that need to be considered in treating and managing this resource.

The challenges described above in developing an understanding of the processes governing the fluxes of dissolved and suspended contaminants in the subsurface are highly problematic for determination of well setback distances from contaminant sources (including development of riverbank filtration systems), and estimation of downward fluxes of colloids/microbes resulting from grey water recycling and long term aquifer storage and re-use. Accounting for these complications requires knowledge from dissolved and suspended contaminant transport studies in natural media at scales beyond the size of a packed column (still most commonly used due to tractability for single investigators). Flow cells are needed at scales ranging between 10s and 100s of meters, with densely arrayed monitoring points in 3 dimensions in frameworks that are intensively characterized (hydrologically, geologically, geophysically), and that are well administrated. Development of well instrumented, well characterized, and staffed flow cells is presently beyond the capabilities of academic researchers. The lack of these resources severely limits the rate of progress of understanding of transport and fate processes in the subsurface.

Previous attempts at such sites have not achieved a useful degree of prior site characterization (Cape Cod - USGS), have been terminated due to program elimination (Oyster, VA - DOE NABIR), or are devoted to a relatively narrowly-focused group of researchers (ORNL NABIR site - DOE NABIR). Addressing the problem is not simply a matter of developing a limited number of well-instrumented and well-characterized flow cells, since the variety in subsurface environments is too great to be captured by a limited set. Furthermore, well-instrumented and well-characterized flow cells also have limited life times if non-conservative tracers are used. Therefore, to me this need, infrastructure and personnel centers are needed for external deployment of geophysical (e.g. seismic tomography, resistivity, e.g. [Hubbard et al., 2001](#)), geochemical, and other characterization techniques for characterization of subsurface sites. As well, support is

needed for development of inexpensive and non-intrusive multi-level subsurface sampling infrastructure. This technology would allow the development of a large number of well-characterized and well-instrumented flow cells that would greatly enhance understanding of the processes governing fluid flow and contaminant transport in the subsurface.

The emergence of large scale monitoring technologies warrants special attention. A new era of aquifer monitoring is now possible using a combination of remote sensing tools, including the global positioning system (GPS), interferometric synthetic aperture radar (InSAR), and repeat high-precision gravimetric techniques. GPS data have proven valuable for determining the three dimensional position of a point anywhere on the Earth almost instantaneously and can be used in two ways to monitor vertical displacements of the ground surface associated with deflation and inflation of confined aquifers. In the base station mode a GPS receiver is installed at a single position and used to continuously record changes in position. This mode of operation provides a dense time series of points, with a vertical accuracy on the order of millimeters. Thus the temporal response of the ground surface to seasonal inflation and deflation of the aquifer is recorded. In the campaign mode a GPS receiver is used to record spatial variations in the ground surface at discrete points and time. There is a tradeoff between accuracy and time spent at a station, but with an hour of occupation relative elevations are precise to approximately 2 cm.

Synthetic aperture radar (SAR) uses the amplitude of microwave pulses reflected from the ground surface. InSAR uses the difference in phase between two repeat-pass scenes to effectively measure the difference in path length. Using knowledge of the distance between satellite positions during the repeat-pass (the baseline distance) and the effect of topography, changes in ground surface elevation at the sub-centimeter level can be determined over large spatial areas. Under favorable conditions this technique is capable of resolving temporal ground surface elevation displacements of ~ 1 cm. This technique can provide seasonal snapshots of ground surface elevation changes at the seasonal scale, given favorable satellite passes. GPS surveys can ground-ruth displacements inferred from InSAR, and together these surveys can provide the temporal and spatial extent of ground surface inflation and deflation associated with aquifer recharge and discharge.

Observations of temporal variations in gravity, when combined with independent estimate of changes in ground surface elevation, are an effective and economical method for monitoring subsurface changes in mass. Gravity measurements at the Earth's surface can now be made to approximately 5 μ Gal. If the entire aquifer can be monitored, the total mass change can be calculated from the total gravity change. In groundwater applications, temporal changes in gravity are usually dominated by changes in pore saturation, whereas spatial change may include the effects of varying pore saturation, porosity, and indirectly from the permeability and the pressure regime.

With launch of NASA's gravity recovery and climate experiment (GRACE) which includes two satellites in tandem it is possible to monitor gravity changes over large aquifers from space. Using precise measurements of the distance between the two satellites, repeat satellite passes will yield estimates of gravity changes over the Earth, and changes in water storage can be estimated. With estimates of soil moisture and

atmospheric pressure, seasonal and monthly groundwater storage changes can be estimated. Because of the altitude of the satellites, aquifers of areal extent 200,000 km² or larger are amenable to characterization, e.g. the Great Basin.

