Urban Groundwater

Ken Howard

THE GROUNDWATER PROJECT

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THE GROUNDWATER PROJECT

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The Groundwater Project Foreword

At the United Nations (UN) Water Summit held in December 2022, delegates agreed that statements from all major groundwater-related events will be unified in 2023 into one comprehensive groundwater message. This message will be released at the UN 2023 Water Conference, a landmark event that will bring attention at the highest international level to the importance of groundwater for the future of humanity and ecosystems. This message will bring clarity to groundwater issues to advance understanding globally of the challenges faced and actions needed to resolve the world's groundwater problems. Groundwater education is key.

The 2023 World Water Day theme *Accelerating Change* is in sync with the goal of the Groundwater Project (GW-Project). The GW-Project is a registered Canadian charity founded in 2018 and committed to the advancement of groundwater education as a means to accelerate action related to our essential groundwater resources. To this end, we create and disseminate knowledge through a unique approach: the democratization of groundwater knowledge. We act on this principle through our website gw-project.org/↗, a global platform, based on the principle that

"Knowledge should be free, and the best knowledge should be free knowledge." Anonymous

The mission of the GW-Project is to promote groundwater learning across the globe. This is accomplished by providing accessible, engaging, and high-quality educational materials—free-of-charge online and in many languages—to all who want to learn about groundwater. In short, the GW-Project provides essential knowledge and tools needed to develop groundwater sustainably for the future of humanity and ecosystems. This is a new type of global educational endeavor made possible through the contributions of a dedicated international group of volunteer professionals from diverse disciplines. Academics, consultants, and retirees contribute by writing and/or reviewing the books aimed at diverse levels of readers from children to high school, undergraduate and graduate students, or professionals in the groundwater field. More than 1,000 dedicated volunteers from 127 countries and six continents are involved—and participation is growing.

Hundreds of books will be published online over the coming years, first in English and then in other languages. An important tenet of GW-Project books is a strong emphasis on visualization with clear illustrations to stimulate spatial and critical thinking. In future, the publications will also include videos and other dynamic learning tools. Revised editions of the books are published from time to time. Users are invited to propose revisions.

We thank you for being part of the GW-Project Community. We hope to hear from you about your experience with the project materials, and welcome ideas and volunteers!

The GW-Project Board of Director,

January 2023

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Foreword

Of the eight billion people on the planet, nearly five billion live in urban areas. Migration to urban areas continues as the rural poor migrate to seek work. Cities are immense users of water because of the concentration of people and industry. Groundwater is often used for all or part of the water supply in cities and has a large influence on how the infrastructure functions and the environmental well-being of the urban area. This book provides a comprehensive view of how and when groundwater is important and describes the benefits and problems of using a systems approach to development.

Urban water management is an immensely complex challenge because many parts of the hydrologic/hydrogeologic system are connected and changes to one can disturb others. Cities that are groundwater dependent must include wells both inside and outside the city proper because a large area is needed to capture the required volume of groundwater, hence the water footprint is large. Cities that depend on rivers or lakes (surface-water dependent) for water supply also draw large amounts of water, thus impacting distant locations. Whether groundwater or surface-water dependent, cities manage a large amount of water in addition to the storm water that needs to escape without causing damage due to flooding. If the groundwater table is too close to the land surface, it will provide too little capacity for subsurface water storage, increasing the likelihood of flooding. Alternatively, in those areas where groundwater extraction is occurring and pumping is suddenly reduced, the water table rise can result in water inflow to basements and underground infrastructure such as parking lots.

A major component of the subsurface water budget is leakage from sewer systems and water distribution pipes. In many cities, the water table elevation is determined by this leakage—and leakage from sewers causes aquifer contamination. In cities dependent on groundwater, too much pumping can cause depressurization of deeper groundwater and result in sea water intrusion and/or land subsidence that can disrupt urban infrastructure. Another consideration when designing infrastructure is that systems that promote infiltration or rainfall can result in both flooding and contamination.

This book describes the complexities that need to be considered for each of these factors when managing urban groundwater. Additional resources that complement the content of this book can be found on the Groundwater Project website including: *Land Subsidence and Its Mitigation*↗ and *Septic Systems Impacts on Groundwater Quality*↗.

The author of this book, Dr. Ken Howard, a professor of hydrogeology at the University of Toronto, began research on urban groundwater in the early 1990s. He has traveled the globe in his quest and is a global leader on the topic.

John Cherry, The Groundwater Project Leader

Guelph, Ontario, Canada, August 2023

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Preface

I have been working on urban groundwater issues for over forty years and am very grateful for this opportunity to draw together many of my observations, findings, and experiences into this single book. There is nothing new or novel to be found here. Much of the material presented has been mercilessly culled from my research papers, book chapters, conference articles, and special publications, many of which were prepared collaboratively with research and industry colleagues throughout the world.

Other material is extracted from articles written by authors I hold in very high regard. I make no claim to be the originator of many of the opinions and ideas expressed here, but they are the ones that I have acquired and honed over the years, and I stand by them.

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Special thanks go to numerous urban groundwater friends, colleagues, and co-authors I have enjoyed meeting and working with these past four decades. One way or another, they have all contributed to this book. At the risk of offending somebody I may have inadvertently missed, they include (with current or former affiliations, and listed alphabetically):

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[1] United Nations Educational, Scientific and Cultural Organization.

[2] International Association of Hydrogeologists

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University of Hong Kong, China; David N. Lerner, University of Sheffield, United Kingdom; Peiyue Li, Chang'an University, Xi'an, China; Steve Livingstone, Geocentric Environmental Incorporated, Canada; John W. Lloyd, University of Birmingham, United Kingdom; Brian Morris, British Geological Survey, United Kingdom; Kent Murray, University of Michigan-Ann Arbor, USA; Dubravka Pokrajac, University of Aberdeen, United Kingdom; Lynda Radke, Australian Geological Survey, Australia; Mike O. Rivett, University of Strathclyde, United Kingdom; Dan Rogers, University of Michigan-Ann Arbor, USA; Dave Rudolph, University of Waterloo, Canada; Ken Rushton, University of Birmingham, United Kingdom; John (Jack) M. Sharp Jr., University of Texas at Austin, USA; Makoto Taniguchi, Research Institute for Humanity and Nature, Japan; Richard Taylor, University College, London, United Kingdom; John H. Tellam, University of Birmingham, United Kingdom; Joanne Thompson, R. J. Burnside and Associates Limited, Ontario, Canada; Kelly Warner, United States Geological Survey (USGS), USA; and Leif Wolf Karlsruhe, Institute of Technology, Germany.

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1. Urban Groundwater and the Urban Sustainability Challenge - The Context

1.1 Global Population Growth and the Growth of Urban Areas

The world's human population continues to increase at an unprecedented rate and much of this growth takes place in urban areas (Figure 1). Between 1990 and the turn of the twenty-first century, the global population grew by 15 percent (from 5.3 to 6.1 billion) while the population of urban areas increased by 24 percent to nearly 3 billion. This represents an urban growth rate of nearly 200,000 people per day.

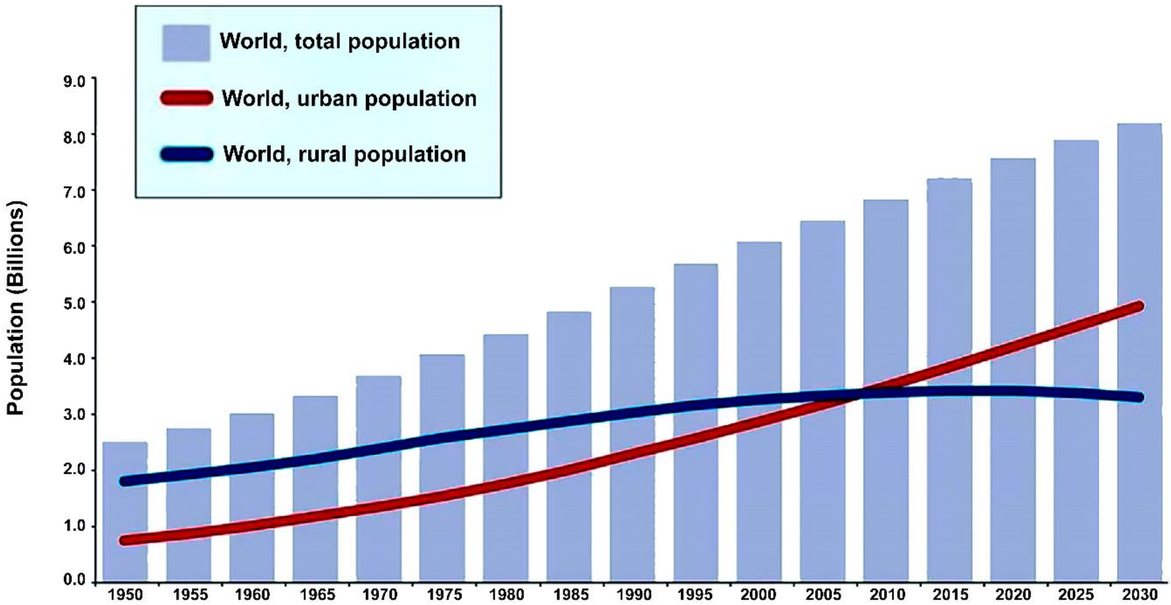

Figure 1 - The urban and rural population of the world, 1950–2030 (modified from the United Nations, 2006).

By 2010, over half the world's population was living in urban areas; by 2030, the number of urban dwellers is anticipated to reach almost five billion—around 60 percent of the projected global population of 8.2 billion. In China, less than 20 percent of the population lived in cities as recently as 1980, while over 50 percent of China's population lived in urban areas by 2011, a figure that analysts predict will increase to around 70 percent by 2030.

By 2050, the world's urban population is expected to increase to 6.3 billion, which would have represented the entire population of the world in 2004. It is expected that virtually all this urban population growth will occur in less-developed countries where the total population of around 2.5 billion in 2009 is projected to rise to 5.2 billion by 2050 (Figure 1). While the simple excess of births over deaths will generate much of the growth, a very significant part will be associated with migration to cities of rural inhabitants who are either

- compelled to move by environmental crises and political conflicts (World Water Assessment Programme, 2009); or

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- attracted to urban areas in the hope of finding work, alleviating poverty, and improving living conditions.

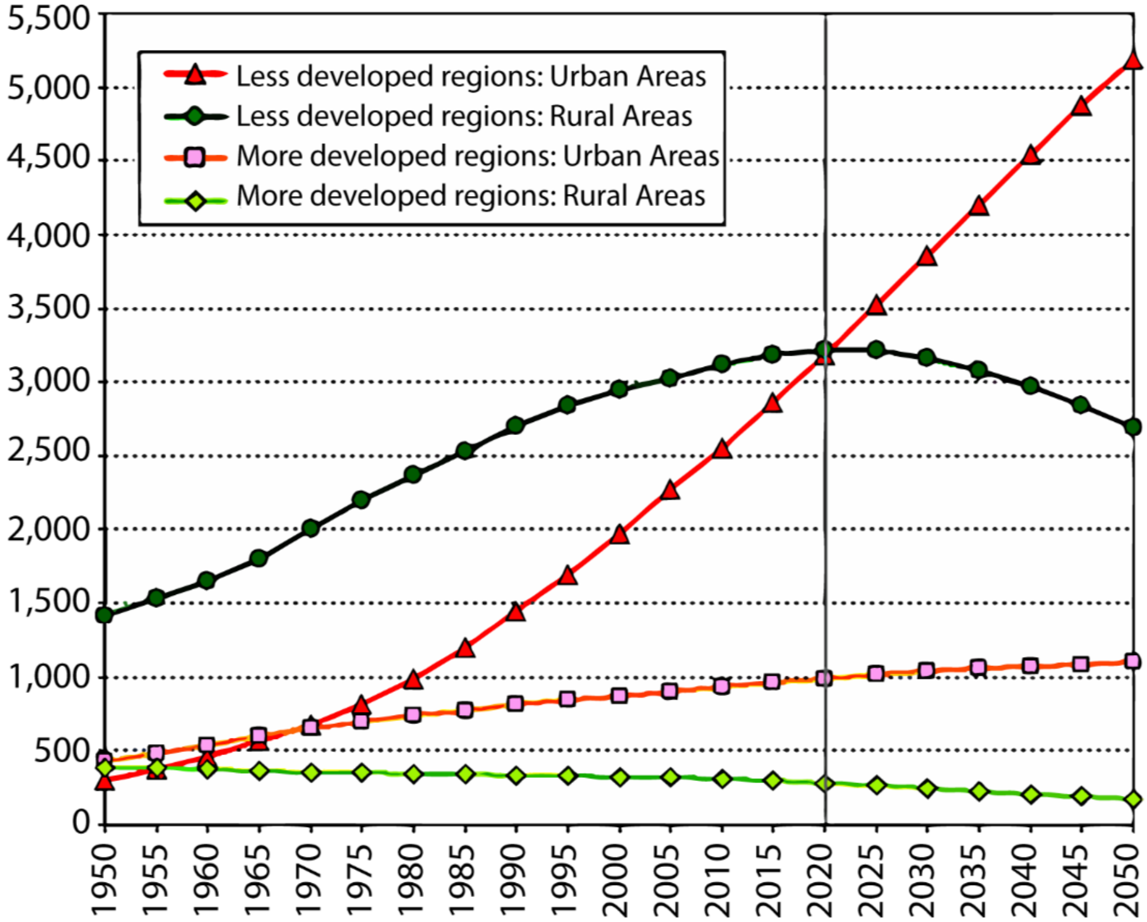

Figure 2 - Urban and rural population change for the more developed and less-developed regions of the world (in millions). Beyond 2020, the population of urban areas is projected to exceed the population of rural areas in the less-developed regions (from United Nations, 2006).

Large, vibrant, growing cities represent the engines of the world's economy, generating enormous benefits by concentrating human creativity and providing infrastructure and a workforce for intensive industrial and commercial activity. However, keeping these cities sustainable with healthy living conditions has become a major global challenge. Central to this undertaking is the provision of safe and sustainable water supplies for drinking and sanitation. Water is also essential for the food supply and industry.

The growth of megacities (defined as having populations above ten million) is a particular concern. In 1950, only New York, USA, Tokyo, Japan, and Mexico City, Mexico, qualified as megacities. By 2010, the number of megacities in the world had grown to 21, and by 2018 over 30 cities had achieved megacity status. Megacities and the complex, dynamic, peri-urban areas that evolve around them (i.e., the zones of transition from urban to rural settings) merit special attention when considering groundwater supply.

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1.2 Urban Groundwater - Out of Sight, Out of Mind

Global awareness of urban population growth and the need to address a rapidly escalating demand for water is remarkably recent. It was first explicitly embraced in the Dublin Statement on Water and Sustainable Development↗ (Tessendorff, 1992) when the scarcity and misuse of fresh water was recognized as a *"serious and growing threat to sustainable development and protection of the environment"* (p. 3). The statement noted "*human health and welfare, food security, industrial development and the ecosystems on which they depend, are all at risk*" (p. 3).

In response, the document set out a series of guiding principles and proposed an action agenda that included sustainable urban development as one of the primary beneficiaries. Since that time, the challenge of supplying rapidly growing cities with adequate supplies of water for drinking and sanitation has remained a regular topic of global discourse, most notably at World Water Forums (e.g., Marrakech, 1997; The Hague, 2000; Kyoto, 2003; Mexico City, 2006; Istanbul, 2009; Marseille, 2012; Daegu, 2015; Brasilia, 2018; Dakar, 2021) and has been recognized in United Nations World Water Development Reports (WWDR) as shown in Figure 3 (WWDR: 2003, 2006, 2009, 2012, 2014, 2015, 2016, 2017, 2018, 2019, 2020, 2021).

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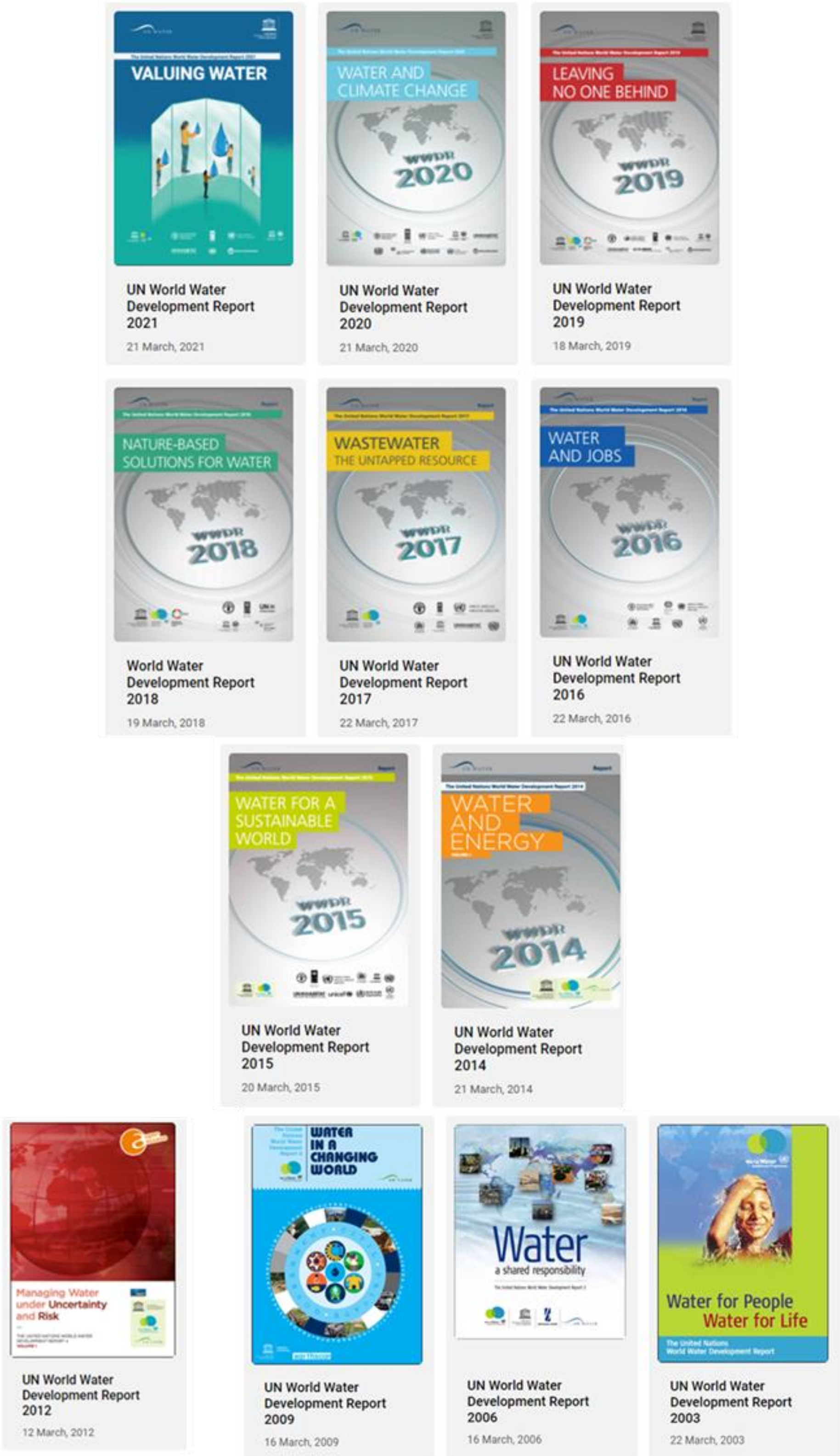

Figure 3 - The United Nations World Water Development Report (WWDR) was originally published triennially, with the first four editions launched in conjunction with World Water Forums in 2003, 2006, 2009, and 2012. In 2012, the decision was taken to revise the scope of the report and better meet the needs of readers with an annual, more concise publication. All the reports are accessible on the United Nations Water website↗. (photography by United Nations Water).

Between 2002 and 2007, urban water issues were also examined in some detail during the sixth phase of UNESCO's (United Nations Educational, Scientific and Cultural Organization) International Hydrological Programme (IHP-VI Focal Area 3.5 "Urban Areas and Rural Settlements"). Nine urban water projects implemented under UNESCO's IHP-VI↗ culminated in an International Symposium in Paris on "New Directions in Urban Water Management" during September 2007. This meeting concluded with the adoption of *The Paris-2007 Statement on New Directions in Urban Water Management* that contained a summary of its main deliberations and built strongly on the findings of previous World Water Forums and key international conferences, for example:

- Beijing Declaration and Platform for Action (United Nations Fourth World Conference on Women, 1995);
- Paris Statement of 1997 (Symposium on Water, City and Urban Planning, 1997);
- United Nations Millennium Declaration (2000);
- Marseille Statement (Maksimovic & Tejada-Guilbert, 2001);
- Johannesburg Plan of Implementation (World Summit on Sustainable Development, 2002); and
- United Nations CSD-13 policy recommendations on practical measures and options to expedite implementation of commitments in water, sanitation, and human settlements (13th session of the United Nations Commission on Sustainable Development, 2005).

The Paris-2007 Statement re-emphasized the stress placed on water resources by unprecedented rates of population growth and urbanization and identified a series of key concepts that set the stage for future action.

Unfortunately, despite all this activity, most urban water studies and in-depth analyses have examined urban water management in a broad sense, making little or no effort to articulate and incorporate critical behavioral differences between groundwater and surface water. Groundwater and surface water derive from the same atmospheric source (precipitation) and are indistinguishable when they emerge from a pipe in the home, factory, or farmer's field. However, that is where the similarities end.

Although they share the same water cycle, groundwater and surface water behave on very different spatial and time scales (Table 1) and require distinct management approaches (Theesfeld, 2010) that are rarely fully appreciated. In far too many cases, the role of groundwater in the urban water cycle (Howard & Gelo, 2002; Howard, 2004) is ignored—an *out of sight, out of mind* mentality (Foster, 1996) that has led to serious neglect of urban aquifers in many parts of the world.

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Table 1 - Key differences between surface water and groundwater (modified after Puri & Naser, 2002).

Surface Water (Rivers)	**Groundwater (Aquifers)**
Long, linear features.	Bulk 3-dimensional systems.
Use of resource either limited to close to the river channels or requires transport via pipeline.	Aquifers naturally bring the resource to within one well's length of users.
Replenishment always from upstream sources.	Replenishment may take place from any or all directions.
Rapid and time-limited gain from replenishment. Limited storage and prone to drought.	Slow response to replenishment. Very large natural storage, allowing net gain to be drawn upon over long time period. Resilient to drought.
Little opportunity to manipulate storage within the natural confines of the river.	Significant opportunity to manipulate storage in aquifer body.
Abstraction has an immediate downstream impact.	Abstraction has an impact in all directions that slowly manifests over years and decades.
Little impact on upstream riparian zones.	May have an equal impact on both upstream and downstream riparian zones.
Pollution rapidly transported downstream (on the order of meters per second).	Very slow movement of pollution (on the order of meters per annum).
Pollutant transport invariably downstream, while upstream source may be unaffected.	Pollutant transport controlled by local hydraulics. An operating well may induce upstream movement of pollution toward itself.
Excellent candidates for full and effective remediation following pollution event.	Once polluted, very difficult and expensive to remediate. Full remediation rarely achieved.

Urban groundwater problems typically begin when rapid urban growth impacts the availability of fresh groundwater through a combination of increased demand (Shahin, 1990) and degradation of groundwater quality resulting from release of urban-sourced contaminants. The overall effect is a significant increase in water supply cost that, without appropriate intervention, can negatively affect human health, undermine economic stability, and lead to socio-economic and environmental decline. In some parts of the world, problems associated with ill-managed urban water supplies have become a major threat to social and political stability. The provision of sustainable water supplies for the world's rapidly growing cities has emerged as one of the greatest challenges facing humankind (Figure 4).

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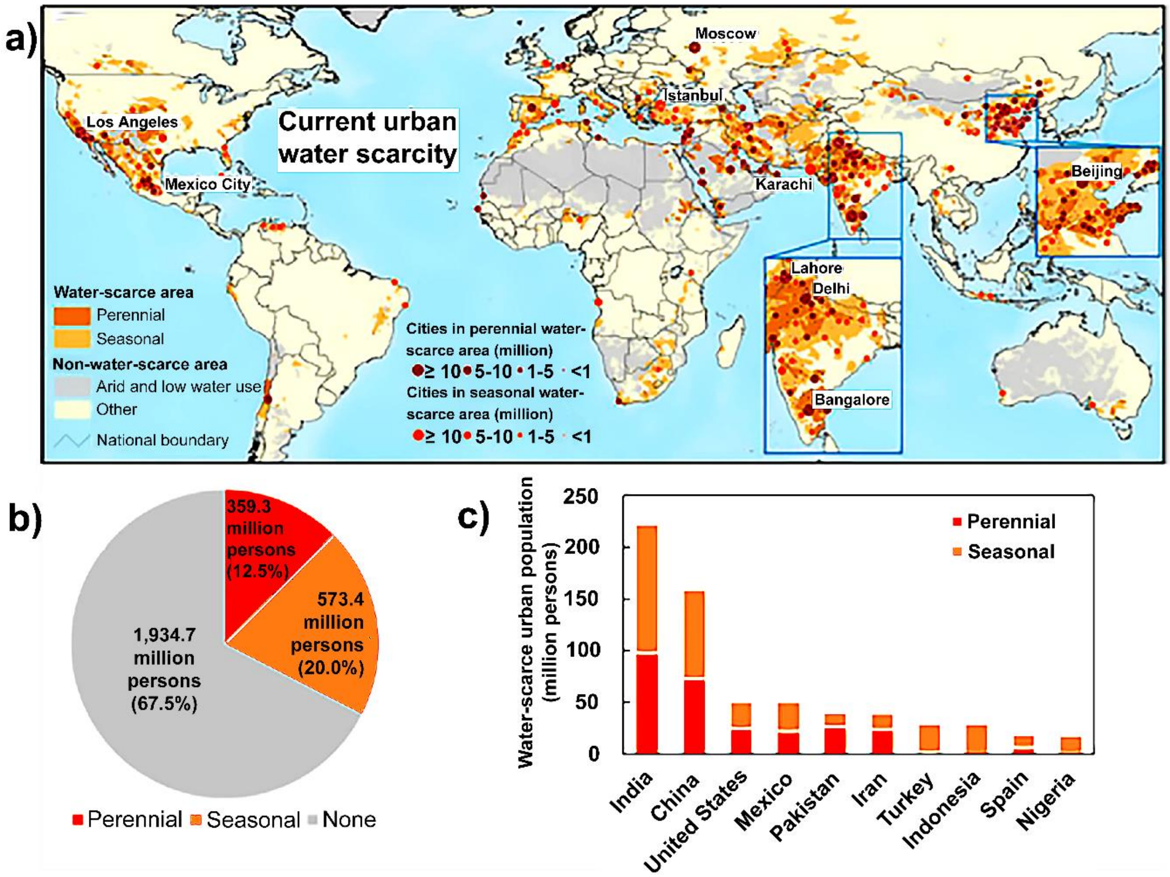

Figure 4 - Urban water scarcity: a) large cities in water-scarce areas (cities with population > 10 million in 2016 are labeled); b) water-scarce urban populations globally; c) water-scarce urban populations nationally (ten most seriously affected countries listed) (modified from He et al., 2021).

1.3 Book Outline

In this book on urban groundwater, I examine a wide variety of issues beginning with Section 2, *The Evolving Role of Groundwater in Urban Areas and the History of the Science*, with a historical perspective exploring the vital role groundwater has played in the growth and development of urban centers. Subsequently, I describe how scientific research has responded to a range of urban groundwater problems with respect to both quality and quantity. I consider impacts on quantity and quality in detail in Section 3, *Impacts of Urbanization on the Urban Water Balance - Quantities and Flows*, and Section 4, *Impacts on Water Quality*, respectively.

Some of the globe's most pressing challenges including megacities and peri-urban areas are visited in Section 5, *Major Global Challenges*. In Section 6, *Solutions to the Urban Sustainability Challenge*, I provide some relief from the apparent "gloom and doom" by exploring potential solutions. In practice, none of the possible solutions are likely to be effective without good groundwater governance, which I discuss in Section 7, *Urban Groundwater Governance*. To wrap up, I briefly describe the key takeaway messages in Section 8, *Key Takeaways and Priority Data Needs*.

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2 The Evolving Role of Groundwater in Urban Areas and the History of the Science

2.1 Evolving Role of Groundwater in Urban Areas

Groundwater represents, by far, the world's largest and most ubiquitous source of fresh, accessible water (Aeschbach-Hertig & Gleeson, 2012; Shiklomanov, 2000). It should therefore come as no surprise that many of the world's most populated cities can attribute their early origins to the good quality groundwater obtained from shallow private wells. Where aquifers are available, groundwater tends to be preferred over water from lakes, reservoirs, and rivers, because it

- is normally well-protected from surface sources of contamination;
- is rarely influenced to any great extent by drought and climate change; and
- can be brought online incrementally, just one borehole at a time, to meet the city's increasing private, municipal, and industrial demands with minimal upfront expenditure.

Problems arise when the available groundwater resource becomes stressed by a combination of increased demand from a growing urban population and the size of the resource becoming depleted due to urban pollution.

Researchers in the United Kingdom have observed that urban areas tend to evolve through a series of distinct phases as they slowly develop and mature, as shown in Figure 5 and Figure 6 (Morris et al., 1997; Foster et al., 1998; Morris et al., 2003). Associated with these phases are major developments in infrastructure, most of which are notably related to water supply and sanitation.

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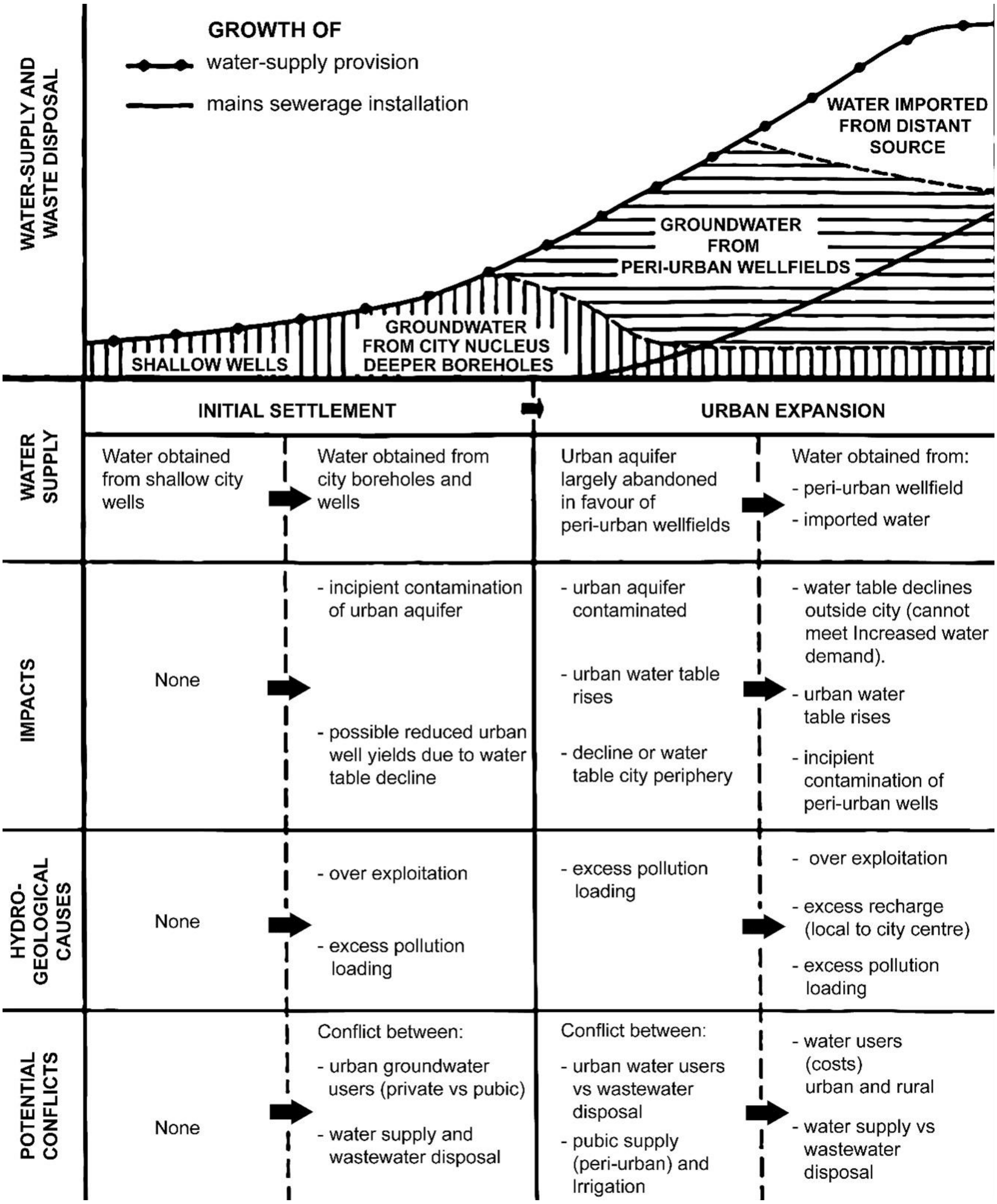

Figure 5 - The role of groundwater in the evolution of a city (modified from Morris et al., 1997).

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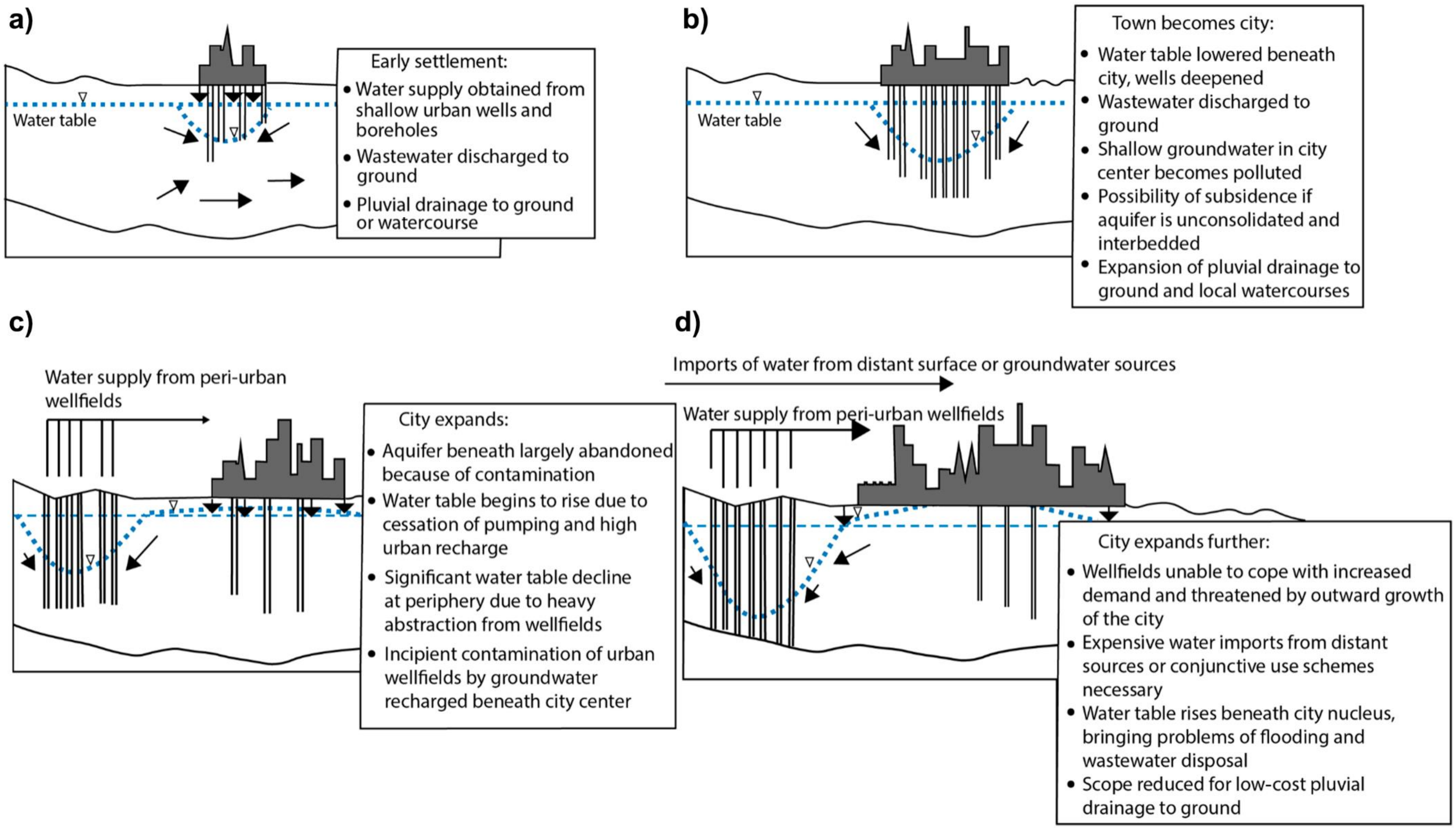

Figure 6 - The role of groundwater during the evolution of a city (from Barrett, 2004, after Morris et al., 1997, and Foster et al., 1998).

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During the first phase of development, the village or small settlement gradually grows into a market town (Barrett & Howard, 2002). Water is normally obtained from springs and shallow unplanned private wells that are centrally located; wastewater is discharged locally to the subsurface. As the town grows into a city, demand for water increases and wells are deepened due to a gradual decline of the shallow water table and localized, near-surface aquifer pollution.

During the next phase, industry takes a foothold and population growth accelerates. Meeting increased demand becomes a challenge—a problem exacerbated by extensive degradation of shallow groundwater quality. In the short term, the growing problems can sometimes be eased by further deepening of municipal wells. Eventually, many cities need to shift to new pumping wells in peri-urban areas, especially for municipal drinking water supplies. Peri-urban wells will normally provide much better quality and will also reduce pumping stress on the urban aquifer. Cities prone to land subsidence will benefit considerably as the introduction of peri-urban wells will arrest the problem and, in some cases, initiate rebound.

The installation of mains sewerage, underground wastewater disposal pipelines designed to service the entire urban community, frequently lags behind the provision of water supplies (Foster, 1990; Morris et al., 1997) as shown in Figure 5. This means the adoption of peri-urban wells will cause the city to become a major net importer of water, the excess water inevitably adding to aquifer recharge. Lerner (1990a) has suggested that for cities where sewage is not exported via canals or mains sewerage, as much as 90 percent of the total water imports may eventually recharge the local groundwater system.

Many cities, notably in the developed world, eventually go into industrial decline and enter a post-industrial phase. Even for cities with mains sewerage, the combination of declining industrial demand for groundwater and the additional aquifer recharge generated by water imports leaking from pressurized pipes causes water levels to rise throughout central parts of the city, risking flooding of underground structures including transportation tunnels. Locally, springs may re-emerge, discharging water that is severely contaminated. In some cases, pumping must be reinstated to resolve the problem, but since shallow water quality is very poor, the well discharge must be directed to wastewater facilities. Meanwhile, water levels in peri-urban and rural areas become depressed due to ongoing pumping (Figure 6).

Clearly, the sustainable management of groundwater in growing and temporally complex urban areas is a formidable challenge (Howard, 2007). Urban development impacts the quality and quantity of the groundwater resource. By the same token, the quality and quantity of the available groundwater exerts a major influence on the rate and nature by which urban growth can occur.

During the past 30 years, many of the world's cities have grown at an unprecedented rate, and much can be learned from the experience gained. As many as 1.5

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billion city dwellers across the globe rely on groundwater for their basic water needs (Foster et al., 2010b). Barrett and Howard (2002) conclude that effective management of urban groundwater requires clear, highly integrated, long-term planning and a thorough understanding of the entire urban groundwater system. This includes a thorough knowledge of the available aquifers, details of the water balance, and a full appreciation of groundwater quality. Urban groundwater management needs to be proactive, adequately funded, and based on sound science. In fact, we are fast learning (Howard & Gerber, 2018) that urban groundwater needs to be managed even where it is not used for supply.

2.2 History of the Science of Urban Groundwater

Global interest in the relationship between urban development and water dates to the mid-1900s when the post-World War II reconstruction boom in many European cities generated serious hydrological problems. For example, the construction of houses, commercial buildings, parking lots, and paved roads and streets rapidly increased the impervious cover in many watersheds, reducing direct infiltration, and stimulating the hydraulic efficiency of flow through artificial channels, gutters, and stormwater collection systems (Chow et al., 1988). The overall effect was to dramatically increase the volume and velocity of surface-water runoff, leading to unexpected increases in the frequency and intensity of urban flood events. Within a few short years, the discipline of urban hydrology (surface water science) became firmly established (e.g., Lazaro, 1979; Hall, 1984) and attracted well-funded researchers from a wide range of disciplines.

Today, the key hydrological processes are well understood, and methods for calculating and predicting surface water flows in urban areas are highly advanced. These methods include the sophisticated United States Environment Protection Agency (USEPA) Storm Water Management Model (SWMM), which had its early roots in the 1970s (Huber et al., 1975; Heaney et al., 1975; Huber & Dickinson, 1988) and is now used across the globe for planning, analysis, and design related to stormwater runoff, mains sewerage, and similar drainage systems. Notwithstanding the special challenges posed by climate change, predictive modeling skills and developments in the design and construction of engineered structures such as barriers, floodways, and storm detention ponds are now able to provide a high degree of flood protection to most major cities in the developed world.

In comparison to urban surface water problems, issues associated with groundwater have taken longer to emerge, largely due to the buffering effect of high storage aquifers and the relatively slow movement of water in the subsurface. As a consequence, the scientific study of urban groundwater is comparatively young. Urban groundwater first appeared on the world stage at Urban Water ’88, a UNESCO symposium focused on hydrological processes and water management in urban areas. In 1992, urban groundwater was recognized once again at the United Nations Conference on Environment and Development in Rio de Janeiro, Brazil, where Agenda 21 responded to a growing

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concern about rapid urban population growth; this agenda item specified the need to protect the quality and supply of freshwater resources through an integrated approach to its development, management, and use in a sustainable way. These were timely events, as urban groundwater problems were becoming prominent in the scientific literature throughout the 1980s and early 1990s. Some examples are listed here.

- Baxter (1982) documented the impacts of sewage effluent disposal on groundwater in the United Kingdom.
- Cavallaro and others (1986) and Rivett and others (1989, 1990) revealed the presence of organic contaminants in aquifers underlying industrialized urban areas in Europe.
- Flipse and others (1984) and Morton and others (1988) demonstrated the detrimental impacts of lawn fertilizer on urban groundwater quality.
- Eisen and Anderson (1979), Locat and Gélinas (1989), and Pilon and Howard (1987) raised awareness of the impacts of road de-icing chemicals on groundwater.
- Lerner (1986, 1990a, 1990b, 1990c, 1990d) and Foster (1990) drew attention to the enhancement of urban recharge due to leakage from septic systems, sewage mains, and pressurized water distribution pipes.

Research into urban groundwater issues accelerated throughout the 1990s. The International Association of Hydrologists (IAH) Commission for Groundwater in Urban Areas was formed in Oslo, Norway, in 1993 (it became the IAH Urban Groundwater Network↗ in 2011). In 1997, the IAH dedicated its XXVII Congress to the topic of "Groundwater in the Urban Environment." The proceedings of this meeting were published in two volumes (Chilton et al., 1997; Chilton, 1999), and a meeting held at this congress sparked the development of a book, *Urban Groundwater Pollution*, which was refined during discussions at the IAH Congress in 2000 and published as an edited work by Lerner (2004).

The turn of the century proved to be a very busy time for urban groundwater scientists. In May 2001, a NATO (North Atlantic Treaty Organization) Advanced Research Workshop on "Current Problems of Hydrogeology in Urban Areas, Urban Agglomerates and Industrial Centres" was held in Azerbaijan (Howard & Israfilov, 2002). Three years later, a NATO Advanced Study Institute was held to further promote the urban groundwater research agenda (Tellam et al., 2006). Urban groundwater also featured strongly at the 3rd World Water Forum held in Japan in 2003 (Howard, 2004). This was immediately followed by a special session on urban groundwater organized by the IAH Commission on Urban Groundwater at the 32nd International Geological Congress (IGC) in August 2004 (Howard, 2007). Meanwhile, in 2003, the Center for Sustainable Urban Regeneration (cSUR) was established at the University of Tokyo, Japan, for the purpose of creating new integrated knowledge for sustainable urban regeneration. Work at this center led to a book entitled *Groundwater Management in Asian Cities*↗ edited by Takizawa (2008).

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Toward the end of this busy publication period (Figure 7), the importance and potential role of numerical modeling of urban water systems emerged (e.g., Wolf et al., 2006; Pokrajac & Howard, 2010). This interest in modeling has continued, most notably in Europe, where several centers for urban groundwater research have developed with funding support from the European Union (EU). One of the most important activities funded by the EU has been the TUD-COST Action TU1206 "SUB-URBAN - A European network to improve understanding and use of the ground beneath our cities"↗. This venture has been a catalyst for innovative urban groundwater research as witnessed by the presentations given at various workshops including Bucharest, Romania (May 13–15, 2015) and Trondheim, Norway (February 3–6, 2016).

Figure 7 - Between 1997 and 2010, a range of books appeared dealing with various urban groundwater issues (photography by Ken Howard).

2.3 Scientific Progress in Urban Groundwater - Highlights

To a large extent, urban groundwater research during the past 40 years or so has tended to focus more on problem resolution (water supplies, spills, contaminant plumes, rapidly rising/falling water tables) than on urgently required proactive measures such as wellhead protection, database development, resource monitoring and the advancement of strategies for sustainable aquifer management (Hiscock et al., 2002; Tellam et al., 2006). Nevertheless, considerable scientific progress has been made on a wide range of urban groundwater issues, and a wealth of knowledge has been generated. Much of this progress is documented in some detail throughout the remainder of this book. As a preview,

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however, highlights of the more important scientific advances are provided in this section of the book.

2.3.1 Urban Karst

Small cracks in the impermeable pavement and permeable zones associated with underground pipe networks are sometimes referred to as urban karst or urban epi-karst (Sharp et al., 2001). These cracks strongly influence urban recharge, shallow urban groundwater flow, and contaminant migration (e.g., Perera et al., 2013).

2.3.2 Made Ground/Urban Fill

Materials used to level the land surface in urban areas prior to the construction of buildings and roads can be far more important than the local geology with respect to defining the shallow aquifers, recharge conditions, and the resulting nature of groundwater flow (Howard & Tellam, 2011).

2.3.3 Deep Building Foundations

Deep foundations and similarly engineered structures can profoundly affect groundwater flow directions by creating impermeable barriers to groundwater flow that locally influence aquifer water levels.

2.3.4 The Urban Water Balance

Strong evidence shows the loss of direct aquifer recharge in urban areas due to the vast expanse of impermeable surface is more than compensated by sources of aquifer recharge entirely new to the region. These sources may include

- leaking sewer pipes (Eiswirth, 2002; Wolf & Hötzl, 2007),
- leaking water mains—typically leaking 10 to 20 percent of the volume delivered to them but sometimes 30 percent or more (Hibbs & Sharp, 2012),
- septic tank discharge,
- over-irrigation of parks and gardens, and
- infiltration of stormwater runoff (usually as indirect recharge).

Cities that import water (from external or remote sources) but export sewage for treatment (natural or otherwise) outside of the city can expect losses to the shallow aquifer of about 25 percent. In contrast, cities that import water but dispose of sewage waste internally (e.g., via latrines and septic systems) contribute approximately 90 percent of the imported water to aquifer recharge. In some cities, this can represent substantial amounts of aquifer recharge (e.g., several meters of water per year).

2.3.5 Managed Aquifer Recharge (MAR)

Urban areas are net "creators" of water since limited vegetation and extensive impermeable cover result in relatively little of the incoming precipitation returning to the atmosphere as evaporation and transpiration. Recently developed technologies now allow for the high volumes of stormwater runoff to be efficiently directed into the subsurface as artificial recharge for storage and eventual beneficial use. Managed (artificial) aquifer

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recharge (MAR) is not limited to stormwater, however; treated wastewater can be "polished" to potable standards using MAR techniques (Ward & Dillon, 2011).

2.3.6 Pollutant Source Characterization

Considerable progress has been made in the use of major and minor ions, environmental isotopes, trace elements, and trace organics to fingerprint the sources of urban groundwater contamination (e.g., Carroll et al., 2013). This work is important as sources of contamination are numerous, and many cases of urban groundwater pollution involve multiple sources over different periods of time. Particularly strong headway has been made on the use of xenobiotics (e.g., synthetic chemical compounds including drug medications) and artificial sweeteners to identify and track the subsurface movement of sewer leakage (Reinstorf et al., 2007; Kaufman et al., 2011; Wolf et al., 2012).

2.3.7 Contaminant Migration

Since the 1980s, considerable progress has been made in knowledge and appreciation of contaminant plume migration (for example, Domenico & Schwartz, 1998; Fetter, 1999). Processes such as advection, directional dispersity, volatilization, degradation, and chemical retardation are now well understood for a wide range of polluting chemicals such that the subsurface movement (i.e., plume shape, velocity, and concentration) and fate of the contaminants can be predicted with a considerably greater level of certainty.

2.3.8 Disposal Methods for All Types of Domestic and Industrial Waste

Modern-day engineering techniques allow for the construction of safe, secure landfills for a wide range of urban wastes. In many cases, novel landfill designs include technologies that promote the breakdown of waste and allow for the collection of degradation products (e.g., methane) for beneficial use.

2.3.9 Approaches to Monitoring

In recent years, major advances in monitoring well technology permit water samples to be collected at multiple levels with minimal interference to the flow system and negligible cross-contamination. Moreover, improvements in the design of groundwater data loggers now allow water level, temperature, and electrical conductivity to be monitored at selected time intervals ranging from less than one second to several months, over extended periods (e.g., many years), and at depths as great as 300 m below sea level. Some instruments extend this facility to specific ions of interest (e.g., chloride).

2.3.10 Aquifer Vulnerability Mapping and Methods of Groundwater Protection

Since it was first introduced by Margat (1968), the concept of aquifer pollution vulnerability has been refined considerably (Aller et al., 1987; Vrba & Zaporozec, 1994; Witkowski et al., 2007). During the 1990s, assessment and mapping of aquifer pollution vulnerability was increasingly used as a screening tool for protecting groundwater quality

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(Foster et al., 2013). The approach has subsequently been adopted in various shapes and forms by most countries throughout the world.

Alongside the scientific advances, major progress has been made on the development of powerful aquifer modeling and evaluation tools. These tools can greatly facilitate urban water management decision making by

- performing urban water budget assessments,
- providing three-dimensional, transient simulations of aquifer systems,
- conducting aquifer susceptibility and vulnerability assessments,
- identifying Well Head Protection Areas (WHPAs)—areas around production wells most in need of protection,
- predicting groundwater travel times for contaminants in the system,
- estimating optimal pumping and water abstraction rates, and
- testing and evaluating alternative water management scenarios.

Some of the more advanced urban groundwater modeling and evaluation tools are AISUWRS (Assessing and Improving Sustainability of Urban Water Resources and Systems) by Wolf and others (2006), and UGROW (Advanced Simulation and Modeling for Urban Groundwater Management) by Pokrajac and Howard (2011). These tools include modules to simulate the various interactions that occur among groundwater, surface water, leaking sewers, and pressurized water supply systems. In turn, these modules can be linked via GIS (Geographic Information Systems) and urban databases to groundwater and contaminant simulation models to provide detailed insights into the flow of groundwater and contaminants in the subsurface.

2.4 Scientific Challenges - Urban Hydrogeology and its Unique Characteristics

As the science of urban groundwater moves forward, it is appropriate to examine the unique and special attributes of urban groundwater and the challenges they pose. A broad review of urban groundwater studies (for example, as described in Howard & Israfilov, 2002, and Tellam et al., 2006) suggests that few, if any, basic hydrogeological and hydrochemical processes are truly unique to urban systems.

Urban hydrogeology is distinguished from the rest of hydrogeology by the frequent occurrence of certain urban features that can strongly influence both the groundwater flow system and the chemistry of the groundwater. All are present to an extent in non-urban areas but usually play a minor role—and have a minor influence in any regional groundwater assessment. By the same token, features characteristically associated with rural aquifers (e.g., vegetation and the use of diffusely spread agrichemicals) can be found in some urban areas but generally have only minor significance in urban aquifers.

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Features of greater importance in urban areas due to their common occurrence are listed in Table 2 and grouped into four themes:

1. geology,
2. infiltration/recharge,
3. aquifer discharge, and
4. groundwater chemistry.

These themes are not discussed in any detail here but are provided as a background reference to Section 3, *Impacts of Urbanization on the Urban Water Balance - Quantities and Flows*, and Section 4, *Impacts on Water Quality*, of this book, where their potential effects (briefly summarized in Table 2) are described in greater detail.

Table 2 - Features of greater importance in urban areas due to their common occurrence (modified after Pokrajac & Howard, 2010).

Urban Features	Potential Effect
1. Geology	
fill/made ground	changes in hydraulic properties affecting recharge, shallow groundwater flow and transport of solutes
building foundations, cut-off walls, tunnels, subway stations	changes in groundwater flow patterns
induced landslides	changes in hydraulic properties
pumping-induced subsidence	changes in surface hydrology; promotion of pipe leakage; change in aquifer system properties
2. Infiltration/Recharge	
urban micro-climates	changed evaporation/evapotranspiration, rainfall amounts and frequency
paved cover	increased surface water runoff; reduced soil zone infiltration; reduced evapotranspiration; enhanced fingering/funneling; locally there may be increased recharge if drains associated with paved areas are connected to soakaways (infiltration pits and basins) rather than to storm sewers
compacted soils	shallow soils compacted during construction and subsequently revegetated may reduce infiltration and are significantly less permeable than natural soils which increases risk of groundwater mounding during precipitation events
interception by buildings and roads	reduced recharge unless soakaway pits are present
pipeline leakage	increased recharge, especially with pressurized pipes; risk of rising water tables, commonly described as under flooding
sewer leakage	often relatively small increases in recharge
latrines, septic systems	often relatively small increases in recharge
industrial discharges	often relatively small increases in recharge
stormwater runoff	increased indirect recharge
changing shallow water table levels	change in recharge rates
implementation of managed aquifer recharge (artificial recharge)	increased recharge

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Table 2 continued - Features of greater importance in urban areas due to their common occurrence (modified after Pokrajac & Howard, 2010).

3. Aquifer Discharge	
pumping/abstraction	variation in magnitude and time leading to complex, rapidly changing water levels and flow patterns
passive drainage	drain systems rapidly divert runoff and potential recharge to storm sewers
evapotranspiration	strongly influenced by available vegetation, rooting depth and shallow water levels in aquifer
dewatering for construction	significant, often unexpected disruption to the groundwater flow regime
discharge to surface waters	change to flow regime; possible change in surface water ability to dilute pollutants in discharging groundwaters
4. Groundwater Chemistry	
atmospheric inputs	acid precipitation; construction wash-out
surface water runoff quality	often relatively good quality, except in industrial areas, and where excessive agrichemicals and de-icing agents are used
pipeline leakages	good quality water albeit with risk of chlorination by-products; threat of chemical pipeline leakages in industrialized cities
sewer leakages	poor quality inorganically, organically, and microbiologically
releases from fill/made ground	potential long-term source of pollutants
surface water infiltration	quality dependent on surface water quality and interactions with riverbed sediments
surface water exfiltration	dilution by surface water may increase or decrease quality of surface waters; modification of surface water drainage systems will affect dilution characteristics
industrial discharges/spills	wide range of quality; short-term to long-term releases, range from plume fragments to large plumes
pumping/abstraction for supply and/or dewatering	promotes migration of plumes, often to considerable depth, and mixing with higher quality water
total loading and attenuation capacity	may exceed total attenuating capacity of aquifer on regional or local scale; non-aqueous phase liquids (NAPLs) important in latter context
new chemicals	new synthetic organics; manufactured nanoparticles; uncertain environmental behavior
old chemicals	obsolete chemicals may still be present in aquifer, i.e., potential legacy issues
major changes in pumping/ abstraction rates	change in groundwater level may change chemical processes, especially between aerobic and anaerobic conditions; ceasing pumping may cause a rise of water table and upward displacement of pollutants present in vadose zone
mixing involving multiple urban-sourced contaminants	risk of mixing, creating products of greater toxicity

2.5 Exercises Related to Section 2

Exercises Related to Section 2Exercises Related to Section 2↴.

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3 Impacts of Urbanization on the Urban Water Balance - Quantities and Flows

Urbanization has a profound effect on all components of the water balance, significantly affecting aquifer recharge, and even influencing the hydrological characteristics that affect aquifer storage and shallow groundwater flow. Cities reliant on groundwater for their water supply have a vested interest in monitoring their aquifer systems and proactively managing the resource in a manner appropriate to the hydrological changes that are taking place. Cities that rely exclusively on surface water for their supply needs are at greater risk for changing groundwater conditions because they rarely monitor underlying aquifers until problems such as flooded basements and tunnels prompt an urgent response. A lesson being learned quickly is that urban groundwater needs to be carefully monitored and managed even where it is not used for supply.

In this section, I discuss the impacts of urbanization on the urban water balance in detail, that is, quantitative aspects such as recharge, discharge, and subsurface flow. Impacts on water quality are similarly profound; these issues are discussed separately in Section 4, *Impacts on Water Quality*.

3.1 Aquifer Replenishment (Recharge)

Until the 1980s, it was widely believed that urban development caused a reduction in aquifer recharge due to the impermeable seal created across large areas of land surface by asphalt, concrete, and roof-topped structures. Numerous studies have demonstrated that urbanization will significantly deplete natural aquifer replenishment in the form of direct recharge due to the soil-zone transfer of incident precipitation. However, substantial evidence (Foster, 1990; Lerner, 1990a; Custodio, 1997) now shows such losses are readily compensated by indirect recharge resulting from overland flow, supplemented further by new sources of recharge entirely associated with urban infrastructure.

3.1.1 Direct Recharge

In natural systems, most groundwater recharge is due to a combination of direct and indirect infiltration of rainfall and snow melt (Figure 8). Mechanisms of direct and indirect recharge can be complex with the timing and volume of the recharge dependent on such factors as soil type, soil moisture, soil permeability, ground cover, ground surface slope, drainage conditions, depth to water table, free water evaporation, and the intensity and volume of precipitation. Urbanization can affect these parameters and influence recharge, either in a subtle way by altering the microclimate, or more overtly by sealing large areas of the ground surface with impermeable materials and increasing surface water runoff.