Specific questions that could be addressed using a combination of remote sensing technologies include 1) what is the seasonal magnitude of exploitation of these aquifers? Is seasonal exploitation causing long-term deflation of basins and loss of storage? 2) What are the spatial patterns of seasonal and long-term inflation or deflation of these basins? 3) What are the elastic and inelastic responses to pumping in time and space? 4) Does the aquifer behave as a single basin or individual sub-basins? 5) Is subsidence limited by basin bounding faults?

Hydrologic Science and Water Management

One of the challenges for the future in optimizing water management is increased communication between scientists and water decision makers. Scientists performing research in the hydrologic arena would benefit from knowledge regarding how decision makers that influence the urban hydrologic system receive their information. Urban decision makers routinely make non-hydrologic decisions that have important hydrologic consequences, and scientists routinely investigate hydrologic processes that are not recognized as carrying important implications for water managers. In order to envision the future of urban-affected watersheds, researchers should know how urban system decisions are made, specifically: Who plays what role in water-related decision-making? What water-management-relevant information that is now lacking will be needed for decision-making over the coming decades?

Decision-making within the water management arena includes personal choices, management policies, corporate decisions and institutional or agency decisions made within a complex interplay of regulations, competing interests, and hydrologic contexts. Although often not explicitly considered in scientific hydrology research, these decisions can have important impacts on the way that hydrologic systems operate. People who might affect water management policies and decisions include hydrologic scientists, water consultants, lay citizens, government water resource and environment regulators, corporate managers, lawmakers, environmental advocacy organizations, academic researchers, and educators.

Personal choices are largely those made by ordinary citizens who may be unaware of the ways that their choices (both water-related and otherwise) might affect hydrologic systems. Personal choices might include conservation strategies (minimizing landscape irrigation and using in-home water saving technologies in order to save money or to use water more efficiently), residential pesticide and fertilizer use, hazardous waste management practices, living in a compact urban neighborhood (reducing infrastructure construction and maintenance costs), voting for water infrastructure construction bonds (dams, pipelines, pumping facilities, water treatment plants, etc.), or electing officials who make policies and decisions that affect hydrologic systems. The choices of lay citizens may be influenced by information sources with a sound foundation in hydrologic science (advocacy groups, adult education programs,

school programs, the news media, and government extension services) or, by poorly substantiated opinions generated by these, and other, information sources.

Management policies set at the municipal, state and federal levels are developed and passed by elected officials who rely on technical staff and lobbyists for advice, but the officials are often poorly versed in the principles of hydrologic science. Such policies set up the legal and regulatory context that directly, or indirectly, affects various aspects of the hydrologic system (water rights laws, water quality regulations, water supply system requirements, highway construction policies, land management policies, regional metropolitan planning, land development regulations, property taxes, construction and maintenance of water infrastructure, and water use fees). Parties with vested interests in the financial benefits of water quantity and quality management policies include land developers who modify surface water systems and affect groundwater recharge rates, power companies who extract water for power plant cooling, water retailers who extract and redirect surface water and groundwater for sale to other parties, industrialists whose water use leads to reduced water quality, municipalities responsible for providing water treatment services, and agriculturalists who use federally-subsidized irrigation water. These parties can often afford to hire professional hydrologists that help to influence policy making while advocacy groups and lay citizens have fewer resources to draw upon when attempting to influence policy-making.

Water law varies throughout the lower 48 states and is complicated by the fact that water is considered both a renewable, yet finite, natural resource and a common good required for sustaining human life. The following paragraphs are largely based on an outline of U.S. water rights law by [Feldman and Elmandorf \(2000\)](#). States located east of the Mississippi River have generally adopted laws based on riparian common law (landowners next to a river are granted reasonable use of flowing water from a watercourse but the water is owned by the public). More arid states located west of the Mississippi (AZ, CO, ID, MT, NV, NM, UT, WY, AK) use prior appropriation laws (first person to appropriate water has the right to put a specified amount of water to beneficial use). Several states located in less arid regions convey correlative rights with varying blends of riparian and prior appropriation (WA, OR, CA, ND, SD, NE, KS, OK, TX). Laws governing groundwater are more variable with different states conveying absolute ownership, reasonable use, correlative rights (proportional to ownership of overlying land), or prior appropriation. Federally administered water rights include those of tribal nations and water reserved for federally managed forests, parks, recreation areas, etc.

The lack of water in prior appropriation states leads to a relatively high level of litigation, when compared to that of eastern, water-rich states. Thus, important decisions affecting hydrologic systems in western states are often made by judges or juries after reviewing information provided by attorneys and expert witnesses. Hydrologic systems are more likely to be degraded under prior appropriation doctrine because senior water rights holders are legally able to pump rivers and streams dry in order to obtain the water granted for their beneficial use. Increasingly, however, some appropriation states are adopting a more riparian approach to sharing water in order to

preserve in-stream flows needed to support riparian ecosystems. On the other hand, the move towards water marketing or water banking as a method for reducing the impacts of water shortages is most readily accomplished in prior appropriation states where parties have the right to transfer ownership of a specified quantity of water to another party for a specified sum of money. Permits to withdraw water from watercourses for reasonable use in riparian states, however, are neither quantifiable nor transferable because the water is publicly owned.