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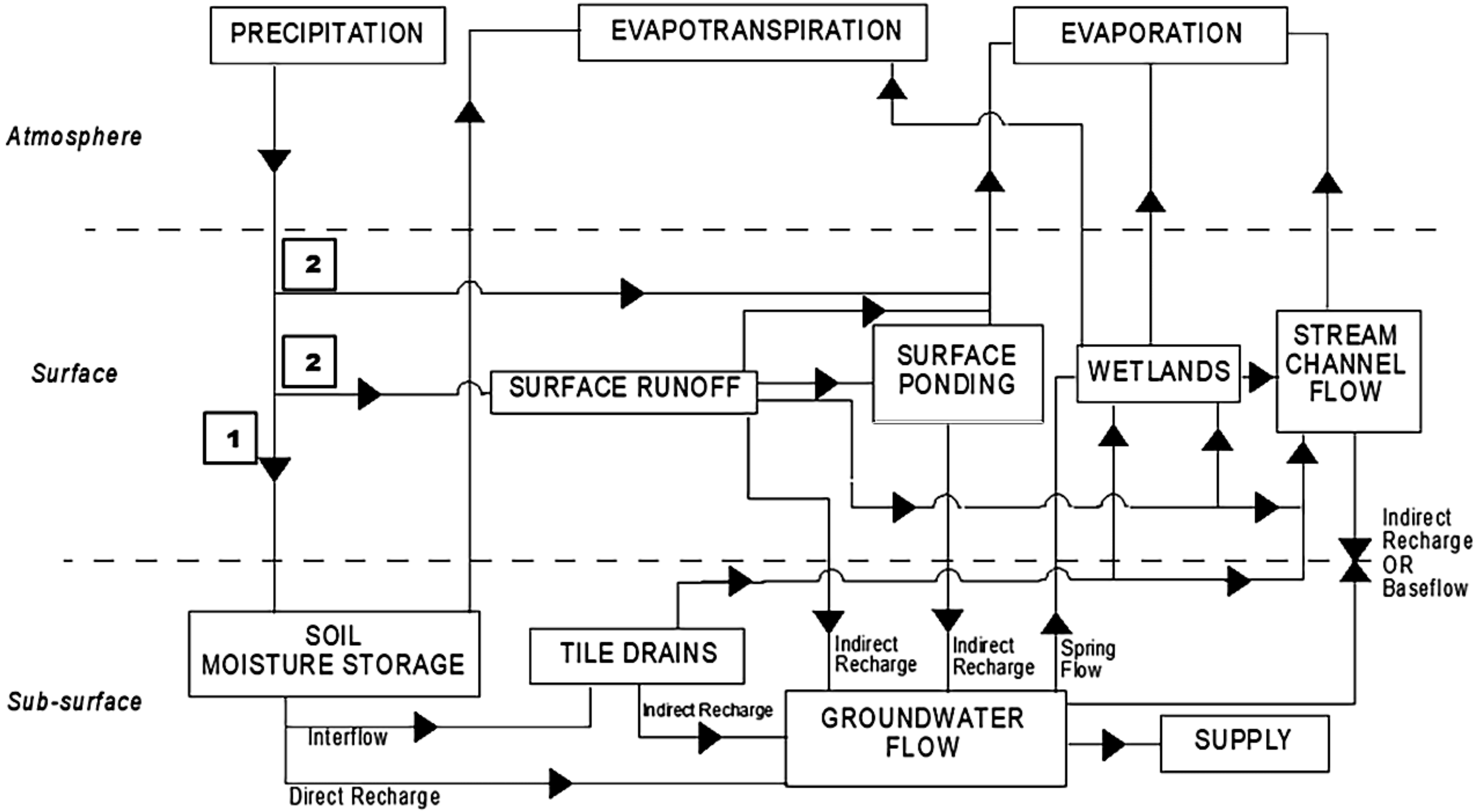

Figure 8 - Preferred water flow paths in urban areas. Pathway 1 (leading to direct recharge) is the preferred route for precipitation that falls on vegetated surfaces. Direct recharge refers to water that infiltrates into the soil soon after its arrival as incident precipitation (Howard & Lloyd, 1979) and passes directly to the underlying aquifer in approximately the location where it falls. Pathway 2 (leading to indirect recharge) is the route for precipitation that encounters impermeable surfaces. Indirect recharge normally occurs when water infiltrates following transport across the land surface as runoff. It most commonly occurs in vegetated swales, stream channels or beneath shallow temporary ponds that form following intense periods of rain (Howard, 1997).

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According to Oke (1982), the effect of urbanization on microclimate has been well documented but poorly understood. Urban heat islands are generally warmer, cloudier, and slightly wetter than surrounding areas (Landsberg, 1981). They also tend to be less windy and less humid. Overall, the effect is to increase rainfall volumes and intensities (Changnon et al., 1977; Grimmond & Oke, 1999). However, since the heat island effects on evapotranspiration are difficult to quantify (van de Ven, 1990), the net effect on direct recharge rates is virtually impossible to estimate. Increasingly, urban areas are using groundwater and heat exchange systems to control building temperatures throughout the year, further affecting flow systems and potentially affecting hydrochemical behavior.

However, the effects of microclimate on the urban water balance pale in significance when compared to the effects of introducing large areas of impermeable cover (Figure 9).

Figure 9 - Impermeable cover in an urbanized watershed includes paved a) roadways, sealed roads, b) parking lots, c) and d) sidewalks, pedestrian pathways, and roofs. In practice, it may also include sports fields and parks where underground drains have been installed to intercept any water that is in excess of basic evapotranspiration requirements (photography by Jamie Bain).

In relatively densely populated cities such as Birmingham, United Kingdom (Figure 10 and Figure 11), 70 percent or more of the land surface can be impervious. On the other hand, the degree of imperviousness may be only 20 to 30 percent in low density residential developments.

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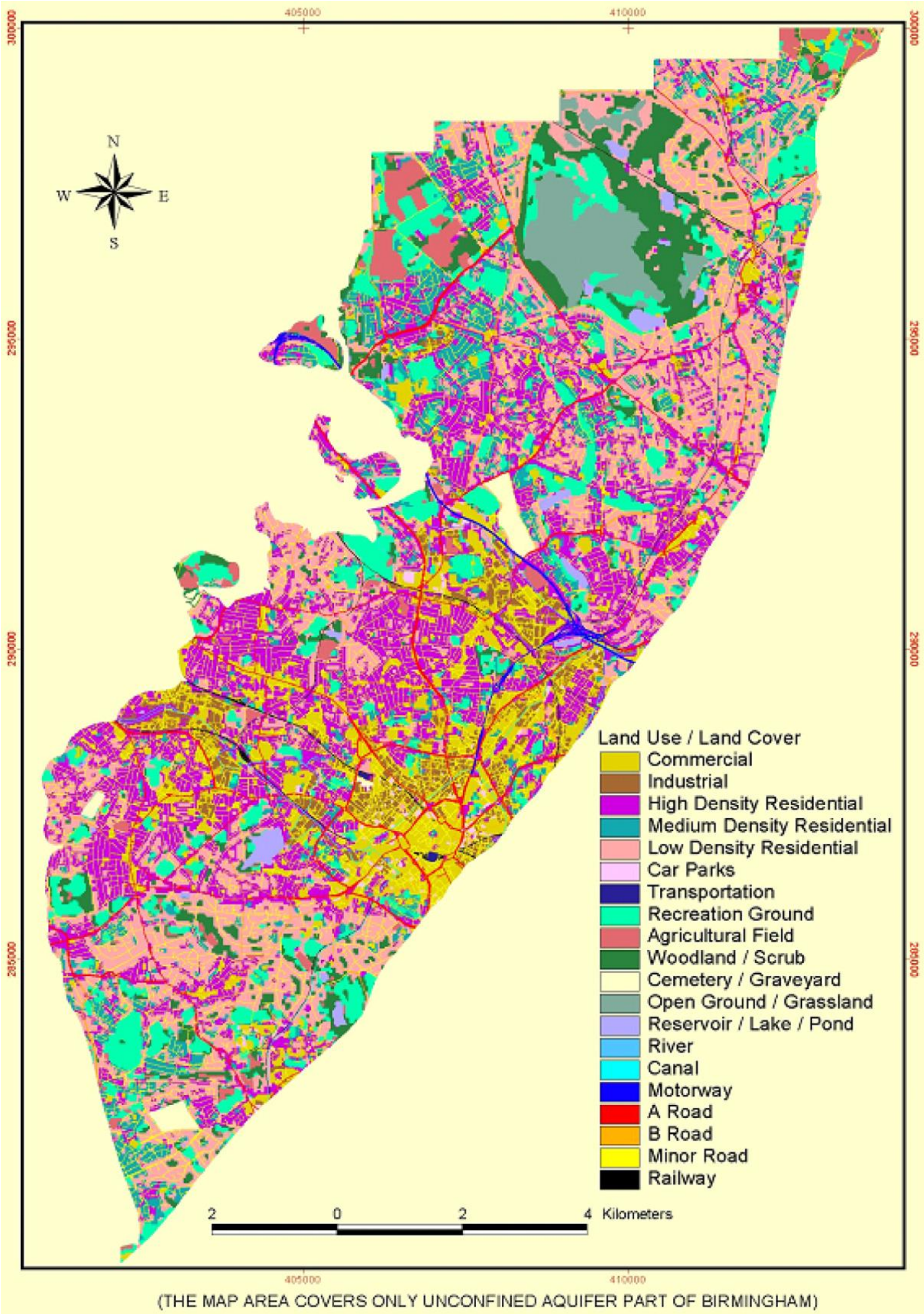

Figure 10 - Land cover on the unconfined sandstone aquifer underlying Birmingham, United Kingdom (after Thomas & Tellam, 2006b, based in part on Ordnance Survey data: © UK Crown copyright Ordnance Survey. All rights reserved).

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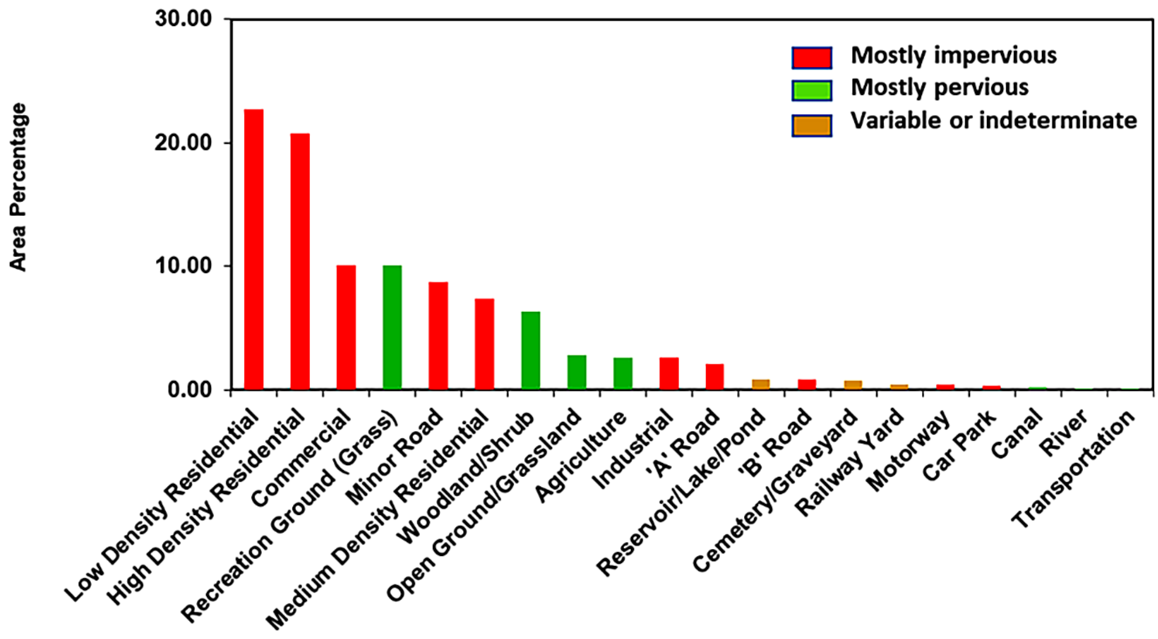

Figure 11 - The relative proportions of land cover in the unconfined portion of the Birmingham urban sandstone aquifer, United Kingdom (modified after Thomas & Tellam, 2006a).

In a typical urban environment where 50 percent of the land area is impermeable, direct recharge would be reduced by a comparable amount, that is, in a temperate climate it will reduce from about 250 mm to 125 mm per year (Figure 12). Such a large reduction can be critical where urbanization occurs in the recharge area of a major aquifer. It will be less important where the subsurface materials exhibit a low permeability and water passing through the soil zone is forced to move laterally as interflow until it eventually discharges to drains and surface streams.

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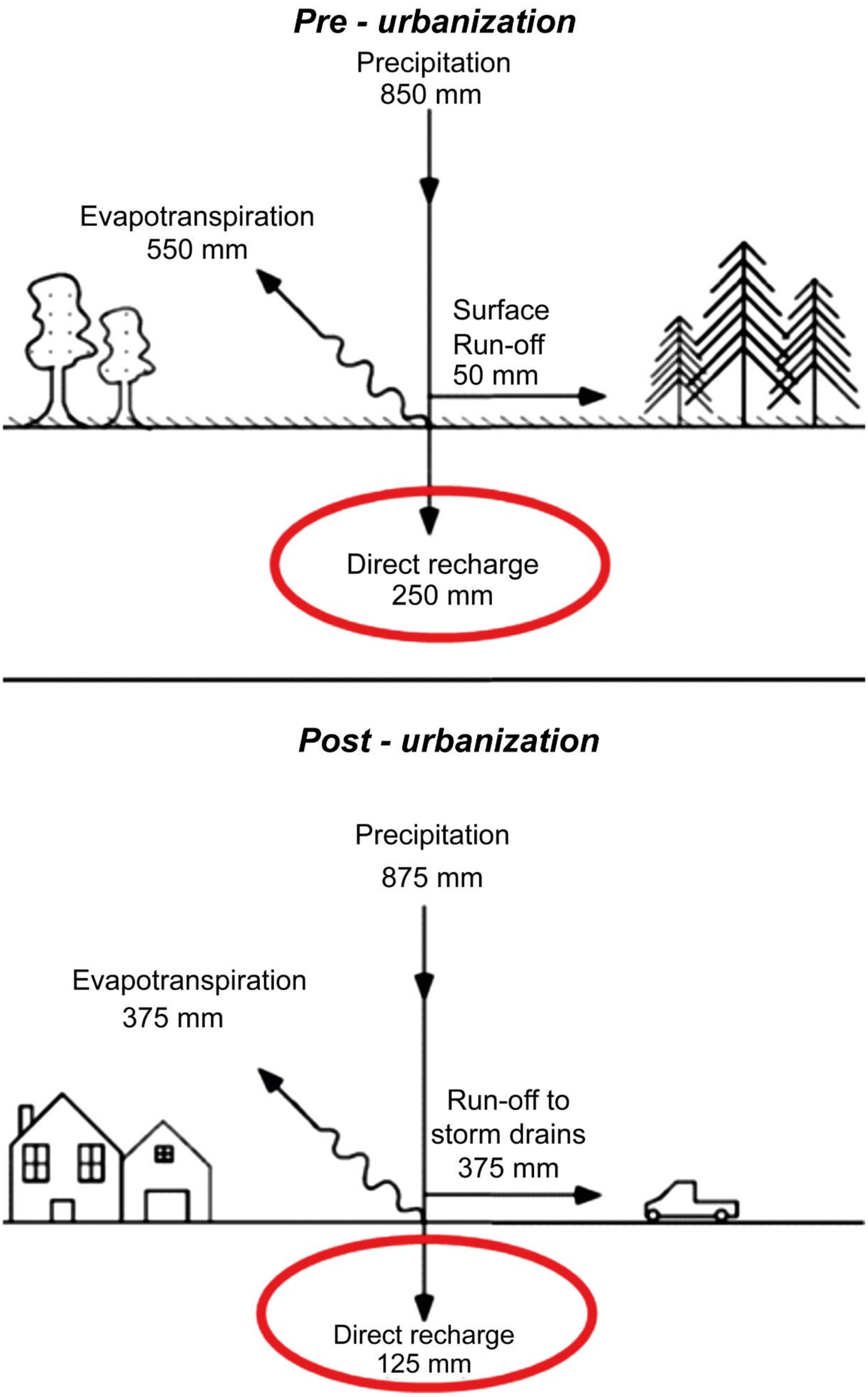

Figure 12 - Reduction in direct recharge in a typical urban area under temperate climate conditions where a small increase in precipitation due to the urban heat island effect is offset by increased runoff resulting in less recharge (after Howard, 1997).

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3.1.2 Indirect Recharge

The depletion of direct recharge in urban areas is often offset by an increase in indirect recharge. Large areas of impermeable cover will cause a significant rise in surface water runoff and a proportion of this water will infiltrate naturally depending on the surface topography, permeability of the soil and underlying rocks, local groundwater levels, and hydraulic efficiency of the stormwater collection systems. In some cases, indirect recharge can be encouraged artificially by

- downspout disconnection programs (Figure 13) that divert water destined to enter storm sewers,
- permeable pavement (Martin et al., 2001), and
- the use of soakaway pits and recharge basins that allow drainage to the subsurface with minimal evaporation.

Figure 13 - Downspout disconnection programs in Toronto, Canada, help resolve stormwater management problems but are not quantitatively assessed and, in the absence of water level monitoring, may easily overload the aquifer system and cause flooding of basements, tunnels, and electrical utilities. In Toronto, Canada, landslides along the Scarborough Bluffs are primarily facilitated by the discharge of groundwater (also Section 3.2, *Aquifer Outflow (Discharge*) (photography by Ken Howard).

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On Long Island, New York, USA (Figure 14), for example, it has been suggested that stormwater recharge basins fully compensate for losses that result from urbanization (Seaburn & Aronson, 1974), even though the spatial and temporal distribution of the recharge is radically altered (Ku et al., 1992). Similar results have been reported in Bermuda (Thomson & Foster, 1986), South Africa (Wright & Parsons, 1994), and Australia (Martin & Gerges, 1994).

Figure 14 - Recharge basin, Long Island, New York, USA. (photography by Ken Howard)

In some areas, the process of artificial recharge can be taken one stage further with the use of injection wells to recharge stormwater into underlying aquifers. As explained by Dillon and others (1994) in their review of injection well methodology, the approach is most appropriate where aquifers are confined by overlying strata of low permeability. However, while any attempt to conserve water by artificial recharge should normally be commended, it is not without risk. Particular problems include the introduction of urban contaminants as shown in Section 4 (Atkinson & Smith, 1974; Lacey & Cole, 2003; Atkinson, 2003) and the clogging of infiltration lagoons, recharge wells, and even the aquifer as a result of sedimentation and chemical and biological processes.

Estimating the volume of water that recharges aquifers as indirect recharge via permeable pavement, grass swales, and stormwater management ponds can be difficult. The problem is compounded in many cities that deliberately use the subsurface as a convenient means of solving their stormwater management issues and do so indiscriminately with little quantitative assessment of the volumes involved and no consideration of the local aquifer's ability to accept this water. In the short term, impacts

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on groundwater are rarely a problem; however, over time, excess recharge can lead to rising groundwater levels and the risk of upward flushing of any contaminants that have accumulated in the shallow unsaturated zone. As I discuss in Section 3.3, *Aquifer Response to Inequities in the Urban Water Balance Rising and Falling Water Levels*, the problem of rising groundwater levels can be particularly severe when direct and indirect recharge are further supplemented by additional sources of urban recharge.

Estimating the volume of water that recharges the aquifer indirectly via so-called sealed surfaces is also a challenge. Few paved systems are totally impervious, with infiltration readily occurring at margins and through numerous joints, cracks, and similar defects that develop over time, allowing trees to prosper in most of the world's large cities (Figure 15). Wiles & Sharp (2008, 2010) have estimated a secondary hydraulic conductivity for so-called impervious cover in older parts of Austin, Texas, at 10^{-4} to 10^{-5} cm/s which is similar to the hydraulic conductivities of very fine sand, silt, loess, and loam.

Figure 15 - Despite large expanses of so-called impermeable pavement, recharge occurs via numerous joints, cracks, and similar defects, which allows large trees to prosper in most of the world's large cities: a) London, United Kingdom; b) Shanghai, China; c) Avignon, France; and, d) Paris, France (photography by Ken Howard).

In many cases, infiltration via discrete discontinuities is encouraged by surface water ponding (e.g., Cedergren, 1989). Water entering in this way may result in rapid funneling of water through the unsaturated zone (Kung, 1990), with almost no evapotranspiration and possibly only limited opportunity for natural chemical attenuation (Thomas & Tellam, 2006a). As infiltration in this case will be almost evapotranspiration-free, recharge will show much less variation seasonally than is the case

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where vegetated surfaces are present. Rapid inflow of water via discrete entry points may also result in perching, above either low permeability lenses or coarser units with high air entry pressures (Glass et al., 1988).

3.1.3 Additional Sources of Urban Recharge

Under natural conditions, aquifer replenishment depends entirely on direct and indirect aquifer recharge. Urban development seriously depletes direct recharge; the resulting deficit is compensated to varying extents by an increase in indirect recharge. This, however, is just one small part of a much larger story.

Urban groundwater research has made considerable progress during the past four decades, and one of the most critical findings dates back to the 1980s and 1990s when studies in several cities (Lerner, 1986; 1990a; 1990b; 1990c; 1990d; 1997; Custodio, 1997; Foster, 1990) revealed major sources of aquifer recharge in urban areas—sources that had previously received little attention. These sources, listed in Table 3 and illustrated in Figure 16, can radically alter the entire water balance of an urban area and readily compensate for any loss of direct aquifer recharge. They include

- leaking pressurized water mains,
- septic tank discharge,
- leaking sewers, and
- over-irrigation of parkland, golf courses, and domestic/municipal gardens.

Table 3 - Urban factors that enhance or decrease aquifer recharge (modified after Lerner, 1990a, 1990b, 1990c, 1990d, and 2002).

Tend to enhance	Normally decrease
Urban runoff to dry wells and catch basins and subsequent infiltration	Impervious surfaces preventing direct recharge via precipitation
Impervious cover reduces evapotranspiration	Infiltration to sewers and water pipes
Artificial recharge wells and basins (managed aquifer recharge - MAR)	Infiltration to storm sewers
High density of septic tanks	Extraction (pumping) and export to other basins
Exfiltration from sewer and water pipes	
Exfiltration from storm sewers	
Imported water return flow	
Excessive irrigation	
A changing urban microclimate may enhance and/or decrease direct recharge	

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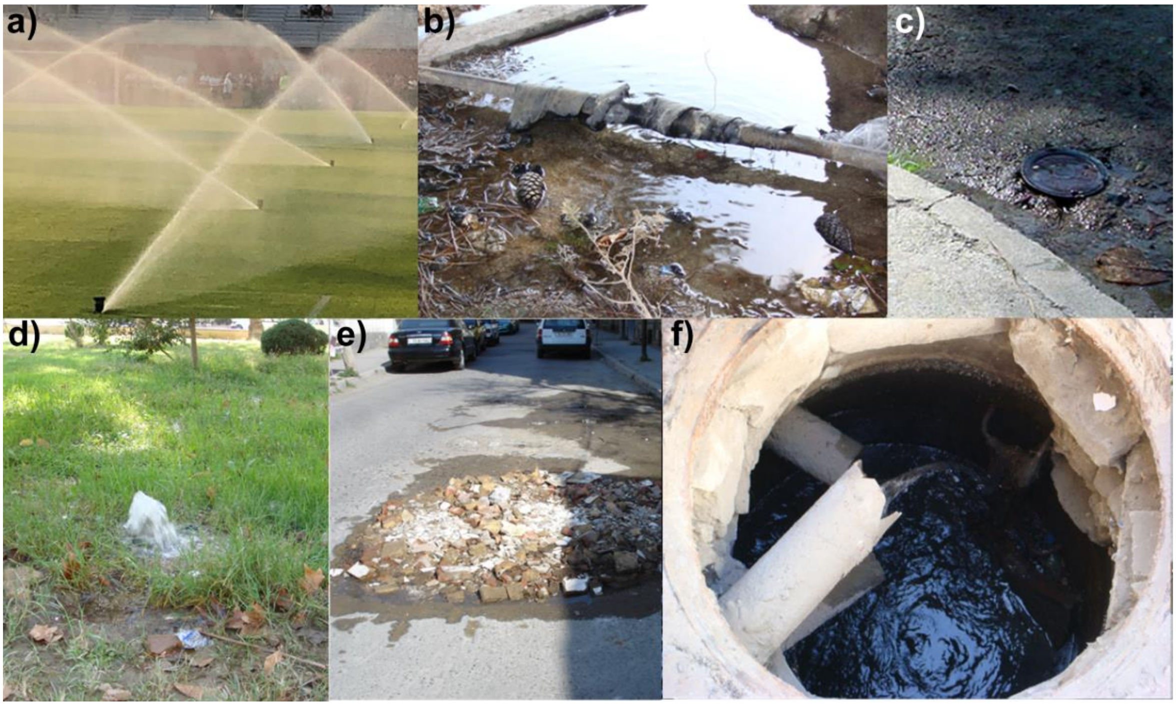

Figure 16 - Additional sources of urban recharge include: a) over-irrigation of parks, gardens and sports fields, b) c) d) e) leakage from pressurized water mains, and f) leaking sewers (photography by Ken Howard).

The contributions of these additional sources of urban recharge can be difficult to quantify as local practices and system maintenance play a major role. Globally, however, leaking pressurized water mains (Rushton et al., 1988; Lerner, 1997, 2002; Knipe et al., 1993; Lerner, 2004) probably represent the most important source of additional aquifer recharge, and this is closely followed by the release of water from septic systems in cities where mains sewerage is unavailable.

The increase in recharge due to additional urban sources is most profound in arid and semi-arid areas. In Figure 17, the extent to which urbanization enhances recharge is represented by the vertical arrows. Data for cities such as Sumgayit and Baku (Azerbaijan) and Lima (Peru), located in semi-arid and arid areas, reveal that as a result of urbanization, recharge is increased by two orders of magnitude (from around 10 mm/a to 1,000 mm/a). As indicated by the shorter vertical arrows, cities in humid areas tend to experience less extreme recharge enhancement.

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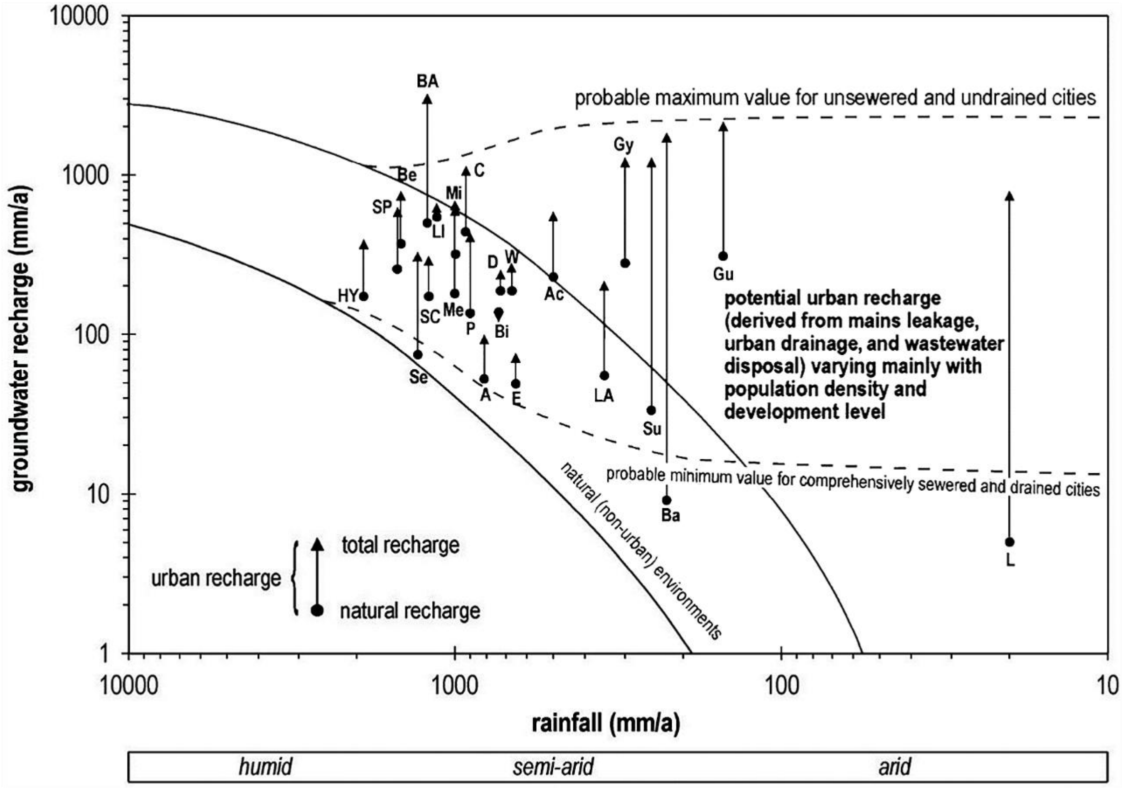

Figure 17 - Urban-enhanced groundwater recharge in 23 cities around the world. *HY*: Hat Yai, Thailand (Foster et al., 1994); *SP*: São Paulo, Brazil (Menegasse et al., 1999); *Be*: Bermuda, United Kingdom (Lerner, 1990a); *Se*: Seoul, Korea (Kim et al., 2001); *BA*: Buenos Aires, Argentina (Foster, 1990); *SC*: Santa Cruz, Bolivia (Foster et al., 1994); *LI*: Long Island (New York), USA (Ku et al., 1992); *Mi*: Milan, Italy (Giudici et al., 2001); *Me*: Mérida, México (Foster et al., 1994); *C*: Caracas, Venezuela (Seiler & Alvarado Rivas, 1999); *P*: Perth, Australia (Appelyard et al., 1999); *A*: Austin (Texas), USA (Garcia-Fresca, 2004); *Bi*: Birmingham, United Kingdom (Knipe et al., 1993); D: Dresden, Germany (Grischek et al., 1996); *W*: Wolverhampton, United Kingdom (Hooker et al., 1999); *E*: Évora, Portugal (Duque et al., 2002); *Ac*: Aguascalientes, México (Lara & Ortiz, 1999); *LA*: Los Angeles (California), USA (Geomatrix, 1997); *Ba*: Baku, Azerbaijan (Israfilov, 2002); *Su*: Sumgayit, Azerbaijan (Israfilov, 2002); *Gy*: Gyandja, Azerbaijan (Israfilov, 2002); *Gu*: Gulistan, Uzbekistan (Ikramov & Yakubov, 2002); *L*: Lima, Perú (Foster et al., 1994) (modified from Foster et al., 1994, and Garcia-Fresca, 2007).

As observed by Lerner (1986, 1990a) and Cabrera (1995), all water supply networks leak, especially when they are pressurized. Well-maintained systems may lose < 5 to 10 percent of supply; older, irregularly monitored, and poorly serviced systems can lose 20 percent or more (Table 4). In some cases, losses above 50 percent have been reported (Reed, 1980). Hueb (1986) suggests rates of leakage average 17 percent for 18 Latin America cities, a rate that, according to Foster (1990), often doubles the natural pre-development rate of aquifer replenishment. Jones (1997) estimated that average leakage rates in the United Kingdom approach 25 percent, although this has improved in recent years. In North America, leakage rates are closer to 15 percent.

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Table 4 - Estimates of leakage from pressurized water utility systems (modified after Hibbs & Sharp, 2012; Garcia-Fresca, 2004, 2006).

City	Water Main Loss (%)*
Hull, United Kingdom	5
Los Angeles, CA, USA	6–8
Hong Kong, China	8
San Antonio, TX, USA	8.5
Évora, Portugal	8.5
Milano, Italy	10
Austin, TX, USA	12
New Auckland, New Zealand	12.3
Toronto, Canada	14
Calgary, Canada	15
USA average	16
São Paulo, Brazil	16
Dresden, Germany	18
UK general rates	20–25
Baltimore, MD, USA	23
Goteborg, Sweden	26
Round Rock, TX, USA	26
Tomsk, Russia	16–30
Amman, Jordan	30
Kharkov, Ukraine	30
Sana'a, Yemen	30
St. Petersburg, Russia	≈30

*Rates may change considerably with time and spatially within an urban area. For example, in recent years leakage rates in Hong Kong have reached 25 percent, and currently stand at around 15 percent (J.J. Jiao, personal communication, January 6, 2022).

Of course, high rates of additional recharge due to supply network losses tend to mean very little in cities where the water supply is derived entirely from local groundwater. They simply represent an inefficiency of delivery that, if rectified, would not lead to a net change in the groundwater budget. Reducing leakage would simply save considerable pumping, treatment, and delivery expenses.

Instead, the effects of leakage tend to be most crucial in cities where large quantities of water are imported for supply from external sources such as peri-urban wellfields or lakes and reservoirs fed by surface water. Typically, for example, water imports for a large city may amount to an equivalent water depth of between 300 and 5,000 mm per annum, the upper estimate applying to densely populated urban centers such as Hong Kong, China. Clearly, network losses of 15 to 25 percent would make a substantial contribution to the underlying aquifer. In temperate regions, the leakage may merely offset the loss of direct recharge caused by extensive impermeable cover; in arid and semi-arid areas, however, leakage can be the primary source of groundwater recharge, sometimes orders of

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magnitude greater than natural recharge. In parts of the Middle East, recharge from imported water exceeds natural recharge to such an extent that the capacity of the aquifer to receive the water has been surpassed (Chilton, 1998).

In many large cities, the release of water from septic systems also represents an important source of additional recharge. In Bermuda, for example, septic discharge is believed to account for over 35 percent of the total annual aquifer replenishment. In Buenos Aires, Argentina, over 50 percent of the urban area is served by septic systems, the gross recharge from which is estimated to be 3,000 mm/a. This amount represents a sixfold increase over recharge in uninhabited areas (Foster, 1990). Lerner (1990a) has suggested that for cities where sewage is not exported, that is, is released to underlying aquifers by septic systems, as much as 90 of the water imported can be expected to recharge the local groundwater system.

In other cities, wastewater is exported using canals and sewage pipes. While this reduces the amount of sewage that leaks to underlying aquifers, losses can still be significant. In Mexico City, Mexico, for example, 90 percent of the sewage is discharged untreated into a sewer network that relies heavily on the use of unlined drainage canals. Two problems have arisen.

1. Land subsidence in central parts of the city has created local reversals of flow in these canals, and pumping stations are required to maintain the outward flow of wastewater.
2. Leakage through the walls of the canals returns a significant quantity of contaminated water to the aquifer. The dilemma facing the Mexican authorities is that while leakage of wastewater may degrade groundwater quality, local groundwater resources are so seriously overexploited that groundwater recharge is at a premium.

In most modern cities, underground sewer pipes are used to collect wastewater and transport it to treatment plants. Key considerations are sewer infiltration and exfiltration (Figure 18a). Most research of seepage quantity has been carried out on sewers located below the water table that tend to receive water from the aquifer as infiltration, as shown in Figure 18a and Figure 18b. In these cases, the resulting volumes of water overload the capacity of the treatment plant. Infiltration rate *q* depends on the difference between the groundwater level, *H*, and a representative hydraulic head for the sewer, *Hs.* The values for coefficients k_a, k_b, and k_c depend on the nature of the sewer (crack/joint size, frequency, and condition) and the hydraulic conductivity of the surrounding soil.

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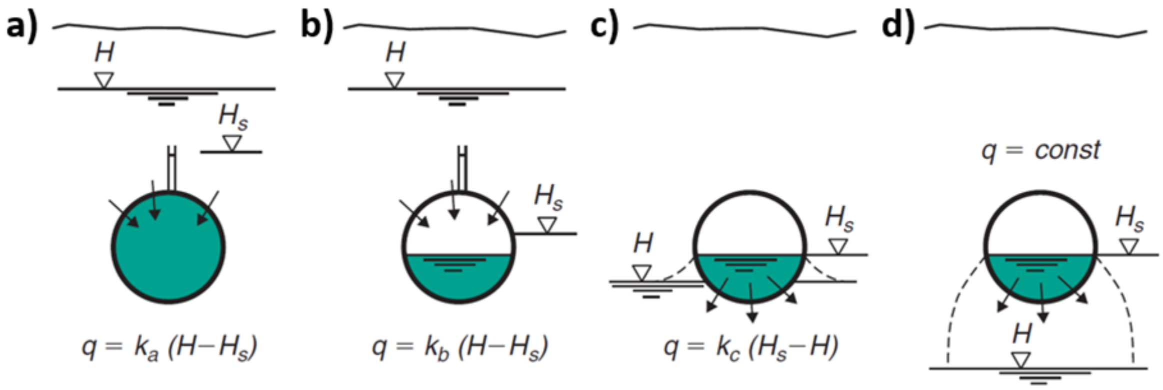

Figure 18 - Formulas for calculating sewer infiltration rates a) and b) and exfiltration rates c) and d) for typical water table conditions (a) and b) after Mills and others, 2014; c) and d) after Pokrajac & Howard, 2010).

Sewer pipes can leak due to faulty seals along joints, damage by subsidence, or deterioration with age (Eiswirth, 2002; Hornef, 1985; Seyfried, 1984). Although comparable flows might be expected to occur in the reverse direction (exfiltration, shown in Figure 18c and Figure 18d), when sewers are constructed above the water table, little is known regarding leakage rates, since most studies of sewer pipe exfiltration tend to focus on threats to water quality. In one study of the Permo-Triassic aquifer underlying Liverpool, United Kingdom, a water balance conducted by the University of Birmingham (1984) suggested that leakage from a very old, combined storm-sewer system was comparable in volume to water mains leakage. However, Lerner (1986) suggested that since sewer pipes are normally unpressurized, leakage from sewer pipes should normally be quite small. Studies in Australia have estimated that exfiltration from sewers is about 1 percent (Eiswirth, 2002), but Hibbs and Sharp (2012) suggested—based on engineering design principles—the number is closer to 6 percent.

The additional recharge that results from over-irrigation of parks, gardens, and golf courses is difficult to estimate. Typically, this category of additional recharge is most pronounced in affluent, arid, and semi-arid regions of the world where irrigation is provided, often at great expense, to meet public demand for green vegetation, especially turf grass. In the USA, Arizona (Figure 19), California, and New Mexico are well-known candidates for turf irrigation. However, cities such as Riyadh, Saudi Arabia, (Stipho, 1997; Walton, 1997) and Doha, Qatar, are equally culpable of over-irrigating amenity lands, especially when the water is applied using flooding techniques sourced by irrigation channels or hosepipes (Foster et al., 1998). In high-income districts of Lima, Peru, irrigation was found to generate 250 mm/a of additional recharge (Geake at al., 1986), a tenfold increase over natural rates of aquifer recharge.

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Figure 19 - Sedona Golf Resort in Arizona, USA. Hole number 10, par 3. Intensive irrigation (and inevitable leakage losses to the subsurface) allows the year-round establishment of lush grass in regions where normal levels of precipitation would promote a barren desert landscape (photography by Dennis Begin).

3.2 Aquifer Outflow (Discharge)

The outflow from urban aquifers can be just as wide-ranging as inflows, especially where local groundwater is used for supply. For cities that import water from external ground and surface water sources, increased recharge due to factors described in Section 3.1 leads to an increase in natural groundwater discharge—for example, as urban springs and baseflow contributions to urban streams and rivers. Provided there are no serious impairments to water quality (Section 4), increases in natural discharge can usually be safely accommodated. Exceptions occur when increased discharge of groundwater exacerbates land and slope stability issues such as illustrated in Canada by landslides along the Scarborough Bluffs in Toronto (Figure 20).

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Figure 20 - a) Regular landslides along the Scarborough Bluffs, Toronto, Canada, are primarily facilitated by the discharge of groundwater (Eyles & Howard, 1988), despite statements by the local conservation authority that landslides are mostly related to water levels in adjacent Lake Ontario. Locally, groundwater discharge is enhanced by downspout disconnection programs that redirect potential stormwater flows to the subsurface (Figure 14). b) In some locales, the resulting land instability limits use of lands that would benefit the community (photography by Ken Howard).

Where cities overlie important aquifers that can be used for supply, aquifer discharge in terms of production/abstraction can vary significantly. Typically, as discussed in Section 2.1, many cities begin their development as users of local groundwater, often from springs and shallow wells, and as they grow, so does groundwater production. There comes a time, however, when groundwater supply constraints emerge and important resource management decisions need to be made.

The most common constraints are declining groundwater levels, which increase pumping costs, diminishing water quality, and the need for expensive water treatment. In some cases, emerging problems with water quality can be overcome by deepening wells and drawing water from previously unutilized aquifers. In other cases, the problems escalate, with declining water levels leading to land subsidence and/or incursion of saline water.

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Globally, numerous examples demonstrate how rapidly groundwater production may change in an urbanized area in response to population growth and/or an unanticipated constraint. For illustrative purposes, three examples are given in Figure 21, Figure 22, and Figure 23.

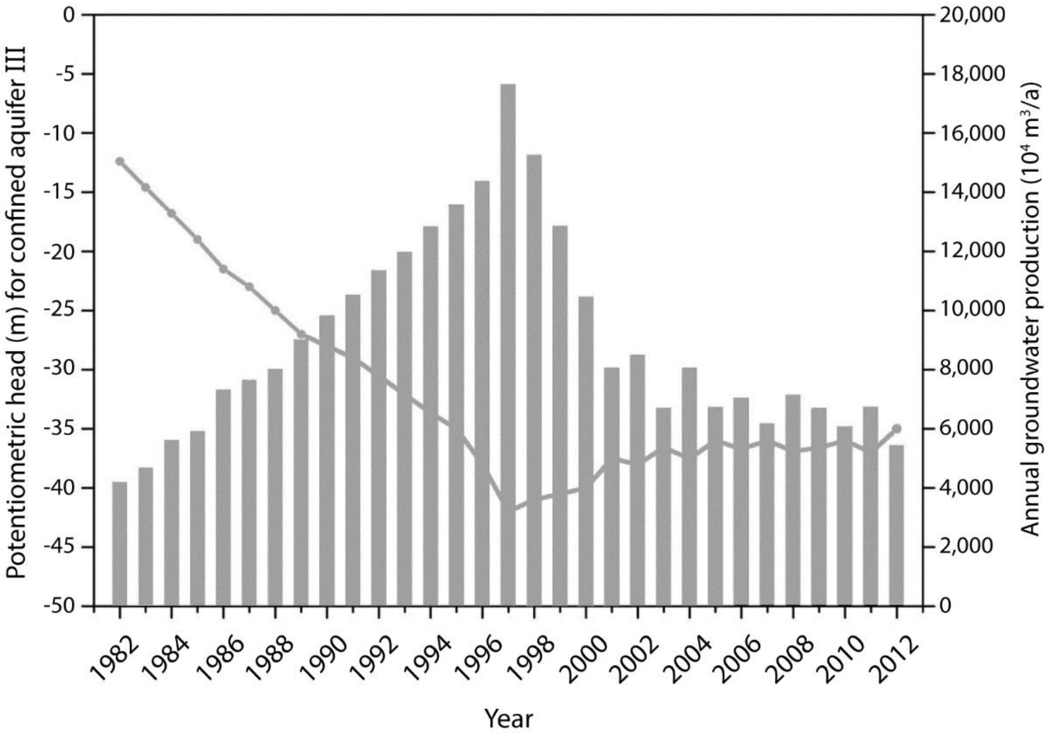

Figure 21 - Annual rate of groundwater abstraction (bars) from confined aquifer III, Nantong, China, between 1982 and 2012 in relation to the potentiometric head (continuous line) in observation well no. 62771502, screened in the same aquifer (Ma et al., 2018). The figure shows how, over a period of just 15 years, a four-fold increase in groundwater production in Nantong led to a 30 m drop in aquifer water level. In the late 1990s, concerns over land subsidence prompted a rapid cutback in production and a steady albeit slow water level recovery. Aquifer modeling is now being used to investigate options for distributing pumping regionally and across all aquifers to optimize the yield of the aquifer system (Ma et al., 2018).

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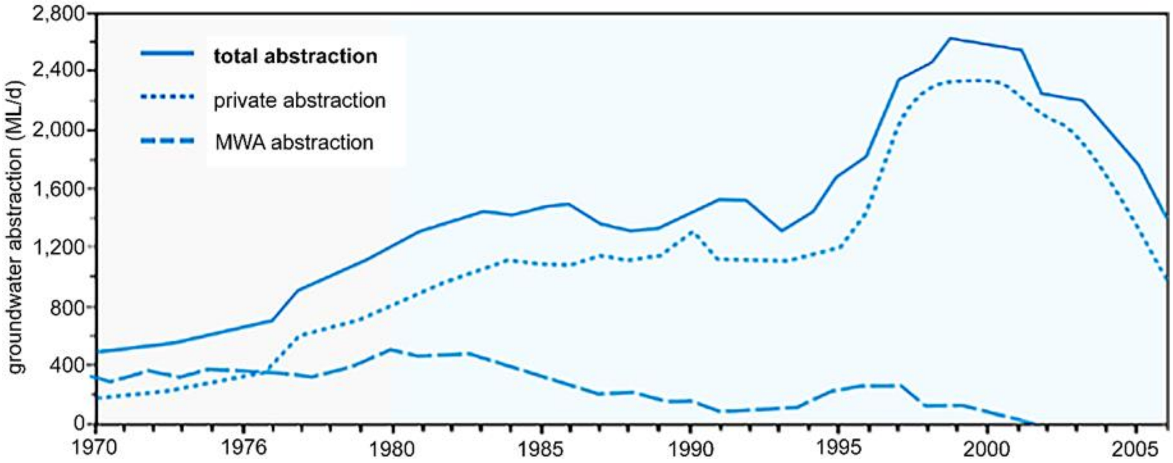

Figure 22 - Bangkok, Thailand, experienced a severe lowering of potentiometric levels in the 1980s and 1990s, and this led to both land subsidence and an increased threat of seawater intrusion (Buapeng & Foster, 2008). Initially, the problem was approached by closing wells operated by the Metropolitan Waterworks Authority (MWA) in favor of surface water sources drawn from the north, outside the city. However, the increased domestic, commercial, and industrial tariffs for mains water supply (imported from more distant sources and requiring treatment) triggered a massive increase in the drilling of private water wells, whose total production reached over 2,000 ML/d (mega liters per day) by the late 1990s. Recent years have seen more concerted action to constrain private groundwater abstraction within environmentally-tolerable limits, guided primarily by the temporal and spatial trends in the rate of land subsidence (modified from Buapeng & Foster, 2008).

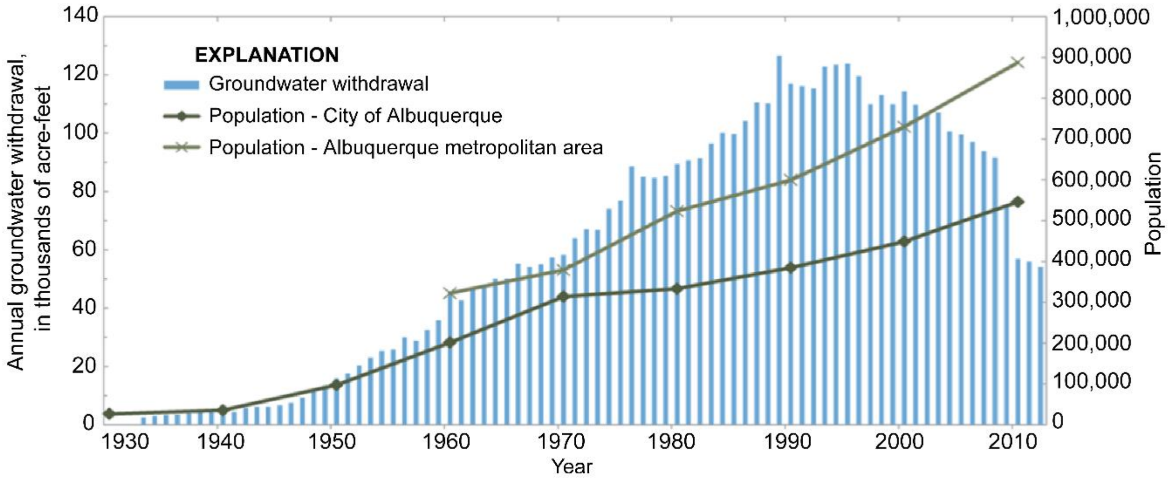

Figure 23 - In Albuquerque, New Mexico, USA, a rapidly growing population led to over-exploitation of groundwater reserves, which led to a 50 percent reduction in groundwater withdrawals from the city's aquifer. The city's strategy for water supply now involves the conjunctive use of groundwater and surface water (Turin et al., 1997).

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3.3 Aquifer Response to Inequities in the Urban Water Balance - Rising and Falling Water Levels

When there are inequities in the aquifer water balance, that is, aquifer inflow ≠ aquifer outflow, the potentiometric surface will change over time, with water moving into or out of storage until a new equilibrium is reached. In urban areas, where water balance inequities can be severe and change rapidly, variations in the potentiometric surface can be just as severe; they can also be unexpected. A new stable equilibrium is rarely reached.

3.3.1 Cities that Use Local Groundwater for Supply

For cities that use local groundwater for supply, the most common scenario is a declining potentiometric surface in response to a growing population that increases demand, that is, increases aquifer outflow. To some extent, the decline is ameliorated by increases in recharge associated with urban growth (indirect recharge and additional sources of recharge described earlier). However, for the most part, excessive use of the groundwater leads to a steady reduction in potentiometric level.

The use of groundwater at rates in excess of recharge is referred to as overdraft, over-exploitation, over-development, or groundwater mining. While such practices can generate considerable social and economic benefits, particularly in the short-term, the lowering of the regional potentiometric surface will gradually reduce well yields and significantly increase pumping costs. Well-known examples include São Paulo, Brazil, (Diniz et al., 1997) and Ljubljana, Slovenia (Mikulic, 1997). Reduced groundwater heads can also induce poor quality water to enter deeper parts of the aquifer from rivers and polluted shallow aquifer systems (e.g., Ahmed et al., 1999). More seriously, it may also lead to

- land subsidence, and
- inflow of saline water from deeper geological formations or the sea.

Land subsidence and the intrusion of seawater represent some of the earliest manifestations of excessive groundwater mining in urban areas. In Mexico City, Mexico, for example, the detrimental effects of groundwater mining have been recorded in the form of land subsidence for almost 100 years. Heavy production from deep aquifers began in the late 1920s (Sanchez-Diaz & Gutierrez-Ojeda, 1997) and, by 1959, the central parts of the city were locally subsiding at a rate of 40 cm per year (Hunt, 1990; Howard, 1992). In some locations the land surface declined by over 9 m (Poland & Davis, 1969).

A redistribution of wells has now moderated the severity of the problem throughout much of the capital; however, serious damage remains. Problems include disruption of underground water mains and sewer pipes leading to severe losses, structural damage to roads and buildings, and major alterations to surface drainage conditions.

Land subsidence due to over-exploitation of groundwater is also well documented in other large cities. Many of these cities—such as Houston, Jakarta, Shanghai, Venice, Calcutta, Taipei, Tokyo, and Bangkok—are in coastal areas where subsidence can expose

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parts of the city to invasion by the sea. In Tokyo, Japan, for example, the most heavily populated coastal city in the world, ground subsidence due to excessive use of groundwater was first observed in the 1910s. Damage to industry during World War II reduced demand for groundwater in the early post-war years and provided some relief; however, subsidence resumed at an accelerating rate in the early 1950s when industry revived and demand for groundwater increased dramatically. Some parts of the city reported subsidence of over 4 m, with the land reaching as much as 1 m below mean sea level. This is particularly serious in an area prone to the storm surges and high waves of typhoons and tsunamis. Countermeasures introduced during the 1960s included the raising of riverbanks and the construction of a sea barrier, coupled with a plan for major reductions in groundwater withdrawal. Today, the problem appears to have been resolved. Tokyo is no longer regarded as a groundwater-dependent megacity.

Problems of excessive groundwater use in coastal cities are not limited to subsidence. It can also lead to a deterioration of groundwater quality due the intrusion of seawater. Intrusion of seawater in coastal aquifers is a natural consequence of the density contrast between fresh and saline water, with the denser seawater forming a wedge that can extend for many kilometers inland (Bear, 1972; Raudikivi & Callender, 1976).

Potential problems arise when the saline water body is drawn further into the aquifer by pumping the fresh groundwater reserve. Carefully and purposefully managed, seawater intrusion can prove beneficial (Howard, 1987) by reducing the rate at which groundwater levels are lowered during periods of over-development. As a result, the long-term recovery of freshwater reserves is enhanced, pumping costs are minimized, and potential subsidence issues are lessened. However, when saline intrusion is not managed effectively, the saline water can enter pumping wells and seriously degrade water quality. Once degradation has occurred, it can take a much longer period of substantially reduced pumping for the aquifer to recover.

Intrusion of saline groundwater is common to many coastal cities. Megacity examples include Manila, Philippines, and Jakarta, Indonesia, but any coastal city that utilizes groundwater in significant quantities can be affected such as Dakar, Senegal (Faye et al., 1997). Comparable problems also occur in inland cities where excessive pumping can draw deep bodies of connate or fossil saline water toward pumping wells.

In Manila, groundwater use has lowered the potentiometric surface locally to between 70 and 80 m below sea level. In some locations, this decline has taken place at a rate of 5 to 12 m per year. Not surprisingly, saline water from Manila Bay extends inland as much as 5 km, and samples drawn from wells in coastal areas commonly exhibit chloride concentrations above 200 mg/L. Concentrations as high as 17,000 mg/L chloride have been observed. By comparison, groundwater pumping in Jakarta has not only caused substantial water level decline, but it has also led to serious land subsidence. Today, Jakarta is recognized as "the world's fastest sinking city" (available at BBC News Website↗) and this

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is one of the reasons the 7th Indonesian president Joko Widodo vowed to build a new administrative capital on the once jungle-covered island of Borneo.

Further discussion of subsidence is provided in Sections 5.2, *Seawater Intrusion* and 5.3, *Land Subsidence*.

3.3.2 Cities that Do Not Use or No Longer Use Local Groundwater for Supply

Many cities do not use or no longer use local groundwater for supply. They use local surface water such as rivers or streams or (more likely) import water from external ground and surface water sources. In either case, indirect recharge—supplemented by additional sources of aquifer recharge such as leakage from aqueducts and pressurized water mains, septic tank discharge and/or leaking sewers, and over-irrigation of parkland, golf courses, and domestic/municipal gardens—will readily offset reductions in direct recharge.

The overall effect is to significantly increase aquifer recharge and this, in turn, may translate into a significant rise in the regional water table. In turn, this can cause flooding of streets, cellars, sewers, septic systems, utility ducts, and transport tunnels; reduce the bearing capacity of soils/structures; and impact amenity space by waterlogging sports fields and killing trees (Heathcote & Crompton, 1997). The problem is particularly acute in low-storage, poorly transmissive aquifer systems where additional water is not readily accommodated and distributed.

In Baku, Azerbaijan, the pressurized water mains have been poorly maintained for decades. As a result, water imported from remote sites to the north leaks at an alarming rate, causing the local water table to rise to within meters of the ground surface. The high water table and associated elevated porewater pressures triggered a major landslide around the turn of the century (Figure 24). High water tables are also responsible for the widespread flooding of basements and quarries (Figure 25).

Figure 24 - Major urban landslide initiated in Baku, Azerbaijan (site X), was directly associated with high groundwater levels caused by a combination of heavy rain and leaking water mains (photography by Ken Howard).

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Figure 25 - In Baku, Azerbaijan: a) leakage rates from imported water exceed 50 percent; b) Well lies beneath cover shown in a). Groundwater levels have risen locally by 20 m in 30 years, c) Rising water levels were recorded in local wells and visually observed by rising water levels in urban quarries. Image shows flooding quarries and basements (photography by Ken Howard).

The effects of water table rise are particularly significant in cities that pumped large quantities of groundwater during major growth periods but subsequently abandoned the groundwater resource in favor of imported surface water supplies. In such cases, rising water levels due to leakage from services are combined with the natural long-term recovery of water level. In the United Kingdom, for example, the long-term effects of importing water and rejecting previously used groundwater reserves has been documented in areas such as Brighton, Birmingham, London, Liverpool, and Nottingham. Locally, the problems these areas experienced include the re-establishment of urban springs, water-logging of low-lying residential areas, and an upward flushing of salts and contaminants that had previously accumulated in the shallow unsaturated zone (Lerner, 1994; Barrett & Howard, 2002). Similar problems have been encountered in Brisbane, Australia (Cox & Hillier, 1994), and in Jodhpur, India (Paliwal & Baghela, 2007), as shown in Figure 26.