Adjudication of non-surface water rights (groundwater and atmospheric water) is more variable. Although riparian states originally allowed the landowner to pump whatever groundwater he needed from underlying aquifers, most riparian states limit pumping in ways that attempt to reduce the impact of pumping on adjacent landowners pumping from the same aquifer. In many, but not all, states a close connection is assumed between groundwater and surface water so that excessive groundwater pumping can be limited to protect surface water rights. In the case of atmospheric water, weather modification to enhance rainfall as a consequence of cloud seeding and to suppress hail has led to weather modification regulations and common law that help to address issues that arise when farmers believe that they have been denied the precipitation that would otherwise have fallen on their land (Standler, 2002). Scientific and technical uncertainties regarding the direct and indirect effects of weather modification, however, complicate the ability to resolve questions of public vs private rights to atmospheric moisture (Templar, 2004). Could the impact of megacities on precipitation patterns and rates lead to litigation in the weather modification arena?

In addition to state and federal water law, federal regulations exist that govern safe drinking water standards (Safe Drinking Water Act), contamination cleanup (Resource Conservation and Recovery Act), and pollutant discharges (Clean Water Act). Some states have enacted regulations that extend the federal regulations. Increasingly, the U.S. Environmental Protection Agency and corresponding state agencies are using community engagement processes to learn more about public concerns and help citizens to better understand both the policy-making context and the underlying hydrologic science; while also providing citizens a greater say in the policy- and decision-making process.

Environmental advocacy groups have frequently antagonized those who hold the reins on water management decisions. As a consequence, their concerns are frequently not heard or seriously considered. One advocacy group in Utah (Utah Rivers Council, URC), however, has adopted an innovative and successful strategy for influencing policy making by the Utah State Legislature. Knowing that they have a valuable message to deliver (tax subsidies should be removed from water, dam sites should be eliminated from consideration, strengthen water conservation act), but recognizing that they would not be credible messengers, the URC convened panels of water resources, powerful stakeholders and economic experts to help prepare and present their arguments to the legislature. A strong spirit of cooperation is helping the URC to effect changes in water management policy that are better tuned to recognizing principles of hydrologic science and the needs of both citizens and aquatic ecosystems.

Increasingly, one solution proposed to eliminate water shortages in the western United States involves using water rights originally granted to supply agricultural irrigation water to supply water to urban areas. In addition to generating concerns regarding the socioeconomic and cultural consequences of reducing water for crop production, such water transfers can: (1) significantly modify the hydrologic system by reducing agricultural runoff, (2) change the chemical character of agricultural runoff (explain how, this point is not clear), and (3) modify the timing of water flows in watercourses (water is mostly released from artificial reservoirs for agricultural purposes during the growing season while a constant urban supply is desired all year long). In 2003, the Imperial Irrigation District (IID) of Imperial County, California signed a Quantification Settlement Agreement (QSA) with urban water users elsewhere in California to deliver about 400,000 acre-feet per year, over a 45 year period (with option for 35 year continuation) to the urban areas in order to help reduce California's overly large draw on water from the Lower Colorado River. Accomplishing this goal involves at least 15 years of fallowing a portion of the agricultural land that generates the relatively fresh runoff that supports avian and ichthyian habitats in the Salton Sea. Ultimately, farmers using IID water are expected to adopt conservation methods that reduce their net water use while maintaining the health of the agricultural economy. These efforts will lead to reduced runoff of increasingly saline water and complicate efforts to preserve the Salton Sea by reducing its size and developing management plans aimed to maintaining or reducing current levels of salinity. This water transfer agreement will likely become one of many cases where hydrologic systems traditionally supported by irrigation runoff water lose their water as a consequence of water transfers to urban use. Unfortunately, we lack the understanding needed to fully assess how such agriculture to urban water transfers will affect both hydrologic systems and the socioeconomic circumstances of the affected communities.

All too often policies are developed and decisions are made in the absence of a sound foundation in hydrologic science. In some cases information is lacking because data are sparse, monitoring systems are inaccurate, analytical tools are inadequate, uncertainty is high, or theoretical understanding is incomplete. In other cases, scientists are unable, or unwilling, to help transform what they know into information that can be understood and used by non-specialists. Furthermore, powerful players in the water management arena may intentionally, or unintentionally, misuse technical information to further their own interests. Previous sections of this paper outline areas of research in hydrologic science that can help to suggest preferred courses of action in water management while clarifying the possible consequences of personal choice, policy setting and decision-making. Truly understanding the way that hydrologic systems operate and evolve in urbanizing environments requires greater communication between hydrologic researchers and those involved in decision-making so that the consequences of human activity are fully considered in our efforts to understand and manage hydrologic systems.

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