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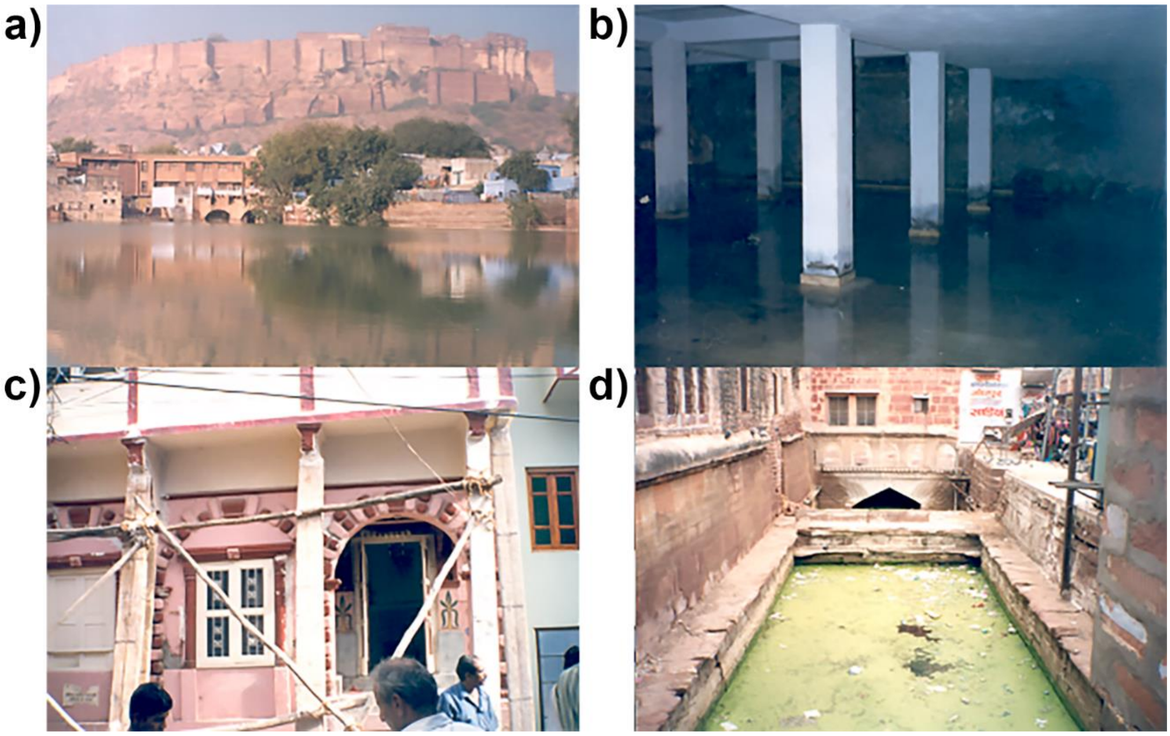

Figure 26 - In Jodhpur, India, groundwater levels were found to be rising 1 to 1.5 m per year despite persistent drought conditions. a) Gulab Sagar Lake remains filled with water even in the summer, b) basements flood, c) house foundations in the old city are damaged, and d) Gorinda Baori (an old step well) overflows (after Paliwal & Baghela, 2007). The problem was caused by construction of the Indira Gandhi Canal, which brought water into the city from the Himalayas. Leakage from the canal, poor drainage, leaking sewers, and the abandonment of local dug wells, step wells, and pond supplies all contributed to the problem (photography by B.S. Paliwal and A. Baghela).

3.4 The Critical Roles of Urban Fill, Urban Karst, and Deep Foundations

Three features that tend to be unique to urban areas and have a profound influence on hydrogeological conditions are the presence of

- fill or made ground;
- a complex network of openings associated with underground pipes, utility trenches, storm drains, poorly compacted fill, and tunnels that create a secondary permeability often referred to as urban karst or epi-karst (Figure 27); and
- deep foundations and similarly engineered structures that create barriers to groundwater flow.

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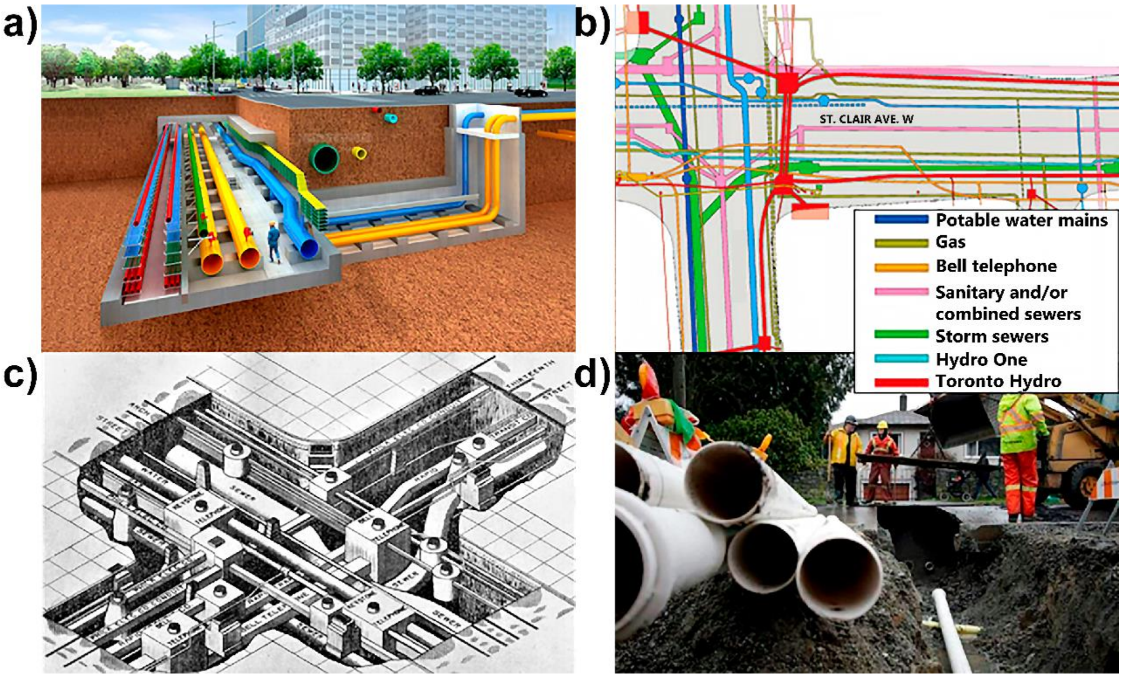

Figure 27 - Beneath urban areas, a complex network of drains, pipes, and tunnels create urban karst or epi-karst, a shallow artificial aquifer that exerts a major—yet often unpredictable—influence on aquifer recharge, groundwater flow, and contaminant transport. The term urban karst has been used for many years (Sharp et al., 2001; Krothe, 2002; Krothe et al., 2002; Sharp et al., 2003; Garcia-Fresca, 2007) to describe the secondary permeability and porosity that develops in association with subsurface construction activities. While trenches and drains lined with permeable material are usually the primary contributors, the term is often used to include any enhancement of permeability and porosity that is created by urbanization, either directly or indirectly (photographs from personal collection of Ken Howard).

Fill or made ground refers to anthropogenic material—typically building, industrial, or domestic wastes (Rosenbaum et al., 2003)—used to infill depressions and provide a level surface for construction. Material used for leveling tends to be highly heterogeneous in composition; as a result, its hydraulic properties and chemical properties often display a similar degree of heterogeneity. Some types of fill may enhance the permeability of the subsurface, while other types may reduce it, especially when compacted. While most components of fill are chemically inert, others (plaster or leachable industrial or domestic wastes, for example) can be strongly reactive.

Fill can also be used to line trenches cut into non-anthropogenic or made ground deposits (Brassington, 1990; Heathcote et al., 2003). The type of fill used in trenches is normally a prescribed mix of clean sand and gravel, and this can create zones of high permeability that can extend for hundreds of meters. When trenches are used in association with potential pollution sources (for example, sewer pipes), the risk of rapid contaminant transport can be significantly enhanced.

With rare exceptions, deep building foundations and similarly engineered structures create low permeability barriers that can strongly influence groundwater levels and significantly disturb groundwater flow paths (Jiao et al., 2004, 2008). In coastal areas, such structures can affect groundwater discharge to the sea and alter the position of the

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freshwater–seawater interface. Ding and others (2008) suggested that the disruption of flow patterns becomes a concern when deep foundations occupy over 15 percent of the aquifer.

From a hydrogeological standpoint, fill (made ground), urban karst, and deep foundations—all products of urban engineering—play a far greater role in cities than the shallow natural geology. Evaluating their roles is especially important given that water level changes due to changes in the urban water balance and/or changes in the hydraulic properties of the aquifer can have a profound influence on groundwater flow patterns. These, in turn, can substantially influence contaminant transport and the nature by which urban-sourced chemicals are released to urban streams (described in Section 4.3, *Urban Groundwater Quality Compounding Factors*). They can also affect the relationship between fresh water and sea water in coastal aquifers (described in Section 5.2, *Seawater Intrusion*).

3.5 Exercises Related to Section 3

Exercises related to Section 3 are available at this link↴.

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4 Impacts on Water Quality

All urban areas involve the importation, manufacture, storage, transport, utilization, and export of significant volumes of potentially polluting chemicals. Over time, it is inevitable that at least some proportion of these chemicals (Figure 28) will find their way to shallow urban aquifers and threaten groundwater quality as shown in Figure 29 (Howard, 1997; Squillace et al., 2002; Lerner, 1990b, 1990c, 2004; Kaufman et al., 2009).

Figure 28 - Urban areas introduce and host multiple sources of potential groundwater contamination (photography by Ken Howard).

Figure 29 - Urban contaminants seriously threaten groundwater supplies (photography by Ken Howard).

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Figure 30 shows the most common sources of groundwater contamination in urban areas, together with contaminant categories and typically encountered chemicals.

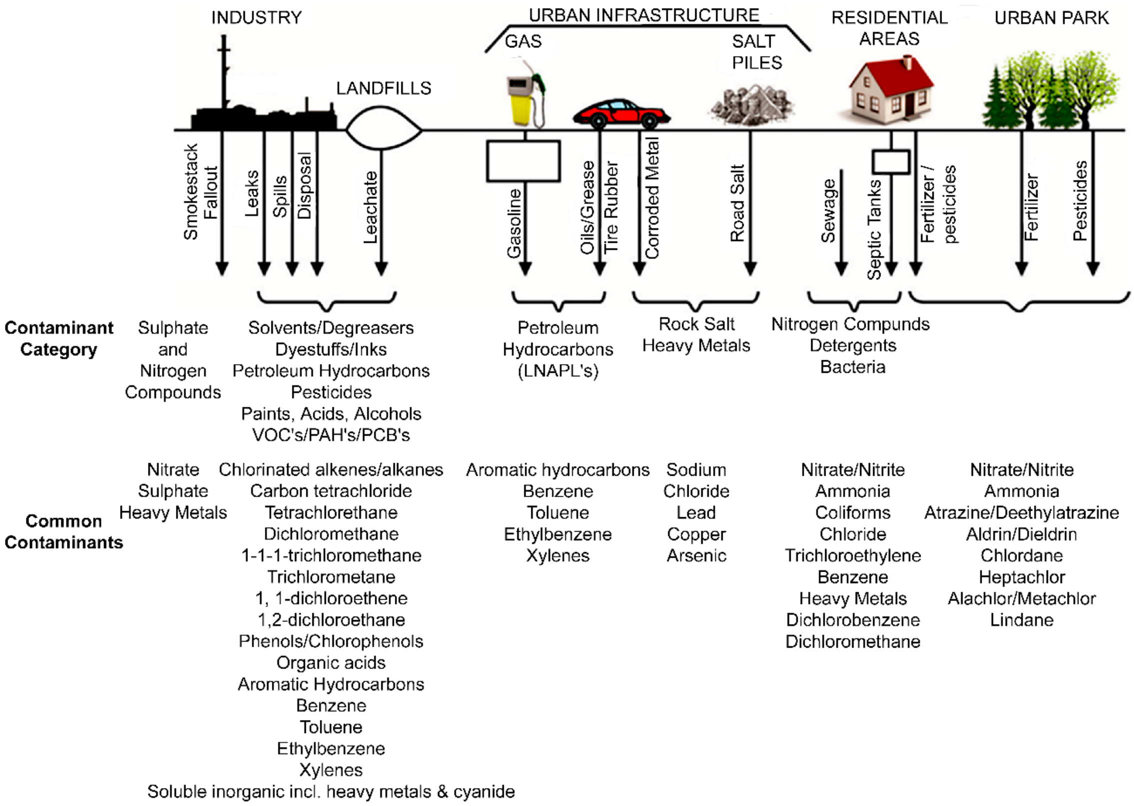

Figure 30 - Common sources of urban groundwater contamination (from Warner et al., 2016; modified after Howard, 1997).

4.1 Point, Line, and Distributed Sources of Contamination

Urban pollutant sources are normally classified into point sources that originate at a specific location and line or distributed sources, which tend to have a widespread impact on the aquifer (Table 5). Some distributed sources are a combination of many point sources spread over an area (e.g., cesspits).

Table 5 - Point, line, and distributed sources of contamination in urban areas (modified after Howard, 2002).

Point sources	Line and distributed sources
Municipal and industrial waste sites (landfills)	Liquid effluent from latrines and cesspits
Industrial discharges, leaks, and spills	Leaking sewers and discharge from septic systems
LUST (leaks from underground storage tanks) - commonly solvents, brines, gasoline, and heating fuels	Leaking canals
Winter snow dumps	Leaks from oil and chemical pipelines
Spills during road and rail transport of chemicals	Fertilizers and pesticides applied to lawns, gardens, and parkland
Leachate emanating from stockpiles of raw materials and industrial waste	Winter road de-icing chemicals
	Oil and grease from motor vehicles
	Wet and dry deposition from smokestacks
	Urban fill containing contaminated construction waste

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In many urban areas, a high density of point and line pollutant sources (e.g., multiple septic systems and networks of sewers and salted roads) combine to create a distributed source. As noted by Conant and others (2016), a remarkable number of pollutants find their way to aquifers because of intentional application or planned burial (Table 6); they are not simply the result of accidental spills and unexpected leaks. Point sources often cause severe degradation of groundwater quality at or close to the point of release but can often be remediated, as their impact is confined to a relatively discrete contaminant plume. Distributed sources, on the other hand, can be a more serious threat to aquifers as they can pollute the entire resource by elevating solute concentrations and bacterial counts to levels that exceed drinking water quality standards, albeit only marginally.

Table 6 - Classification of pollutant sources according to cause and duration (modified after Conant et al., 2016).

Cause	Duration
Accidental releases: spills, leaks, seepage from holding tanks	Single releases: spills, storage tank and pipeline ruptures
Intentional applications or releases: e.g., fertilizer and pesticide application, road de-icing chemicals, septic systems, illegal dumping	Recurring or continuous releases: septic systems, leaking sewers or leaking landfills

Numerous cases of groundwater contamination by products of urban development have been documented. In some instances, they represent a legacy of urban activity dating back several generations. Organic contamination is often considered to be the most serious, since many commonly used organic chemicals are carcinogenic, mutagenic, or teratogenic; many others are acutely toxic and can severely damage the human nervous and respiratory systems. In many cases, even very small amounts of the chemical will severely affect water potability. An example often cited is that a million liters of water will be rendered undrinkable by the organic constituents contained in just one litre of gasoline (Freeze & Cherry, 1979). An application of this concept is discussed in terms of the use of impact potentials in Section 6.3, *Water Conservation*.

From another perspective, most organic chemicals are introduced to groundwater as point sources (i.e., they emanate from a relatively discrete location such as a storage tank), and many are poorly soluble or are subject to chemical sorption, biodegradation, and volatilization, all of which can severely constrain their range of influence (e.g., Lyman et al., 1992). Thus, while some organic chemicals will always be a serious threat to groundwater quality (particularly those that are relatively mobile, chemically persistent, or are associated with regionally distributed sources such as agricultural pesticides), others will rarely move very far from source and will constitute a problem only on a very local scale, that is, for individual wells and springs.

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Significantly, the incidence of organic contamination of groundwater would appear to be far more frequent in developed than in less-developed nations. Likely, this difference largely reflects the high cost of undertaking organic analysis and the limited availability of suitable analytical facilities. Somasundaram and others (1993), for example, describe severe inorganic contamination of groundwaters underlying Madras, India but—in the absence of reliable data—could only speculate that the groundwaters are just as seriously contaminated by organic chemical species.

Compared to the more common organic pollutants, common inorganic chemicals such as chloride, sodium, sulfate, and nitrate are much less harmful to humans. In addition, common inorganic chemicals tend to be more soluble and mobile than organics and are more frequently associated with non-point contaminant sources, namely, distributed and line sources of contamination such as acid precipitation, agriculture, and de-icing salts applied to roads. As previously indicated, point sources can cause severe pollution of groundwater on a very localized scale, whereas distributed and line sources of contamination generally cause widespread contamination of water at relatively low levels. Most pragmatists would argue that once drinking water quality guidelines have been exceeded, mildly contaminated water is just as serious a concern as water that has been contaminated to a much higher degree. An exception would be chemicals that can be cost-effectively removed by end-of-pipe treatment.

Most cases of inorganic contamination of groundwater in urban areas involve the ions chloride, nitrate, and sulfate. Of these, only nitrate—and to a lesser extent, sulfate—is seriously implicated from a health standpoint. However, chloride can impart an unpleasant taste to the water, and its presence is often an indication that other more deleterious chemicals may be present.

Where they are chemically mobile, other inorganics of health-related concern include barium, lead, chromium, zinc, cyanide, arsenic, cadmium, and mercury. The presence of elevated iron and manganese is also frequently documented; however, these metals are not a health concern, and water quality standards, where established, are related to aesthetic issues such as staining of laundry and plumbing fixtures. Similarly, elevated boron is not generally regarded as a human health issue; however, its almost-ubiquitous use in industry and the home, together with its high mobility in groundwater, give it the distinction of being an excellent indicator of anthropogenic pollution. Phosphate is another widely used chemical in the urban/industrial environment and is often found in surface waters that drain urban areas. However, phosphate is rarely a concern in groundwater since the neutral to alkaline pH conditions normally encountered tend to limit its mobility.

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4.2 Primary Sources of Urban Groundwater Pollution

4.2.1 Industry and Industrial Waste

Some of the most serious cases of groundwater contamination are reported in urban centers with a long history of industrial activity (e.g., metal processing including electro-plating, electronics, textiles, laundries, vehicle construction and maintenance, printing, brewing, pharmaceuticals, food processing, tanneries). Many industries store, use, and produce a wide range of organic and inorganic chemicals and, over time, it is inevitable that a portion of this material will enter the subsurface, with the potential to degrade groundwater quality. Table 7 lists the characteristics of common industrial liquid wastes.

Table 7 - Characteristics of industrial liquid wastes (from Jackson, 1980; after Zhang et al., 2003).

Source	**Potential characteristics of effluent**
Food and drink manufacturing	High BOD (biochemical oxygen demand). Suspended solids often high, colloidal, and dissolved organic substances. Odors.
Textile and clothing	Alkaline effluent with high suspended solids, BOD, total solids, hardness, chlorides, sulphides, and chromium.
Tanneries	Alkaline effluent with high suspended solids, BOD, total solids, hardness, chlorides, sulphides, and chromium.
Adhesive/sealant	High in organic solvents.
Pulp and paper	High in inorganic salts.
Chemicals: Detergents Explosives Insecticides/herbicides Synthetic resins and fibers Ink and printing paste Acids	 High BOD. Saponified soap residues. Low pH, high organic acids, alcohols, oils. High TOC, toxic benzene derivatives, low pH. High BOD. High in solvents. Low pH.
Paint and coating	High in organic solvents, some chlorinated; heavy metals including Pb, Zn, Cr.
Petroleum, petrochemical: Refining Process	High BOD, chloride, phenols, sulfur compounds. High BOD, suspended solids, chloride, variable pH..
Foundries	Low pH. High suspended solids, phenols, oil.
Electronic manufacturing	High in copper and other heavy metals, methanol, isopropanol, fluoro- and chlorofluoro-carbon.
Plating and metal finishing	Low pH. High content of toxic heavy metals, sometimes as sludges.
Engineering works	High suspended solids, soluble cutting oils, trace heavy metals. Variable BOD, pH.
Wood treatment	Creosote, pentachlorophenol, some copper, and chromium compounds.
Thermal power	Increased water temperature. Slight increase in dissolved solids by evaporation of cooling wastes.

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For industrial waste, routes of release to the subsurface and aquifer system normally include (Zhang et al., 2003)

- disposal by flooding on industrial sites;
- disposal in pits, ponds, or lagoons;
- co-disposal (with domestic waste) in sanitary landfills;
- disposal in sealed containers in containment landfills;
- disposal at land treatment sites;
- irrigation of effluent on agricultural lands; and
- deep well injection.

In some instances, industrial waste is discharged to sewers, which, as will be discussed further, have a high propensity to leak. Table 8 lists the major sources of selected chemicals in wastewater.

Table 8 - Major industrial sources of selected chemicals (compiled from Eckenfelder, 1989; Craun, 1984; and modified after Zhang et al., 2003).

Chemical	Typical source of wastewater/effluent
Arsenic	Metallurgical industry, glassware and ceramics, tannery operations, dye stuff, pesticide manufacturing, petroleum refining, rare-earth industry
Barium	Paint and pigment industry, metallurgical industry, glass, ceramics, dye manufacturing, vulcanizing of rubber, explosives manufacturing
Cadmium	Metallurgical alloying, ceramics, electroplating, photography, pigment works, textile printing, chemical industries
Chromium	Metal-plating industries
Copper	Metal-process pickling baths and plating baths, chemical manufacturing processes
Fluoride	Glass manufacturing, electroplating, steel and aluminum, and pesticide and fertilizer manufacturing
Iron	Chemical industrial wastewater, dye manufacturing, metal processing, textile mills, petroleum refining
Lead	Storage-battery manufacturing
Manganese	Steel alloy, dry-cell battery manufacturing, glass and ceramics, paint, vanish, inks and dyes
Mercury	Chlor-alkali industry, electrical and electronics industry, explosives manufacturing, photographic industry, pesticide and some preservative industry, chemical and petrochemical industry
Nickel	Metal-processing industries, steel foundries, motor vehicle and aircraft industries, printing, and chemical industries
Silver	Porcelain, photographic, electroplating, and ink manufacturing industries
Zinc	Steelworks, fiber manufacturing, plating and metal processing industry
TCE	Industrial solvent used in dry cleaning and metal degreasing, organic synthesis
Carbon Tetrachloride	Manufacture of chloro-fluoromethanes, grain fumigants, cleaning agents, and solvents
Tetrachloroethylene	Industrial solvent used in dry cleaning and metal degreasing operations
1,1,1-Trichloroethane	Industrial cleaning, metal degreasing
1,2-Dichloroethane	Manufacture of vinyl chloride and tetraethyl lead. Also, a constituent of paint, varnish, and finish removers
Methylene chloride	Manufacture of paint and varnish removers, insecticides, and solvents

Industrial sources of contamination may also reach an aquifer system through a variety of other pathways. These may include accidental spills at industrial sites during a range of activities including various stages of the manufacturing process; accidental spills

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during transport of chemicals by road, rail, or pipelines; and leaks from storage tanks, impoundments, and treatment ponds. In many cases, serious impacts on groundwater quality have been observed (Keswick, 1984; Craun, 1984; Hirschberg, 1986; Goerlitz, 1992; Baedecker & Cozzarelli, 1992; Appleyard, 1993; Grischek, 1996; Stuart & Milne, 1997; Dragišić et al., 1997).

Industrial chemicals and chemicals in liquid form represent a particular threat due to their high mobility and tendency to mix and disperse readily in groundwater. When large volumes are released, shock-loading overwhelms the ability of the natural system to respond adequately with ameliorating processes such as chemical adsorption, biodegradation, or dilution by groundwater (Hirschberg, 1986).

Very few industrial chemicals have not been found to contaminate the subsurface at one time or another. However, many of the more toxic chemicals such as PCBs (polychlorinated biphenyls) and the larger molecule PAHs (polycyclic aromatic hydrocarbons) have a very limited solubility in water and are readily adsorbed on to clay and organic matter found in the soils and sediments. As a result, these materials rarely migrate very far from source and human exposure due to groundwater pathways tends to be rare. The most severe threat comes from toxic organic chemicals that are sufficiently soluble, mobile, and persistent in water to travel some distance from their source and enter wells, surface streams, and lakes.

In industrial environments, the chlorinated hydrocarbon solvents (CHS) represent one such group (Table 9). They belong to the DNAPL group (dense non-aqueous phase liquids) (Feenstra, 1990) and have a specific gravity greater than 1. DNAPLS are only partially soluble in water and tend to form relatively discrete plumes that gravitate towards the bottom of aquifers, sometimes invading cracks and microfractures in the underlying aquitard. In an aquifer, DNAPLs are gradually assimilated into the surrounding groundwater to produce an aqueous or dissolved phase that moves readily in accordance with the prevailing groundwater flow regime. Trichloroethylene (TCE), tetrachloroethylene (also known as perchloroethylene or PCE), 1-1-1 trichloroethane (TCA), carbon tetrachloride (CTC), and chloroform (trichloromethane or TCM) tend to be the most problematic. Such chemicals are typically released into an aquifer as point sources due to inappropriate or inadequate handling, storage, or disposal by industrial users. However, large quantities are also found in the leachate of domestic landfills (Howard & Livingstone, 1997) as discussed in Section 4.2.8. Other DNAPLs commonly found in groundwaters affected by industrial activity include the solvents 1,1-dichloroethane, 1,2-dichloroethane, chlorobenzene, and 1,2-dichloroethylene (Fusillo et al., 1985).

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Table 9 - Commonly used chlorinated hydrocarbon solvents and their properties (after Fetter, 1988, and Zhang et al., 2003). Values of specific gravity greater than 1 confirm their DNAPL status. Their solubilities in water may be low but, nevertheless, exceed drinking water guidelines by many orders of magnitude.

Compound	Specific gravity	Solubility in water (mg/L)
Carbon tetrachloride	1.59	800 (20 °C)
Chloroform	1.48	8,000 (20 °C)
Methylene chloride	1.33	20,000 (20 °C)
Ethylene chloride	1.24	9,200 (0 °C)
1, 1, 1-trichloroethane (TCA)	1.34	4,400 (20 °C)
1, 1, 2-trichloroethane	1.44	4,500 (20 °C)
Trichloroethene (TCE) (Trichloroethylene)	1.46	1,100 (25 °C)
Tetrachloroethene (PCE) (Perchloroethylene)	1.62	150 (25 °C)

In Europe, widespread contamination by CHS has been reported in heavily industrialized regions such as the Midlands of England (Rivett et al., 1989, 1990; Nazari et al., 1993; Burston et al., 1993; Ford & Tellam, 1994), and the city of Milan, Italy (Cavallaro et al., 1986). CHS contamination is also common in North America with early case studies from the USA described in New Jersey by Roux and Althoff (1980), in Indiana by Cookson and Leszcynski (1990), and in Nebraska by Kalinski and others (1994). In Australia, Benker and others (1994) report extensive CHS contamination underlying a residential area in Perth.

In Birmingham, United Kingdom, Rivett and others (1989, 1990) detected CHS in 78 percent of 59 supply boreholes tested; 40 percent of the boreholes contained TCE in excess of the 30 ug/l WHO (World Health Organization) guideline value. With respect to the 1 ug/l guideline value set for all organochlorine compounds in the European Communities drinking water directive (CEC, 1980), TCE, TCA, TCM, and PCE showed exceedance in 62 percent, 22 percent, 17 percent, and 9 percent of the boreholes, respectively. For groundwater supplies in the Coventry area, Nazari and others (1993) and Burston and others (1993) similarly reported TCE as the most common CHS of concern, closely followed by TCA. Similar results were reported for groundwater beneath Milan by Cavallaro and others (1986).

Chlorinated hydrocarbon solvents are also commonly classified as VOCs (volatile organic compounds). VOCs are a large group of chemicals that have high vapor pressures and volatilize readily under normal atmospheric pressures and temperatures. They include pesticides and petroleum hydrocarbons. VOCs are regularly found in industrial settings but can also be associated with landfills, gasoline stations, and urban parkland and gardens.

Between 1985 and 2001, the United States Geological Survey (USGS) (Zogorski et al., 2006) conducted a comprehensive investigation of groundwater quality throughout the USA involving 55 VOCs in samples from 2,401 domestic wells and 1,096 public wells, representing close to 100 different aquifers. At an assessment level of 0.2 μg/L (Figure 31)

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one or more VOCs were detected in around 25 percent of the public wells (Figure 32) and 14 percent of the domestic wells (Figure 33). Trihalomethanes, solvents, and the gasoline oxygenate MTBE were among the 15 VOCs most frequently detected.

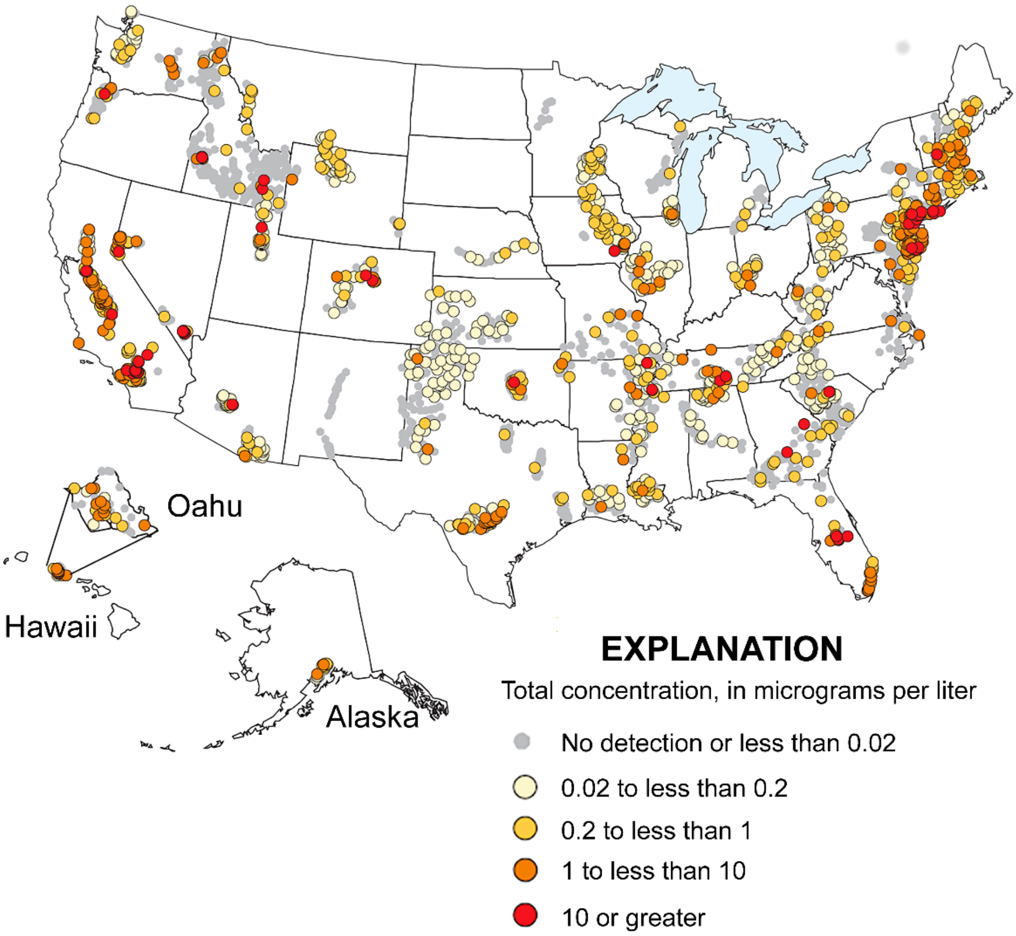

Figure 31 - VOC contamination of groundwater was found throughout the USA (after Zogorski et al., 2006).

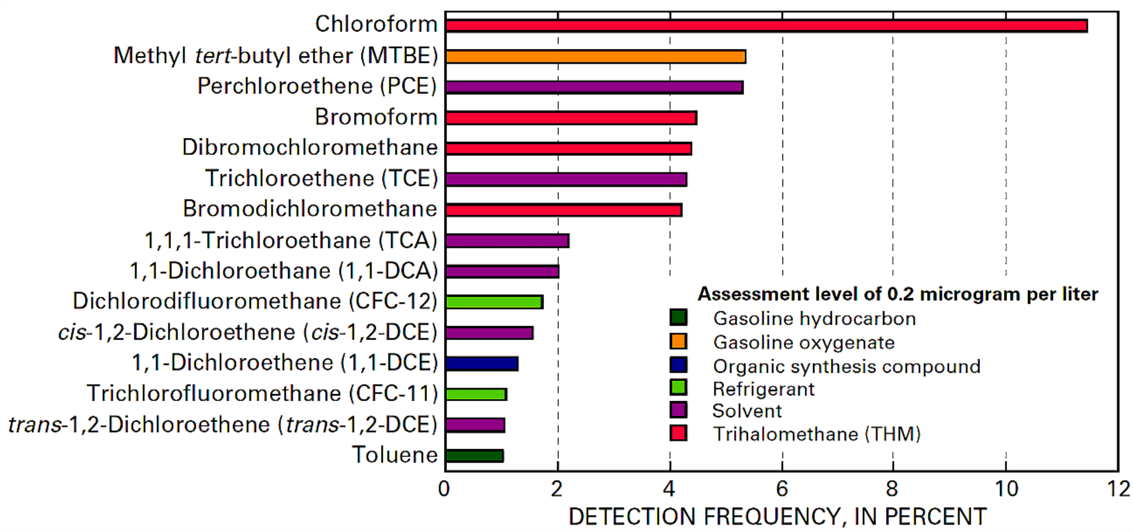

Figure 32 - Detection frequencies in samples from the public supply wells at a detection level of 0.2 µg/L Trihalomethanes, solvents, and the gasoline oxygenate MTBE were among the 15 most frequently detected VOCs (after Zogorski et al., 2006).

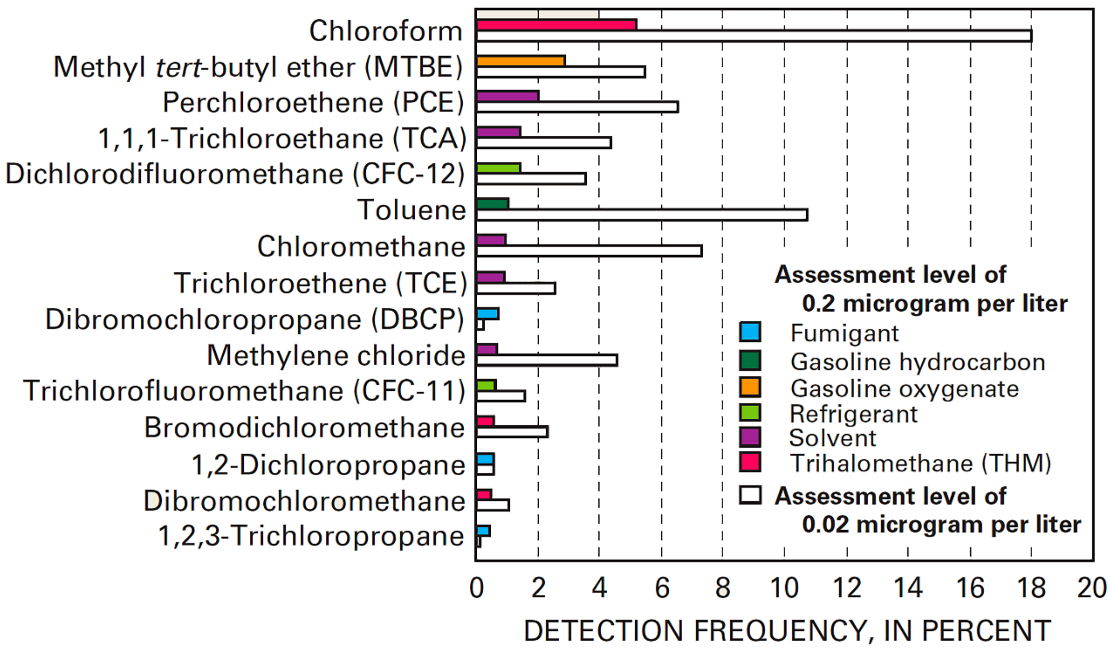

Figure 33 - Detection frequencies in samples from the domestic wells at assessment levels of 0.2 and 0.02 µg/L (after Zogorski et al., 2006).

Groundwater contamination by inorganic chemicals is also regularly reported in industrialized areas, with heavy metals, cyanide, and boron among the most common contaminants of concern (Velea et al., 2009; Benhaddya & Hadjel, 2013). Heavy metals are toxic to living organisms when they exceed certain thresholds (Gao & Chen 2012; Raju et al., 2012) and aquatic ecosystems are particularly at risk. Unlike organic pollutants, heavy metals do not biodegrade and can persist in the soil for long periods of time (Klimek, 2012). For example, Boudissa and others (2006) reported extremely high levels of manganese in

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soil and water near an abandoned manganese alloy operation in Montreal, Canada, despite the plant having been closed for more than ten years.

In Madras, India, Somasundaram and others (1993) associated high groundwater concentrations of arsenic, mercury, lead, and cadmium with a combination of industrial activities and inadequate waste disposal facilities. A lack of sewers in industrial areas has also been blamed by Foster (1990) for heavy metal contamination of groundwaters in parts of South America. In Odessa, Texas, USA, severe contamination of groundwater by hexavalent chromium (in some places as high as 72 mg/L) was linked with the direct application of wastewater and rinse water from chromium plating operations to the soil (Henderson, 1994).

In Birmingham, United Kingdom, (Ford & Tellam, 1994) and Coventry, United Kingdom, (Nazari et al., 1993)—two of the largest and oldest industrial centers in Europe—inorganic contamination of groundwater was not found to be a widespread problem. Only a few wells showed trace metal concentrations that exceeded drinking water quality standards. Where elevated concentrations of zinc, copper, chromium, nickel, and cadmium were found to occur, large metal industry sites appeared to be responsible. A particular concern in the Birmingham area is that many of the metals exhibit a mobility far greater than would be expected in an aquifer known for its near-neutral pH and high sorption capacity. Ford and Tellam (1994)speculated that the enhanced mobility can be explained by high metal supply rates, colloidal transport and/or complexation conditions and, in the case of chromium, its anionic form. Cronin and Lerner (2004) undertook a review of urban groundwater contamination issues in several mature industrial cities in the United Kingdom and drew attention to a lack of political will and a lack of resources for water quality monitoring and enforcement of protection measures.

4.2.2 Underground Storage Tanks (USTs)

Across urban communities and along major highways, large volumes of gasoline are stored in underground storage tanks (USTs). Fetter (1993) estimated that as many as 2.5 million such tanks exist in the USA. Of these, the USEPA has estimated that as many as 35 percent eventually lose product in some way (Figure 34), usually due to spills or leaks associated with pressure delivery pipes. Human error is a major cause.

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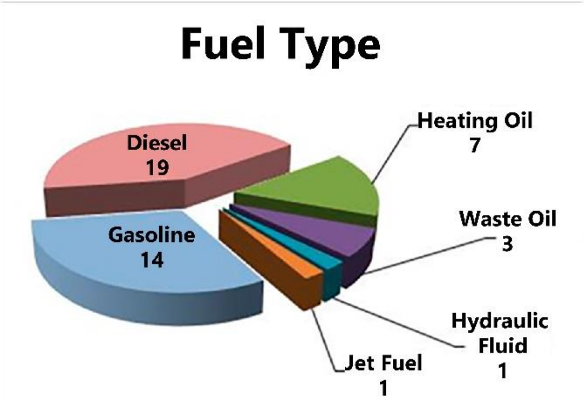

Figure 34 - In 2017, the Montana Department of Environmental Quality (USA) reported a 10-year high of 45 confirmed petroleum releases from underground storage tanks, with the product types displayed in the pie chart. Forty percent of the cases were associated with corrosion and equipment failure; over 50 percent were due to negligence (photography by Montana Department of Environmental Quality).

While some of the product loss will vaporize to the atmosphere, a significant proportion will pass through the vadose zone and move toward the water table. From a water quality standpoint, the potential threat comes from two component groups:

- gasoline hydrocarbons, namely, benzene, n-butylbenzene, ethylbenzene, isopropylbenzene, naphthalene, styrene, toluene, 1,2,4-trimethylbenzene, o-xylene, m-xylene, and p-xylene; and
- gasoline oxygenates (as additives), that is, methyl tert-butyl ether (MTBE), tert-amyl methyl ether (TAME), diisopropyl ether (DIPE), and ethyl tert-butyl ether (ETBE).

In terms of the gasoline hydrocarbons, most attention is normally given to four relatively soluble aromatic hydrocarbons: benzene, toluene, ethylbenzene, and xylene(s), often referred to collectively as BTEX. BTEX chemicals are less dense than water and thereby belong to a group of organic liquids broadly known as LNAPLs (light non-aqueous phase liquids) that move less readily through the vadose zone than their heavier (DNAPL) counterparts; LNAPLs tend to float on the water table or on top of the capillary fringe. If a sufficient thickness of LNAPL collects, it can flow in the direction of the downward-sloping water table as a non-aqueous contaminant plume. Of greater concern are the water-soluble components of the LNAPL, which dissolve in the underlying groundwater and migrate within the body of the aquifer as an aqueous phase.

Based on an audit of groundwater contamination sources in Toronto, Canada, Howard and Livingstone (1997, 2000) ranked BTEX released from leaking underground storage tanks (LUSTs) as one of the most serious potential threats to urban groundwater quality. Fortunately, the severity of the problem is frequently tempered by volatilization, sorption, and biodegradation processes. Volatilization is most effective when the contaminants are in contact with air in soil pores in the vadose zone above the water table. However, the importance of volatilization will diminish considerably when the contaminants enter the aqueous phase. Sorption, by comparison, can virtually immobilize

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hydrocarbons that have low water solubilities and low vapor pressures but has little impact on the relatively soluble and volatile BTEX chemicals other than to slow their migration.

Biodegradation (Davis et al., 1994) appears to be the key process if serious degradation of water quality is to be averted. Barker and others (1987) have shown that BTEX will readily biodegrade if aerobic conditions are maintained; Chiang and others (1989) have shown that the process is very effective provided dissolved oxygen concentrations exceed 0.9 mg/L. Hadley and Armstrong (1991) suggested that biodegradation can explain the unexpected absence of benzene in groundwaters underlying parts of California, USA.

With respect to the gasoline oxygenates, only MTBE—noted for its relatively high solubility—appears to be a serious concern. Gasoline oxygenates are compounds that contain oxygen and are added to gasoline to improve combustion and reduce harmful motor vehicle emissions. MTBE was first introduced in gasoline in 1979 as an octane enhancer resulting from the phase-out of leaded gasoline. In the USGS study of groundwater contamination by VOCs described in Section 4.2.1, *Point, Line, and Distributed Sources of Contamination* (Zogorski et al., 2006), MTBE was the second most frequently detected VOC in samples from domestic and public wells at an assessment level of 0.2 μg/L. The detection frequency of MTBE ranged from ≈3 percent in domestic well samples to ≈5 percent in public well samples.

Public wells tend to have larger pumping rates than domestic wells, draw water from much larger areas, and therefore are more prone to be affected by releases of MTBE. In 2004, MTBE was the gasoline oxygenate most used in the USA. Since then, MTBE has been completely or partially banned in many American states. Globally, MTBE is still used widely and the USA is a major exporter—primarily to Mexico. Demand is especially high from China, Singapore, South Korea, and other countries in the Asia Pacific region.

4.2.3 Disposal of Domestic Sewage

Recognizing that urban areas are characterized by large, often dense, human populations, domestic sewage represents a major potential source of groundwater contamination. In most cities, domestic sewage consists of human body waste, kitchen waste, and waste from personal washing and laundry (Table 10). In terms of human body waste, about 100 to 250 grams (3 to 8 ounces) of feces are excreted by a human adult daily. This waste is generally characterized by high BOD (biochemical oxygen demand) and the significant presence of suspended solids, fecal bacteria, chloride, and ammonia (Zhang et al., 2003).

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Table 10 - Pollution loads from various plumbing fixtures in the USA (mg/d) (modified after Dillon, 1997, and Zhang et al., 2003).

Wastewater source	**Biochemical oxygen demand (BOD)**		**Chemical oxygen demand (COD)**		**NO-N**		**NH-N**		**Phosphates**	
	Mean	%	Mean	%	Mean	%	Mean	%	Mean	%
Bathroom sink	1,860	4	3,250	2	2	3	9	0.3	386	3
Bathtub	6,180	13	9,080	8	12	16	43	1	30	0.3
Kitchen sink	9,200	19	18,800	16	8	10	74	2	173	2
Laundry machine	7,900	16	20,300	17	35	49	316	10	4,790	40
Toilet	23,540	48	67,780	57	16	22	2,782	87	6,473	55
Totals:	48,680	100	119,210	100	73	100	3,224	100.3	11,852	100

As a pollutant source, sewage is characterized by the presence of microorganisms and an extensive range of organic and inorganic chemicals (Howard, 1993; Lerner, 1994). Barber (1992) found more than 200 organic compounds in groundwater contaminated by sewage, including chlorinated hydrocarbons (aliphatic and aromatic), alkyl-substituted hydrocarbons (aliphatic and aromatic), alkylphenols, aldehydes, and phthalate esters. TCE and PCE were observed at greater concentrations than most other compounds and were considered the greatest water quality threat. Analyses of modern-day sewage shows an increasing presence of pharmaceuticals and personal care products (PPCPs), which have been categorized by the USEPA as emerging contaminants of concern because so little is known about their impact on the environment or risk to human health when they are released into ecosystems.

Inorganic constituents of domestic sewage also tend to be wide-ranging and include compounds that contain sodium, potassium, calcium, magnesium, boron, chloride, sulfate, phosphate, bicarbonate, and ammonia (Table 11). Synthetic detergents, where present, are a major source of phosphate (Table 10) but can also introduce significant amounts of chloride, sulfate, and boron. In Cairo, Egypt, Shahin (1990) and Alderwish and others (2003) report that sulfate derived from leaking sewers is responsible for severe damage to the concrete foundations of buildings.

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Table 11 - Mean sewage composition and mean annual input from damaged sewers into soil and groundwater in Rastatt and Hannover, Germany (from Eiswirth and Hötzl, 1997).

Compound	Mean concentration in sewage (mg/L)	Mean annual input (kg/ha/a)
Potassium	38	55
Sodium	111	153
Ammonium-N	55	80
Organic-N	23	33
Chloride	101	150
Nitrate	7	10
Sulfate	38	55
Boron	1.96	3
Phosphate	11	16
Lead	0.034	0.48
Cadmium	0.005	0.007
Chromium	0.001	0.001
Copper	0.062	0.09
Nickel	0.027	0.05
Zinc	0.85	1.2

A complicating factor is that sewage may also contain varying amounts of industrial waste (e.g., spent liquors from industrial processes). The proportion of these wastes will be strongly influenced by local laws and environmental regulations that dictate the nature and volumes of industrial waste that can be released without pre-treatment or specialized disposal. Sewage may also contain various amounts of urban runoff and thereby include a wide range of potential pollutants including fertilizers and pesticides.

A particular concern is that domestic sewage is a major source of microorganisms that cause serious waterborne illnesses throughout the world (Craun, 1984; Keswick, 1984; Powelson & Gerba, 1985). Diseases associated with contaminated groundwater may be bacterial (shigellosis, salmonellosis, and typhoid fever), parasitic (giardiasis), or viral (hepatitis A). According to Keswick (1984), 264 outbreaks and 62,373 cases of waterborne diseases due to contaminated groundwater were reported in the USA between 1946 and 1977.

Of the three types of microorganisms regularly found in sewage, bacteria are significantly more common than viruses, and parasites. Amongst the millions of microorganisms believed to be present are potentially pathogenic bacteria such as *Salmonella, Shigella, Vibrio cholerae*, and *Escherichia coli,* while polio virus, hepatitis A, and virus groups such as echo, coxsackie, rota, adeno, and Norwalk have also been found (Keswick, 1984).

Microorganisms move through the subsurface as suspended particles in water. Although normally associated with a host, pathogenic bacteria can survive and multiply in the subsurface environment where the conditions permit (Keswick, 1984). Viruses can also

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survive for long periods in the subsurface environment but do not replicate in the absence of a host. In the literature, several cases have been documented where bacteria and viruses have traveled over 400 m in sand, gravel, and limestone aquifers (Keswick, 1984; Hagedorn, 1984; Yates & Yates, 1988). The longest distance reported is 1,600 m for a coliphage in a limestone aquifer (Yates & Yates, 1988).

Rates of removal of microorganisms have been studied in the field, laboratory, and water treatment plants by a variety of techniques (Dillon, 1997). Observed complete removal rates ranged from 0.7 days for coliphage, and from 2 to 33 days for polio virus and fecal streptococci. In general terms, the survival and migration of microorganisms are determined by the specific type of microorganism, the soil conditions, and climate (Yates & Yates, 1988).

Typically, factors that favor the survival and transport of bacteria include

- moist conditions, preferably saturation;
- low temperatures;
- alkaline conditions (pH > 7);
- lower salinities; and
- presence of organic matter.

For viruses, their survival and transport are enhanced by

- moist conditions, preferably saturation,
- near neutral to alkaline pH,
- lower salinities,
- high dissolved organic carbon, and
- coarse-textured soils with a low clay content.

Advances in analytical facilities now allow hormones and various classes of pharmaceuticals and personal care products (PPCPs) to be readily detected in groundwater contaminated by sewage (Held et al., 2006; Reinstorf et al., 2007). Pharmaceuticals found in sewage may include (Koopaei & Abdollahi, 2017)

- antibiotics (clarithromycin, ciprofloxacin, doxycycline, erythromycin, metronidazole, norfloxacin, ofloxacin, roxithromycin, sulfamethoxazole, sulphapyridine, tetracycline, trimethoprim),
- antiepileptics (carbamazepine),
- anticoagulants (warfarin),
- analgesics and anti-inflammatories (4-aminoantipyrine, antipyrine, codeine, diclofenac, ibuprofen, indomethacin, ketoprofen, ketorolac, naproxen),
- lipid regulators (clofibric acid, fenofibric acid, bezafibrate, gemfibrozil, ezetimibe),
- steroidal compounds (estrogenic and androgenic drugs);
- beta-blockers (acebutolol, atenolol, celiprolol, metoprolol, propranolol, sotalol),
- diuretics (furosemide, hydrochlorothiazide),

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- contrast media (amidotrizoic acid, diatrizoate, iotalamic acid, iopromide, iomeprol, iohexol, iopamidol),
- cosmetics/fragrances (synthetic musk compounds: galaxolide, tonalid);
- psycho-stimulants (caffeine, paraxanthine), and
- antidepressants (fluoxetine).

The trace presence of hormones and pharmaceuticals (and their metabolites) in contaminated groundwater can be a useful forensic indicator that domestic sewage is a contributory pollutant source.

4.2.4 Exfiltration from Sewers and Sewage Canals

In terms of contaminant pathways, the most severe cases of groundwater contamination due to sewage are associated with losses from underground sewers (isolated point or line sources) and surface sewage canals (normally line sources). Contamination by leaking sewers, shown in Figure 35 (Allen, 2007; Held et al., 2006; Mohrlok et al., 2007), is a common feature of heavily urbanized areas.

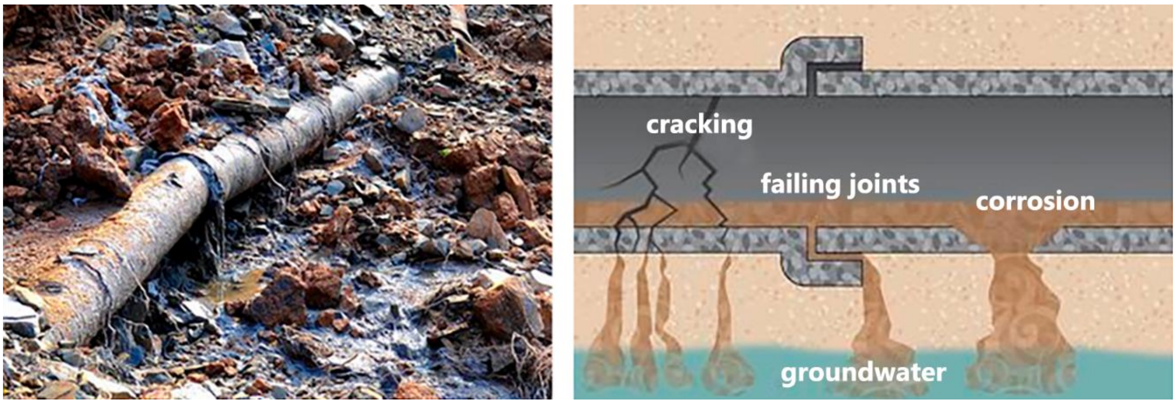

Figure 35 - A 2017 study undertaken by staff at the City of Albuquerque, Texas, USA, estimated that the sewage collection system lost 11 percent of wastewater due to exfiltration (photography from personal collection of Ken Howard; diagram from Bhatia, 2017).

In the Federal Republic of Germany, Eiswirth and Hötzl (1994) estimated that several hundred million cubic meters of wastewater leaked from partly damaged sewage systems each year and posed a significant threat to groundwater quality. Water balance studies in the Rastatt, Germany, where natural recharge is in the range 90 to 200 mm/a (Figure 36), suggest that sewage exfiltration may contribute an additional 2 to 65 mm/a (Wolf et al., 2006).

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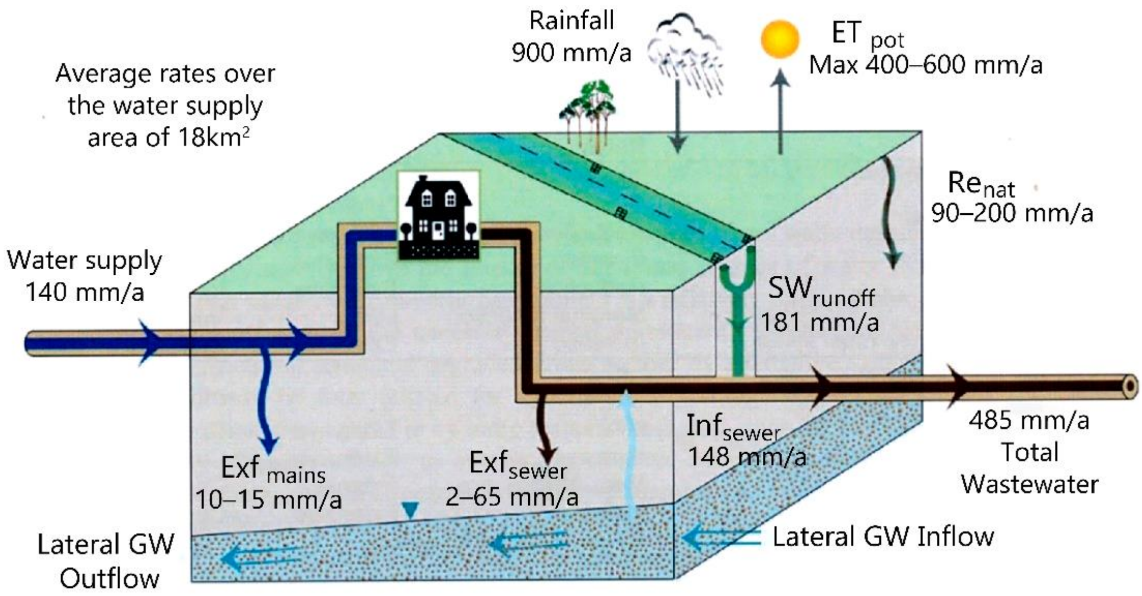

Figure 36 - Water balance studies in the City of Rastatt, Germany, where natural recharge is in the range 90 to 200 mm/a, suggest that sewage exfiltration may contribute an additional 2 to 65 mm/a. ET is evapotranspiration; Inf and Exf are infiltration and exfiltration, respectively; Re is recharge; GW is groundwater (modified after Wolf et al., 2006).

Impacts from sewage wastes are even more serious in many of the less-developed nations where drainage ditches, unlined open canals, and rivers are frequently used to export a large variety of wastes, including sewage, from major urban centers. In Brazil, the Paraíba do Sul river passes through the industrial towns of Barra Mansa and Volta Redonda where a population of over 250,000 contributes 14,200 kg of BOD (biochemical oxygen demand) and 1,790 kg of nitrogen each day (Foster, 1990; Hydroscience Incorporated, 1977).In such situations, even small amounts of leakage to the subsurface can generate severe groundwater quality problems.

In Mexico City, 90 percent of the sewage is discharged untreated into a sewer system that relies heavily on the use of unlined drainage canals. One problem is that land subsidence due to excessive groundwater pumping in central parts of the city has locally caused a reversal of flow in these canals; a series of continually operating pumping stations is now required to maintain the outward flow of wastewater. Leakage through the walls of these canals returns a significant quantity of contaminated water to the aquifer. Excessive nitrate, for example, has been observed in wells located adjacent to the Chalco Canal, one of the main passageways for wastewater leaving the city. The dilemma facing the Mexican authorities is that while leakage of wastewater may degrade groundwater quality, local groundwater resources are so seriously overexploited that groundwater recharge is at a premium.

4.2.5 Septic Systems and Pit Latrines

The other primary pathway for the passage of domestic sewage into the groundwater system is provided by septic systems and latrines. Where centralized sewage facilities are unavailable, individual septic systems (septic tanks and drain tile fields) are

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used worldwide for the disposal of wastewater. In the USA, for example, it has been estimated that septic systems are used by approximately one-third of the population and release 800 billion US gallons of domestic waste, including sewage, to the subsurface every year (Fetter, 2001). When Canada is also included, North American septic tanks may likely constitute as many as 20 million potential point sources of groundwater contamination. In practice, the high density of septic systems found in many urban areas tends to generate a distributed, diffuse, or non-point source of contamination as opposed to a multitude of point sources.

In principle, septic systems operate by discharging wastewater into a tank where fats float to the top and solids settle as a sludge and begin a process of anaerobic decomposition. The remaining liquid effluent is directed into a leach bed, which is a system of buried open-jointed tiles or perforated pipes arranged in parallel rows; from there the liquid drains vertically through the unsaturated zone to the water table. Theoretically, aerobic oxidation of organic carbon and ammonia in the leach bed produces carbon dioxide and nitrate. In turn, the nitrate is reduced to nitrogen gas as the wastewater moves into the saturated zone beneath the drain field. If the system is constructed according to local guidelines/regulations and works efficiently, natural processes should deplete contaminants in the effluent to harmless levels within a short distance of the leaching bed.

Unfortunately, many septic systems do not function the way they were designed. This is especially problematic in jurisdictions with little regulatory control. In some cases, the septic tanks are unable to cope with the volumes and types of waste generated by modern households equipped with dishwashers, automatic washing machines, and oversize bathtubs. At other times, geochemical conditions in the soils and sediments fail to provide the degree of natural attenuation necessary to adequately treat the liquid effluent. For example, as described by Wilhelm and others (1994), if adequate oxygen is not available in the leach bed, aerobic digestion is incomplete, and the accumulation of organic matter can cause the system to fail. Also, the reduction of nitrate to nitrogen gas is frequently inhibited by the lack of labile carbon in the natural setting. For example, in Canada, contaminant plumes over 100 m in length have been observed in sand aquifers (Robertson et al., 1991). Typically, these plumes were found to show depressed levels of pH and dissolved oxygen, and elevated concentrations of chloride, nitrate, sodium, calcium, potassium, alkalinity, and dissolved organic carbon (DOC).

While the precise nature of the dissolved organic carbon was not identified in the Canadian study, the use of septic tank cleaning fluids containing trichloroethylene, benzene, and methylene chloride has been blamed for the contamination of groundwater by these chemicals on Long Island, New York, USA (Eckhardt & Oaksford, 1988). In a study of septic systems at two sites in Australia, Hoxley and Dudding (1994) additionally reported contamination of groundwaters by fecal bacteria. In one case, *E. coli* was detected at distances greater than 500 m from the suspected source.

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In the USA, an estimated 20 percent of septic systems release excessive amounts of contaminants due to poor design, poor maintenance, and inappropriate site conditions (Council of Canadian Academies, 2009; International Joint Commission, 2011). Common problems include high water tables and very permeable soils, both of which can constrain opportunities for the strong attenuation of contaminants.

The primary contaminants associated with septic systems were found to be nitrate, bacteria, and viruses, but elevated concentrations of potassium, boron, chloride, dissolved organic carbon, and sulfate were also identified (Katz et al., 2011). In a study near Detroit, Michigan, USA, Thomas (2000) suggested that degradation of shallow groundwater quality was mostly due to septic system effluent (domestic sewage, household solvents, water softener backwash), and infiltration of stormwater runoff from paved surfaces (road salt, fuel residue). Murray and others (2001) suggest that contaminated groundwater is responsible for high levels of fecal coliform observed in some Michigan streams, including the Rouge River.

Pit latrines (also called soak pits) are the low-technology equivalent to septic systems and are in common use throughout many low- and middle-income countries, including many highly populated urban and peri-urban areas (Barrett & Howard, 2002). Pit latrines are the most rudimentary disposal system and often comprise little more than a shallow hole in the ground. They represent a particularly serious health hazard when built near wells. Major concerns include thermotolerant coliform (TTC), chloride, nitrate-N, and ammonium-N.

Table 12 shows the results of an assessment of the impact of pit latrines on shallow groundwater in Hopley Settlement, Harare, Zimbabwe (Ndoziya et al., 2019). Unacceptable levels of contamination are observed at virtually all the sites tested, with TTC a prominent concern.

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Table 12 - An assessment of the impact of pit latrines on shallow groundwater in Hopley Settlement, Harare, Zimbabwe (after Ndoziya et al., 2019). Sample sites B1, B2, and W3 were located 40 m, 28 m, and 25 m, respectively from the nearest latrine. The remaining sample sites were situated within 15 m of a latrine, W1 being just 3.5 m away. Also included are WHO and SAZ 560 drinking water quality standards.

Sampling point	TTC (cfu /100 mL sample)	Ammonia (mg/L)	Nitrates (mg/mL)	Chlorides (mg/L)	EC (μS/cm)	DO (mg/L)	pH	Turbidity (NTU)
B1	0	0.04	27.9	144.7	1,011	2.1	6.68	31.0
B2	5	0.08	27.9	37.2	300	3.3	6.88	5.2
B3	9	0.08	13.2	42.5	313	3.8	7.13	1.1
W1	240	4.70	191.9	278.3	1570	2.8	6.65	7.0
W2	153	0.46	271.2	173.7	968	4.2	6.15	3.1
W3	24	0.34	50.2	88.6	529	5.4	6.63	2.0
W4	19	0.04	59.2	31.9	250	3.5	6.13	1.3
W5	64	0.06	70.9	31.9	250	3.3	6.73	1.1
W6	155	0.02	68.6	46.1	268	4.1	6.38	2.3
W7	5	0.04	55.5	44.3	336	4.1	6.40	1.9
W8	223	0.10	47.7	75.2	531	4.2	6.65	1.3
Mean	82	0.54	80.4	90.4	575	3.7	6.58	5.2
WHO (2011)	0	< 0.2	50	< 300	400	-	6.5–8.5	< 5
SAZ560(1997)	0	-	10	< 250	700	> 5	6.5–8.5	< 1

"-" indicates values not specified.

4.2.6 Chemical Fertilizers

In high-income countries, the application of chemical fertilizers to urban parks and gardens often competes with septic systems and leaking sewers as the primary source of urban groundwater contamination by nutrients. Nutrients are chemical substances that promote growth in living organisms. In the context of water pollution, the term is usually applied to dissolved inorganic ions that contain nitrogen (N), phosphorus (P), and potassium (K). When nutrient loadings to urban aquifers are primarily associated with sewage waste, water quality concerns usually pale in significance to threats imposed by other sewage components such as fecal coliforms. However, when nutrient loadings in urban areas are mostly a consequence of fertilizers applied to urban parks and gardens, serious consideration must be given to potential impacts of nutrients on

- drinking water quality and
- surface water bodies such as streams and lakes that receive discharging urban groundwater.

In terms of drinking water quality, only the compounds of nitrogen (typically nitrate, nitrite, and ammonium ions) merit serious consideration. The mobility of phosphorus (usually as phosphate) is limited, especially in soils; potassium, while more mobile, is essentially harmless to humans (Lerner et al., 1999). The primary concern is the association between elevated nitrate concentrations in ingested water and a disease known as methemoglobinemia, nitrate cyanosis, also known as blue baby syndrome (Comley, 1945; Spalding & Exner, 1993).

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The disease is caused by the bacterial reduction of nitrate to nitrite in the intestinal tract. The nitrite enters the blood stream and combines with the hemoglobin to form methemoglobin, which reduces the ability of the blood to transport oxygen. The reduction of nitrate to nitrite occurs primarily in very young children because the lower acidity of their gastric juices provides a better environment for nitrate-reducing bacteria. While older children and adults can tolerate relatively high nitrate levels in drinking water, additional concerns have been raised that nitrate may play a role in the production of nitrosamines in the stomach, which are known carcinogens (Hill et al., 1973).

In its uncomplicated form, methemoglobinemia is usually easily diagnosed and its treatment understood. However, to minimize health risk, most jurisdictions throughout the world recommend that levels of nitrate nitrogen ($NO_3^- N$) in drinking water should not exceed 10 mg/L (which is about 50 mg/L nitrate, NO_3). Ammonium nitrogen ($NH_4^- N$) is also undesirable in water for public supplies primarily because of taste and odor that are detectable above concentrations of about 0.5 mg/L.

In many cities, groundwater is not used for public water supply. However, local rivers, streams, and lakes can be valuable from an aesthetic and recreational standpoint, and these delicate ecosystems can be seriously degraded by excessive nutrient concentrations. Most surface water bodies are oligotrophic, meaning that the supply of nutrients is low, photosynthetic production is minimal, and the water remains oxygenated at all depths (Drever, 1997). When nutrients are introduced in excessive amounts, the waterbody becomes eutrophic, and the enhanced biological activity often produces massive algal blooms (Harris, 1994). The rapid growth in biomass is followed by an equally rapid death, and the process of organic decomposition consumes large amounts of oxygen. This commonly creates anaerobic conditions, particularly in the bottom sediments, and normal aquatic life dies off (Figure 37).

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Figure 37 - Lake Dora, Florida, USA, is a highly eutrophic lake that has been “fed” too many nutrients and loses so much dissolved oxygen that normal aquatic life dies off (photography by Nara Souza, Florida Fish and Wildlife Commission).

First documented in the 1980s, contamination of groundwaters by the excessive use of fertilizers has become a common problem in urban areas. In most cases, the fertilizer is applied by urban residents to maintain grass lawns (Morton et al., 1988). However, indiscriminate and excessive use, combined with intermittent heavy watering, frequently leads to large leaching of fertilizers into groundwater. In Long Island, New York, USA, Flipse and others (1985) showed that fertilizer use could account for over 70 percent of the nitrate detected in groundwaters beneath a sewered housing development. In Perth, Australia, Sharma and others (1994) installed suction lysimeters beneath fertilized urban lawns and showed that between 16 and 47 percent of incident water passed below the root zone, carrying nutrients with flow-weighted nitrate-N concentrations up to 5.37 mg/L. In Toronto, Canada, shallow urban springs regularly reveal elevated nitrate concentrations, locally in excess of 29 mg/L nitrate nitrogen (Howard & Taylor, 1998). While the use of fertilizers was strongly suspected to be the primary source of the nitrate contamination, a chemical audit of potential contaminant sources in the area (Howard & Livingstone, 1997) showed that landfills and septic tanks may also play a significant role.

4.2.7 Pesticides

The term pesticide is used to describe those chemicals that control insects, weeds, and a wide variety of life forms that negatively influence the production of crops. The term may also include chemicals used by consumers for such purposes as the extermination of

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termites and roaches, removal of mold from shower curtains, destroying crab grass, killing fleas on pets, and disinfecting swimming pools. By design, pesticides tend to be toxic to living organisms, either selectively (narrow-spectrum pesticides) or non-selectively (broad-spectrum pesticides). Consequently, their chemical behavior and environmental fate have become popular foci for research.

A full list of potential pesticides is shown in Table 13. The vast majority of pesticides are designed to prevent, destroy, or simply control undesirable insects (insecticides); plant life including algae and weeds (herbicides); rodents (rodenticides); and molds, mildew, and fungi (fungicides). Other pesticides of interest include nematicides for the control of nematodes; defoliants designed to strip the leaves from trees and woody plants; and various poisons or repellents for the control of undesirable amphibians, reptiles, birds, fish, mammals, and invertebrates.

Table 13 - Common types of pesticide (US Environmental Protection Agency (USEPA), 2023).

Type	Action
Algicides	Control algae in lakes, canals, swimming pools, water tanks, and other sites
Antifouling agents	Kill or repel organisms that attach to underwater surfaces such as boat bottoms
Antimicrobials	Kill microorganisms (such as bacteria and viruses)
Attractants	Attract pests (for example, to lure an insect or rodent to a trap, however, food is not considered a pesticide when used as an attractant
Biopesticides	Biopesticides are pesticides derived from natural materials such as animals, plants, bacteria, and certain minerals
Biocides	Kill microorganisms
Defoliants	Cause leaves or other foliage to drop from a plant, usually to facilitate harvest
Disinfectants and sanitizers	Kill or inactivate disease-producing microorganisms on inanimate objects
Fungicides	Kill fungi (including blights, mildews, molds, and rusts)
Fumigants	Produce gas or vapor intended to destroy pests in buildings or soil
Herbicides	Kill weeds and other plants that grow where they are not wanted
Insecticides	Kill insects and other arthropods
Miticides	Kill mites that feed on plants and animals
Microbial pesticides	Microorganisms that kill, inhibit, or out-compete pests including insects or other microorganisms
Molluscicides	Kill snails and slugs
Nematicides	Kill nematodes (microscopic, worm-like organisms that feed on plant roots)
Ovicides	Kill eggs of insects and mites
Pheromones	Biochemicals used to disrupt the mating behavior of insects
Repellents	Repel pests including insects (such as mosquitoes) and birds
Rodenticides	Control mice and other rodents
Slimicides	Kill slime-producing microorganisms such as algae, bacteria, fungi, and slime molds

From “Types of Pesticide Ingredients,” by US Environmental Protection Agency (US EPA), 2023 (Types of Pesticide Ingredients | US EPA).

The use of chemicals to eliminate perceived pests dates back several thousands of years. The worldwide production of pesticides grew from a scant 100,000 tons in 1945 to over 1 million tons in 1965 and 1.8 million tons by 1975. Towards the end of the twentieth century, pesticide use stabilized with ≈2.5 million tons of chemical used annually throughout the world (Chiras, 1998). In 2006 and 2007, ≈2.4 million tons of pesticides were

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applied globally, with herbicides in predominant use at ≈40 percent, followed by insecticides (17 percent) and fungicides (10 percent).

Although thousands of chemical pesticides have been synthesized and tested over the years, the list of chemicals approved for use has changed considerably in response to a growing volume of environmental and toxicological research data. Chlorinated hydrocarbons such as DDT (dichlorodiphenyl trichloroethane), aldrin, kepone, dieldrin, chlordane, heptachlor, endrin, lindane, toxaphene, and mirex (Moore & Ramamoorthy, 1984) once dominated the market. However, their chemical persistence in the environment, their tendency to accumulate in the food chain, and their ability to cause cancer, birth defects, and neurological disorders has—in most countries—led to their strict control or outright banishment. As a demonstration of their persistence, concentrations of DDT, lindane dieldrin, and endrin in Lake Ontario, Canada, remained marginally below established drinking water quality standards in 1983 despite being banned or severely restricted over ten years earlier (Biberhofer & Stevens, 1987).

In more recent times, there has been a shift towards to the use of organophosphates (e.g., malathion and parathion) and carbamates (e.g., carbaryl distributed as the brand name Sevin). Both these groups tend to degrade very rapidly (in a matter of days or weeks) and thus do not persist in the environment. However, they are water soluble and highly toxic at very low levels. Moreover, questions have been raised as to their ability to break down under conditions of low temperature, low light, low dissolved oxygen, and low biological activity that are commonly encountered in groundwater. Today, the situation is confused, with little consistency between one country and another in terms of which pesticide is considered safe and which is not.

For example, in 2013, Sevin (carbaryl) was outlawed in many countries including the United Kingdom, Denmark, Australia, Germany, Sweden, Iran, and Angola but remained widely used in the USA. Similarly, Paraquat, a pesticide linked to Parkinson's disease, was permitted in the USA but banned in China and the European Union. In Canada, concerns were raised that pesticides containing malathion should not be used if the product is over a year old, since a harmful breakdown product (isomalathion) may form, particularly when stored at elevated temperatures. The situation remains in flux, with some countries revisiting and removing bans previously instituted (e.g., atrazine). In some cases, specific pesticides are permitted for agricultural use but banned for cosmetic use in the home, particularly in densely populated urban areas. The most recent pesticide regulations for the USA can be found at the USEPA website↗.

Pesticide sources and pathways into the environment are numerous and diverse. Crop protection represents by far the most important use of pesticides, and thus agricultural sources significantly outnumber domestic and industrial sources. In most cases, pesticides can be categorized as non-point or distributed source contaminants, capable of causing low level degradation of groundwater quality over large areas (Priddle et al., 1989; MacRitchie et al., 1994; Kolpin et al., 1997). However, point source

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contamination of groundwater can be a serious problem where pesticides are manufactured, stored, or discarded and at sites where pesticides are mixed and application equipment is loaded or rinsed (Fetter, 1993). Point source pollution can also occur when pesticides are applied locally in excessive amounts such as when aerial spraying.

In urban areas (Figure 38), pesticides are used domestically in and around the house and garden. They are also used on golf courses to maintain greens and fairways and by municipalities to control or eliminate weeds along roads and pathways and to help maintain parklands in pristine condition. While the amount of pesticide used in urban areas pales in comparison to agricultural applications, its use is a serious concern when the chemicals are inadequately stored or disposed of or used indiscriminately and in excess of recommended application rates. In practice, the greatest threat of pesticides in urban areas comes from the direct ingestion of the chemical, perhaps by inhalation of chemical dust or vapor, by dermal contact with contaminated soils, or by consumption of garden produce containing pesticide residues.

Figure 38 - a) Roundup, a pesticide containing glyphosate, is popularly used in urban areas but is raising serious concerns over cancer risk to humans. b) 2,748 locations in New York City, USA, where Roundup was used in 2016. More information is available at InHabitat website↗ (photography by Ken Howard).

Until recently, the contamination of groundwater in urban areas by pesticides has not been regarded as a potential problem. In fact, the vast majority of pesticide studies have ignored wells in urban areas, preferring to focus on agricultural sources of pesticide and impacts on rural well water. It is now recognized that a significant potential for the pollution of urban groundwater exists in the vicinity of urban parkland, golf courses,

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leaking sewer pipes, and landfills, and in areas where large amounts of insecticide or weed control products are used domestically. Serious problems can also occur where urban areas have encroached on land where earlier-generation, persistent pesticides were once used. In Australia, for example, sites used for dipping sheep can be heavily contaminated with potentially leachable arsenic and organochlorine compounds (Knight, 1993). In some areas, as in Canberra, these sites now lie beneath urban developments

Ironically, many of the earliest pesticides developed (organochlorines), which are now banned or heavily restricted, display the least tendency to contaminate groundwater. The majority are poorly soluble and are readily adsorbed onto organic matter in the soils and sediments (Hassan, 1982; Langmuir, 1997; Tindall et al., 1999). They are effectively immobile under conditions of intergranular groundwater flow but have been reported in groundwater where fracture flow is the dominant transport mechanism. In Perth, Western Australia, Gerritse and others (1990) reported negligible contamination of the Bassendean sands aquifer despite the extensive treatment of urban soils for the control of termites with organochlorine pesticides (50 kg ha^{-1} for a housing density of 10 ha^{-1} over a period of ten years). Vertical transport velocities for lindane dieldrin, heptachlor, and chlordane were estimated to be 2.1, 0.2, 0.1 and 0.1 cm/a, respectively. The organophosphate chlorpyrifos—an insecticide—was also monitored during the study. Levels were found to be low, and no correlation with urbanization was observed. However, Appleyard (1993, 1995, 1996) found various incidents of pesticide contamination of groundwaters in Perth, allegedly associated with its manufacturing, storage, and disposal in wastewater.

Some of the more comprehensive pesticide studies have been carried out in the USA where pesticide usage has always been extremely high and groundwater provides drinking water for over 50 percent of the population (Moody et al., 1988; Barbash et al., 1996). Between April 1988 and February 1990, samples collected from 566 community water well systems and 783 rural domestic wells were analyzed for 101 pesticides and 25 pesticide degradants (USEPA, 1992). Approximately 10 percent of the CWS (Community Water System) wells and 4 percent of rural domestic wells were found to contain pesticides or pesticide residues above the minimum reporting limits used in the survey. The most detected pesticides were

- DCPA acid metabolites or Dimethyl tetrachloroterephthalate (until 1998, when its manufacture was discontinued, DCPA or Dacthal was used extensively on home lawns, golf courses, and farms to control annual grasses and broadleaf weeds; it was re-introduced in 2001) and
- atrazine, an herbicide used almost exclusively on agricultural land.

Six pesticides including atrazine, lindane, alachlor, and EDB (Ethylene Dibromide) were found in rural domestic wells at concentrations in excess of USEPA maximum contaminant levels.

In 1993, the USGS initiated the United States National Water Quality Assessment, a survey of groundwater quality in 20 of the country's most important hydrologic basins. The

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study focused specifically on the quality of shallow, recently recharged groundwater (generally younger than ten years) and tested for the presence of 46 pesticide compounds (25 herbicides, 17 insecticides, two herbicide transformation products, and two insecticide degradants) in water from 1,012 wells and 22 springs (Kolpin et al., 1998).

A particular feature of this study was the inclusion of 221 urban well site samples among the 1,034 samples collected. Significantly, 46.6 percent of the urban samples showed evidence of pesticide contamination, a value only slightly less than the 56.4 percent recorded in rural samples. Of the 46 pesticide compounds examined, 39 were detected; atrazine, simazine, and prometon were the most observed. Only atrazine exceeded its Maximum Contaminant Level (MCL) of 0.003 mg/L and this occurred in just one well. In the urban samples, prometon—a triazine herbicide popularly used as an asphalt additive and a treatment for driveways, fence lines, lawns, and gardens—was especially evident. In addition, the urban samples tended to show a relatively high incidence of contamination by insecticides.

Currently, no clear worldwide picture has been drawn of the true nature and extent of pesticide contamination in groundwater, especially in urban areas. Most countries do not include the common pesticides in routine analyses of groundwater. In addition, there is no consensus on the health hazards posed by pesticides and what constitutes a safe level in drinking water. It was once believed that the vast majority of pesticides released into the subsurface would become immobilized in the upper levels of the soil and/or degrade before reaching the water table. This is not the case. Where detailed studies have been undertaken, notably in the USA, they often reveal widespread degradation of groundwater, albeit at very low concentrations. The studies also show that the threat of pesticide contamination is not confined to rural areas and that urban groundwater is at comparable risk.

4.2.8 Landfills

Most urban areas generate high volumes of solid waste that potentially create a substantial source of groundwater contamination. For low-income and high-income cities, respectively, solid waste generation rates are estimated to range between 0.3 and 0.6 kg/person/day and 0.7 to 1.8 kg/person/day. The waste can normally be assigned to one of four categories: domestic, commercial, industrial, and construction/demolition (Table 14), depending on its primary source. However, most existing landfills contain all four categories of waste in varying proportions.

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Table 14 - Characteristics of domestic, commercial, industrial and construction/demolition waste shown in percentages (modified after Zhang et al., 2003).

Waste composition	High-income countries		Middle-income countries	Low-income countries		
	US	UK	Singapore	Delhi, India	Kathmandu, Nepal	Wuhan, China
Vegetable	22	25	5	47	67	16
Paper	34	29	43	6.3	6.5	2.1
Metals	13	8	3	1.2	4.9	0.55
Glass	9	10	1	0.6	1.3	0.61
Textiles	4	3	9	NA	6.5	0.62
Plastic/leather/rubber	10	7	6	0.9	0.3	0.5
Wood	4	NA	NA	NA	2.7	1.76
Dust/ash/other material	4	18	32	36	10	77
Moisture content (%)	22	20–30	40	15–40	NA	30

Table 15 compares typical solid waste compositions for high-, low-, and middle-income countries. In general, waste from high-income countries is composed of relatively large amounts of paper, metals, glass, and plastics and relatively few putrescibles, while waste from low-income countries comprises a fairly large proportion of putrescibles, dust. and ash but relatively little paper, metals, glass, and plastic (Table 15). In New Delhi, India, 47 percent of urban waste is vegetable matter—almost double the proportion observed in the United Kingdom.

Table 15 - Comparison of waste composition from different countries (modified after Zhang et al., 2003, and Palmer Development Group, 1995).

Waste type	Waste source	Examples of components in waste areas
Domestic	Single-family dwellings; multi-family dwellings; low, medium, and high-rise apartments	Food waste, paper, cans, cardboard, plastics, textiles, rubber, leather, garden trimmings, wood, glass, non-ferrous metal, ferrous metal, dirt, ash, brick, bone
Commercial	Shops, restaurants, markets, office buildings, hotels, institutions	Food waste, paper, cans, cardboard, plastics, textiles, rubber, leather, garden trimmings, wood, glass, non-ferrous metal, ferrous metal, dirt, ash, brick, bone
Industrial	Light and heavy manufacturing, refineries, chemical plants, mining, power generation	Industrial sludge, dye stuff, paint, empty containers, tannery waste, tars, waxes, cellulose waste
Construction/ demolition	Urban and industrial construction and demolition sites	Soil, stones, concrete, bricks, plaster, timber, paper, plastic, piping, electrical components

Historically, much of the solid waste generated in urban areas was collected and dumped in landfills that comprised no more than a large land depression or a pit/quarry used previously to extract aggregate for construction purposes. As an example, Figure 39 shows the locations (in the early 1990s) of 1,183 open (active) and closed landfill sites in southern Ontario, Canada, superimposed on a surficial geology map. The greatest number

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(380) are in areas of sand and gravel. For the vast majority, no consideration would have been given to the need to protect underlying aquifers from the waste.

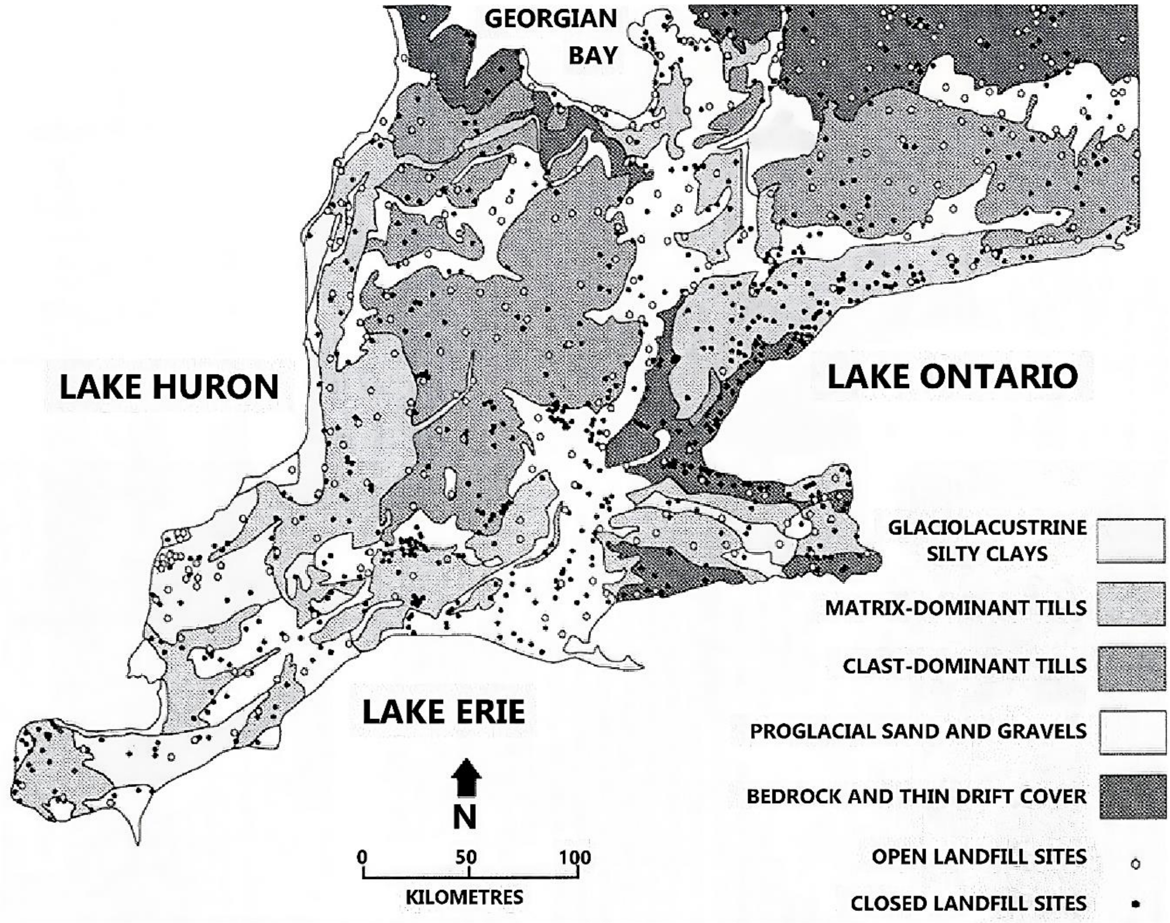

Figure 39 - Location of open (n=311) and closed (n=868) landfill sites in southern Ontario, Canada, in relation to the surficial geology (after Eyles & Boyce, 1997).

Today, modern cities in many countries strive to reduce the waste entering landfills through reuse, recycling, and composting programs. They will also separate types of waste going to each landfill and design or retrofit landfills in such a way as to reduce or eliminate environmental impacts. However, older sites remain problematic especially given the increasing demand for landfill disposal. For example, Figure 40 shows the Jiangcungou Landfill Site that was built in 1994 at high elevation in a natural valley to service Xi'an, China. Only partially engineered, unstable, and offering minimal environmental protection, the site closed in 2019 after being filled to capacity 25 years ahead of schedule and now causes concern about potential groundwater contamination.

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a)

b)

Figure 40 - a) The Jiangcungou landfill in Xi'an, China, lies at high elevation and fills a deep, narrow valley on the outskirts of the city. The landfill is partially engineered and covered b) to reduce leachate generation; however, serious concerns have been expressed over the potential for subsurface contaminant plume migration. The landfill is now closed (photography by Ken Howard).

Despite indications that groundwater can be protected with landfills that are fully engineered, continually managed, and vigilantly maintained, the grim reality is that the vast majority of urban waste currently residing in landfills throughout the world remains a serious threat to groundwater quality. Moreover, most of the world's urban landfills have not benefited from waste sorting and diversion programs, meaning that—in addition to relatively inert domestic waste—most contain the full spectrum of materials found in urban

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areas including oils, phenols, hydrocarbon solvents, cyanide wastes from metallurgical operations, pulp manufacturing wastes, mercury-rich materials from the electrical industry, solid residues from the petrochemical industries (such as polychlorinated biphenyls, pesticide or herbicide residues), and phenol-rich tar wastes. Based on an audit of contaminants stored and utilized in the Greater Toronto Area, Canada, Howard and Livingstone (1997, 2000) ranked unlined landfills as the greatest threat to local groundwater quality.

From a groundwater perspective, the primary concern associated with landfills is the production of leachate, a contaminant "soup" that can migrate from the site and pollute aquifers. Leachate is produced when moisture enters the landfill and gradually dissolves the waste materials. The quantity and chemical quality of the leachate will be determined by numerous physical and chemical factors; nevertheless, analyses of leachate from municipal sites all over North America often show similarities that must reflect the common refuse types in municipal waste (Barker et al., 1989).

Table 16 shows the typical range of constituents found in leachate obtained from municipal landfills in Ontario, Canada. The data were obtained during the 1990s and predate the introduction of programs that not only minimize the disposal of domestic hazardous waste (e.g., oils, paints, electronics, and food waste) but also divert large amounts of essentially inert material such as paper, wood, and plastics. Although published in 1997, the data provided in the table remain relevant; most groundwater contamination in urban areas, whatever its source, represents a legacy of past practices. However, there is an expectation that engineered landfills of the modern era will generate leachate that is significantly different in composition than in the past. While there is no “typical” leachate, there is evidence that many of the chemical constituents listed in Table 16 occur within relatively consistent ranges of concentration. Inorganic parameters are normally dominant and typically range up to 50,000 mg/L. The major inorganic contaminants are chloride, calcium, magnesium, sodium, potassium, iron, bicarbonate, sulfate, ammonia, nitrite/nitrate, and various forms of phosphorous; other inorganics often include nickel manganese, zinc, copper, lead, chromium, arsenic, cyanide, mercury, strontium, molybdenum, aluminum, boron, and fluoride.

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Table 16 - Organic and inorganic constituents of leachate in landfills from Ontario, Canada (from Howard & Livingstone, 1997).

Compound/Ion	Range (mg/L)	Mean (mg/L)	Sample Size
Benzene	< 0.0001–0.149	0.0065	18
Chlorobenzene	< 0.0002–0.046	0.00044	12
Ethylbenzene	< 0.0002–0.404	0.005	14
2-butanone (MEK)	< 0.0067–2.7	0.218	6
Butyric acid	14–229.2	56.7	2
1, 1, 1 -trichloroethane	< 0.0002–1.388	0.0005	14
1, 1, 2, 2 - tetrachloroethane	< 0.0003	< 0.0003	14
1, 1 -dichloroethane	< .0002–0.1205	0.0013	13
1, 2 -dichloroethane	0.0002–2.976	0.0014	14
Ether, extractable	9–160	22.5	28
1, 1 -dichloroethene	< 0.005–4.216	0.0017	16
Tetrachloroethene	< 0.0005–0.36	0.004	16
Trichloroethene	< 0.0002–0.94	0.0017	16
Cresol	< 0.01–0.168	0.026	3
Dichloromethane	< 0.0004–0.39	0.0019	15
Phenol	0.016–472	1.292	119
Diethyl phthalate	< 0.002–0.021	0.006	3
Dimethyl phthalate	< 0.002–< 0.6	0.037	6
1, 2 -dichlorobenzene	< 0.0002–< 0.002	0.0003	11
1, 3 -dichlorobenzene	< 0.0002–0.0084	0.0004	11
1, 4 -dichlorobenzene	< 0.0002–0.024	0.0019	12
Styrene	< .0015–0.0809	0.0081	5
Toluene	< 0.0005–0.95	0.009	15
Trichlorofluoromethane	< 0.001–< 0.085	0.0027	14
Trihalomethanes (THM)	< 0.0002–0.16	0.00075	12
Xylene (total)	< 0.0002–5.245	0.0041	22
Aluminum	ND - 14	0.482	42
Chloride	0.4–12,000	269.8	159
Calcium	2.8–4,580	579.8	113
Magnesium	0.14–1,750	265.8	104
Sodium	13.8–16,000	935.7	118
Potassium	1.5–1,680	259.1	74
Iron	0.01–1,300	58.05	131
Bicarbonate	2,050–18,910	8367	8
Ammonia -N	7.6–1,820	174.5	95
Total Kjeldahi Nitrogen (TKN)	3.5–2,550	154	92
Nitrate -N	0.004–137	2.49	76
Nitrite -N	< 0.001–66	0.077	47
Zinc	ND - 16	1.416	92
Copper	0.007–7	0.045	89
Lead	0.001–2.1	0.068	95
Chromium	0.005–2.5	0.09	101
Sulfate	< 1–4,200	139.4	93
Cyanide	< 0.001–1.2	0.136	36
Mercury	< 0.00003–0.002	0.00016	19
Molybdenum	< 0.005–2	0.057	37
Nickel	0.025–2.9	0.279	84
Boron	0.8–52	10.43	41
Arsenic	< 0.001–0.32	0.0062	44
Manganese	0.03–793	3.54	105

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Leachate may also contain significant concentrations of organic acids and synthetic organic compounds such as components of petroleum, paints, household chemicals, solvents, cleaners, glues, inks, and pesticides. Characteristic organics usually include volatile organic hydrocarbons and halocarbons such as benzene, ethylbenzene, toluene, xylenes, dichloromethane, tetrachloroethylene, trichloroethylene, phenolics, cresol, phthalates, chlorobenzenes, and other substituted benzenes. Fetter (2001) noted that analyses of total organic carbon in leachate from municipal solid-waste landfills in Wisconsin, USA, typically ranged between 427 and 5,890 mg/L (expressed as a range of site medians).

Contamination of groundwater due to leakage from landfills has been reported throughout the world (Borden & Yanoschak, 1990; Kerndoff et al., 1992; Barber et al., 1992; Kjeldesen, 1993; Howard et al., 1996; Grischek et al., 1996). In fact, Borden and Yanoschak (1990) examined data from landfills in North Carolina, USA, and found that 71 percent had caused groundwater pollution in one or more monitoring wells. In most cases, landfill leakage tends to behave as a point source contaminant release, with the outcome being severe degradation of groundwater quality in close proximity to the site with natural attenuation processes lessening the severity of the impact at a regional scale.

4.2.9 Road De-icing Chemicals

Each year, throughout snow-belt regions of the world, road de-icing chemicals (Figure 41) are applied extensively to urban roads and highways to maintain safe driving conditions (Figure 42).

Figure 41 - Pyramids of NaCl road salt seen at the Cargill salt lot, as viewed from Cherry Street bridge, Toronto, Canada July 4, 2014 (photography by Ken Howard).

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Figure 42 - In snow belt regions of the world, application of road de-icing salts is the only cost-effective means of maintaining the flow of traffic on roads and highways (photography by Ken Howard).

Road de-icing chemicals, most notably sodium chloride (NaCl) and calcium chloride ($CaCl_2$) are readily dissolved by rain and meltwaters then enter rivers, lakes, and shallow groundwater. In groundwater systems, they degrade water quality either directly or by mobilizing trace elements such as cadmium, copper, lead, and zinc through ion exchange, lowered pH, and chloride complex formation (Bäckström et al., 2004). Aquatic organisms in small ponds and low-flow streams are particularly susceptible to toxicity due to road salt application (Marsalek, 2003; Marsalek et al., 2000). Road de-icing salts do not degrade biologically, and chemical retardation tends to be minimal.

In terms of drinking-water quality, the guideline for chloride is normally set at 200 or 250 mg/L in recognition of a taste threshold that, for most people, exists between 200 and 300 mg/L. The guideline for sodium is more contentious, since the ion has been strongly linked with the development of hypertension, a condition affecting perhaps 20 percent of people in the USA (Moses, 1980; Craun, 1984; Tuthill & Calabrese, 1979). Raised sodium intake has also been associated indirectly with hypernatremia (an electrolyte problem where serum sodium concentrate rises above 145 mmol/L; WHO, 1984). While most regulatory authorities retain a guideline of 200 mg/L for purely aesthetic purposes, a limit of 20 mg/L is generally recommended for individuals following a low-sodium diet.

Beginning in 1995, de-icing salts used on Canadian roads were subjected to a comprehensive five-year scientific assessment under the Canadian Environmental Protection Act, 1999 (Environment Canada and Health Canada, 2001). The final report concluded that high releases of road salts were having an adverse effect on freshwater ecosystems, soil, vegetation, and wildlife, which prompted the development of a Code of Practice for the Environmental Management of Road Salts (Environment Canada, 2004) designed to optimize salt application and reduce chloride transfer to the environment.

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Road de-icing chemicals take several different forms (Nystén & Suokko, 1998) and are applied using a wide variety of methods as shown in Table 17. However, with rare exceptions, sodium chloride (NaCl) and calcium chloride ($NaCl_2$) are the chemicals of choice. Sodium chloride is the cheapest and most used. It is particularly cost-effective at temperatures above ≈15 °C. Calcium chloride is much more effective for temperatures in the range ≈12 °C to ≈34 °C but is less frequently used because it is significantly more expensive and has been known to make the road surface slippery when wet (Hanley, 1979).

Several studies suggest that the salts used for de-icing purposes are unusually pure and normally contain very few compounds of special environmental concern (Howard & Beck, 1993). In many regions, sodium ferrocyanide ($Na_4Fe(CN)_6\ 10H_2O$) is commonly added to road salt as a de-caking agent; however, while this can release free cyanide upon exposure to light, the cyanide has very limited mobility in the subsurface, being readily adsorbed onto soil particles during overland flow (Olson & Ohno, 1989).

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Table 17 - Physical-chemical properties of road salts[1].

Substance	CAS No.	Specific application	Molecular weight	Eutectic[2] temperature (°C)	Working[3] temperature (°C)	Water solubility (g/100 mL)(°C)
Sodium chloride, $NaCl$	7647-14-5	Road de-icer and anti-icer, de-icing additive for sand	58.44	-21	0 to -15	35.7 (0) 39.12 (100)
Calcium chloride, $CaCl_2$	10043-52-4	Road de-icer, de-icing additive, anti-icer, pre-wetter, dust suppressant, road construction	110.99	-51.1	< 23	37.1 (0) 42.5 (20)
Mixture of sodium/calcium chloride (80/20 mix)		Road de-icer, road anti-icer		N/A	-12	
Magnesium chloride, $MgCl_2$	7786-30-3	Road de-icer, de-icing additive, road anti-icer, dust suppressant	95.21	-33.3	-15	54.25 (20) 72.7 (100)
Mixture of sodium/magnesium chloride (80/20 mix)		Road de-icer		N/A	< -15	
Potassium chloride, KCl	7447-40-7	Alternative road de-icer	74.55	-10.5	-3.89	56.7 (100)
Sodium ferrocyanide, $Na_4Fe(CN)_6\ 10H_2O$	13601-19-9	Alternative road de-icer	484.07			31.85 (20) 156.5 (98)
Ferric ferrocyanide, $Fe_4[Fe(CN)_6]_3$	14038-43-8	Anti-caking additive	859.25			insoluble

[1] Physical-chemical properties are from Weast and others (1989) and Chang and others (1994).

[2] Eutectic temperature refers to the lowest temperature at which the substance will melt ice.

[3] Working temperature refers to the lowest effective de-icing temperature.

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De-icing chemicals are usually applied directly to the road surface in a pure chemical form, but they can also be applied as a solution or used in conjunction with abrasives such as sand. Salt application rates typically range up to 30 g/m^2, which, over the course of the winter, can readily translate to between 10 and 20 tonnes of salt per two-lane kilometer. Annual totals of salt application are given for several European countries in Table 18. For comparison, road salt use in Canada is shown in Table 19.

Table 18 - Use of salt ($NaCl$ and $CaCl_2$) for various European countries during the winter season 1986–87 (after OECD, 1989).

Country	Length of road and highway network (km)	Salt used (10^3 tonnes)
Belgium (1985)	14,058	200
Denmark	7,160	122
Germany (1986)	39,722	627
Italy (1986)	300,922	240
Switzerland	70,000	129
Sweden	30,000	141
United Kingdom	347,000	2,085

Table 19 - Total loading of sodium chloride road salt in Canada during winter 1997 and 1998 (from Morin & Perchanok, 2000).

Province or Territory	Loading of sodium chloride road salt (tonnes)				
	Provincial/ Territorial	Estimated country + municipal	Total provincial + country + municipal	Estimated commercial + industrial[1]	Total
British Columbia	83,458	48,199	131,657	9,874	141,531
Alberta	101,063	67,870	168,933	12,670	181,603
Saskatchewan	44,001	4,844	48,845	3,663	52,508
Manitoba	36,780	28,256	65,036	4,878	69,914
Ontario	592,932	1,123,653	1,716,585	128,744	1,845,329
Quebec	609,550	827,205	1,436,755	107,757	1,544,512
New Brunswick	189,093	75,826	264,919	19,869	284,788
Prince Edward Island	23,051	4,300	27,351	2,051	29,402
Nova Scotia	270,105	77,761	347,866	26,090	373,956
Newfoundland	159,200	47,558	206,758	15,507	222,265
Yukon	1,791	120	1,911	143	2,054
Northwest Territories & Nunavut	1,846	0	1,846	138	1,984
Total	2,112,870	2,302,592	4,418,462	331,385	4,749,847

[1] Commercial and industrial road salt use is assumed to be at 7.5 percent of provincial and municipal use (7.5 percent based on estimate by Canadian Centre for Occupational Health and Safety (CCOHS) Cheminfo, 1999).

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The most common practice is to apply pure sodium chloride to highways and primary urban roads, while mixtures of sand and between 5 and 95 percent sodium chloride are preferred on side streets. Mixtures of sand and 5 percent sodium chloride are also used on gravel roads since pure sodium chloride tends to promote the creation of potholes in the road surface during a thaw. Some jurisdictions use a mixture of sand and calcium chloride with considerable success, finding that while 50 percent less salt is required, melting temperatures below minus 20° C can be readily maintained (Fabricius & Whyte, 1980).

Until the early 1990s, it was commonly believed that most salt applied to roads and highways in urban areas was readily flushed from urban catchments each year via overland flow, with negligible release to the subsurface and only minor environmental damage. This assumption was brought into question when Pilon and Howard (1987) reported chloride concentrations as high as 13,000 mg/L in shallow subsurface waters adjacent to a highway in southern Ontario, Canada, and a survey of springs issuing from shallow sites along the shoreline of Lake Ontario by Eyles and Howard (1988) revealed chloride concentrations approaching 400 mg/L. These findings prompted the development of a detailed salt balance for a 104 km^2 urban catchment in eastern Toronto, as shown in Figure 43 (Howard et al., 1993; Howard & Haynes, 1993; 1997).

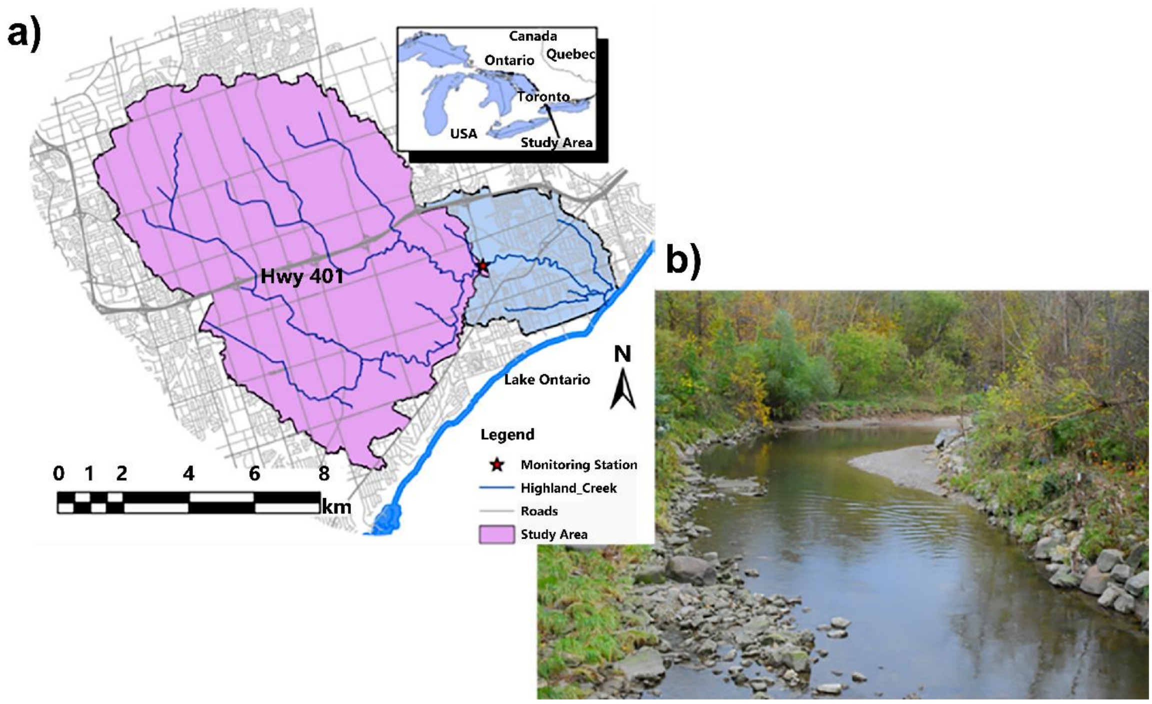

Figure 43 - Highland Creek catchment in eastern Toronto, Canada, showing the location of the gauging station where stream water quality and flow was observed (a) from Perera et al, 2013; b) photography by Ken Howard).

The salt balance for the Highland Creek catchment demonstrated that over 50 percent of the salt applied each year enters the subsurface, causing widespread contamination of the groundwater. Moreover, this salt would be released as baseflow to

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urban streams very gradually in future decades, causing a steady decline in stream water quality. The study showed if existing rates of salt application continued, sodium and chloride concentrations in baseflow groundwater from the region would increase by a factor of 3 until mean steady state concentrations in excess of 250 and 400 mg/L, respectively, were approached. These values represented a threefold increase over baseflow concentrations observed at the time. Follow-up studies by Perera and others (2013) have largely confirmed these findings. Using a more refined analysis, they suggest that if current salt application rates continue, chloride concentrations in late summer baseflow will exceed 400 mg/L by 2030 and continue to rise until a steady state concentration of just over 500 mg/L is reached (Figure 44).

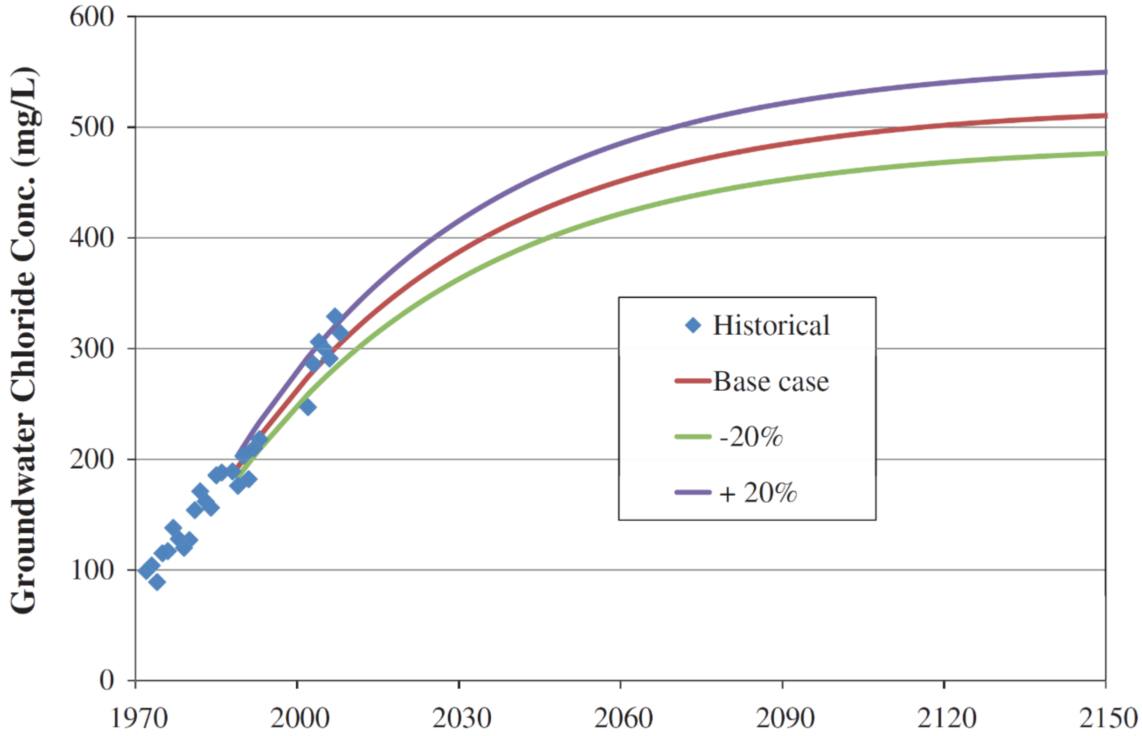

Figure 44 - Projected chloride concentrations in Highland Creek based on existing salt application rates (the base case) and scenarios involving a ±20 percent change in application rate (after Perera et al., 2013).

The impacts of road de-icing chemicals on groundwater have also been documented by numerous other researchers such as Huling and Hollocker (1972), Diment and others (1973), Field and others (1974), and Locat and Gélinas (1989). For example, in Burlington, Massachusetts, USA, the use of sodium and calcium chloride on state highways has been held responsible for chloride concentrations in local groundwaters exceeding the local drinking water standard of 250 mg/L (Toler and Pollock, 1974). Similar degrees of water quality deterioration were reported in the states of Illinois (Wulkowicz & Saleem, 1974), and Wisconsin (Eisen & Anderson, 1979). In the eastern United States, concern for the impact of road de-icing chemicals on water quality and vegetation have led to the ban of these chemicals on sections of highway considered environmentally sensitive.

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As a final point, it needs to be emphasized that, when considering an urban aquifer as a whole, the water quality at natural discharge points—that is, in the baseflow entering an urban stream at the end of a groundwater flow line—entrained contaminants are furthest from their source and at their most dilute. This is demonstrated by Figure 45, which shows simulations of the predicted fate of salt applied to three sections of roadway as described in the remainder of this section.

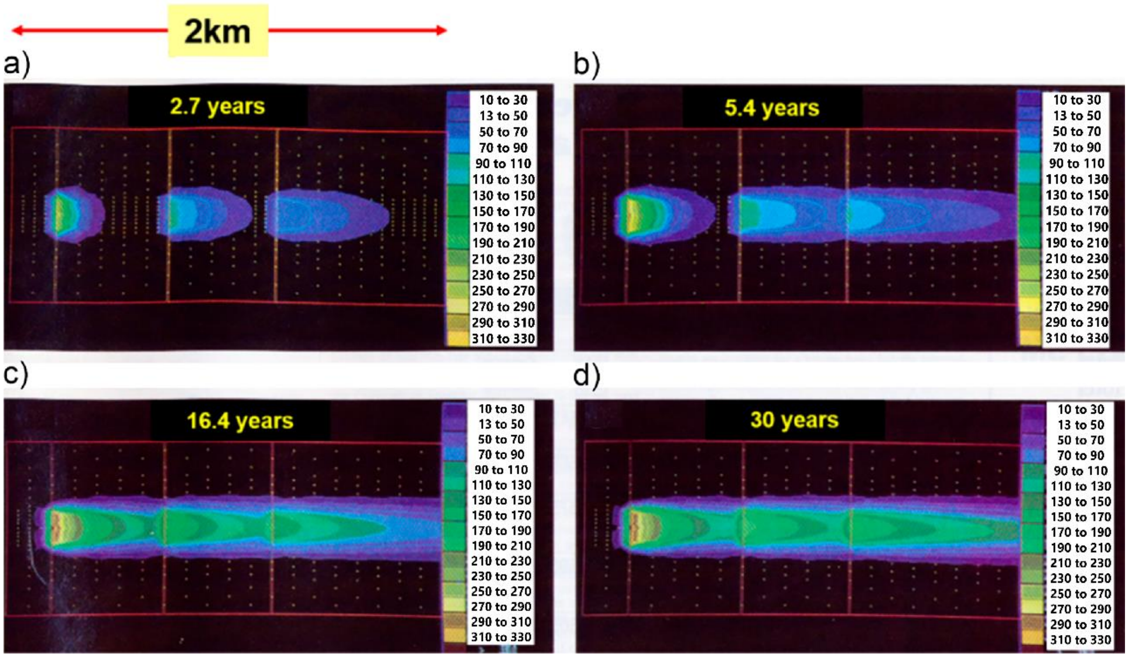

Figure 45 - Model simulations demonstrating that concentrations of chloride in baseflow to streams can be an order of magnitude lower than concentrations in the aquifer, especially when compared to groundwater closely adjacent to the roadways. The model domain is 2 km by 1 km and includes three 10 m wide roadways (north-south red lines). These are 500 m apart and oriented perpendicular to the direction of flow, which is toward the right. The river that runs parallel to the roadways is represented by a constant head boundary along the right-hand edge of the domain. The left-hand edge is a groundwater divide, represented by a no-flow boundary. The aquifer is 5 m thick and is recharged everywhere at a rate of 160 mm/a. The hydraulic conductivity averages 1.2×10^{-4} m/s, and the effective porosity is approximately 30 percent (Howard et al, 1993).

For simulation purposes, salt is applied to a 220 m section of each highway. This gives rise to three plumes that coalesce over time. The concentrations shown in Figure 45 are average values over the depth of the aquifer. Highest chloride concentrations are always present in the plume nearest the ground-water divide on the left, where Darcy velocities are low and little water is available to dilute the infiltrating salt. At earlier times, plume concentrations are relatively low, but they increase with time as more salt is added to the system each winter.

The average concentration of chloride in the baseflow entering the stream increases with time until, a steady condition is reached at which time the annual salt input is balanced by annual salt output (Figure 45d). For this relatively small model domain, steady conditions occur after just 30 years. In large catchments, chemical steady state may not be reached for several hundred years (Howard & Maier, 2007). When steady state is reached,

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concentrations of chloride in the baseflow entering streams are at their most dilute relative to the source concentration and can be one or two orders of magnitude lower than concentrations in the aquifer. Concentrations of chloride tend to be especially high within 200 m of salted roadways.

4.2.10 Urban Stormwater

In many cities, surface runoff (urban stormwater) is a significant source of contaminants and causes a variety of concerns for receiving waters. Issues commonly reported are mostly related to physical habitat changes, water quality changes, public health concerns, and aesthetic degradation (Ellis & Hvitved-Jacobsen, 1996; Marsalek et al.,1997a). Table 20 shows the range of hazardous and toxic substances found in urban stormwater.

Table 20 - Hazardous and toxic substances found in selected urban stormwater runoff, (modified after Zhang et al., 2003; USEPA, 1983, 1995; Pitt & McLean, 1986; Ellis, 1997).

Substances found	Residential areas	Industrial areas
Halogenated aliphatics		X
Phthalates	X	X
PAHs		X
BTEX compounds	X	X
Metals		
Aluminum	X	X
Chromium		X
Copper	X	X
Lead	X	X
Zinc	X	X
Nickel	X	X
Pesticides and phenols	X	X

Significantly, the problem is not limited to industrial areas; residential areas can also supply stormwater with contaminants of concern, notably pesticides such as dieldrin, chlordane, endrin, endosulfan, and isophorone. In most cases, urban runoff migrates rapidly to surface water features such as streams, rivers, ponds, and lakes where it poses a serious threat to aquatic life (e.g., due to thermal enhancement and toxic effects) and potentially endangers human health by polluting recreational waters or contaminating sources of drinking water (Lijklema et al., 1993; Makepeace et al., 1995).

Unfortunately, runoff may also enter the subsurface, either naturally as indirect recharge or via engineered structures such as soakaway pits, infiltration basins, and detention ponds where it can seriously impact groundwater quality. This can be a particular concern in areas where the artificial recharge of groundwater using stormwater (also known as MAR: managed aquifer recharge) has become an attractive and popular means of augmenting groundwater supplies. The potential impacts of urban stormwater on groundwater quality are discussed in Section 4.2.11.

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4.2.11 Nature of Pollutants in Urban Stormwater

The quality of urban runoff can be extremely variable. It depends on rainfall intensity, rainfall amounts, and the nature of the materials the runoff encounters during transport. Timing of sampling runoff relative to the storm duration is important, because the majority of contaminants (including sediment) that have accumulated between storm events and are prone to being mobilized (e.g., road salt) tend to be entrained in runoff that occurs during the period immediately after the start of a storm. This mobilizing event is referred to as the first flush where, by definition, incremental contaminant load exceeds incremental flow (Novotny, 1995; Duncan,1995).

The first flush phenomenon is common in urban areas (Duncan, 1995) due to the large proportion of impervious surface, which creates high runoff coefficients. Pollutants carried in a first flush can create shock loadings to receiving surface waters (McIntosh & Pugh, 1991) and mitigation measures may be required to reduce the problem. The first flush is less of a problem in large catchments, where first flush events are spatially and temporally distributed and mixing of water sourced from various locations in the catchment readily reduces peak contaminant concentrations. It is also less problematic for receiving groundwaters, as potential impacts are damped by various combinations of pre-release storage and slow dispersive transport in the subsurface.

In general, urban stormwater contains pollutants from automobile traffic, litter, domestic refuse, dust fall and spills, lawn chemicals, animal fecal matter, and—in the snow-belt regions of the world—large quantities of road de-icing chemicals (Marsalek & Ng, 1987; Marsalek, 1990). Urban stormwater is particularly well known for its high sediment load, which is in part attributable to the hydraulic efficiency of urban channels and the highly erosive character of fast-flowing water.

Stormwater detention or retention ponds can be effective in removing suspended sediments (Figure 46); the introduction of grass swales and wetlands (Figure 47) can help reduce dissolved loads, either by plant uptake or by chemical transformation and adsorption. These "natural" approaches to treatment are very successful at removing suspended sediment and can virtually eliminate many organics, heavy metals, and nutrients. However, they are largely ineffective when it comes to treating the more mobile contaminants such as chloride. Moreover, such facilities require regular maintenance and, in the case of wetlands, may simply bio-accumulate contaminants—especially metals—that will subsequently threaten the wildlife they inevitably attract.

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Figure 46 - Wet stormwater detention pond: an earthen pond with a permanent pool and temporary storage for attenuating stormwater flows (photography by the Georgia Department of Transportation, USA).

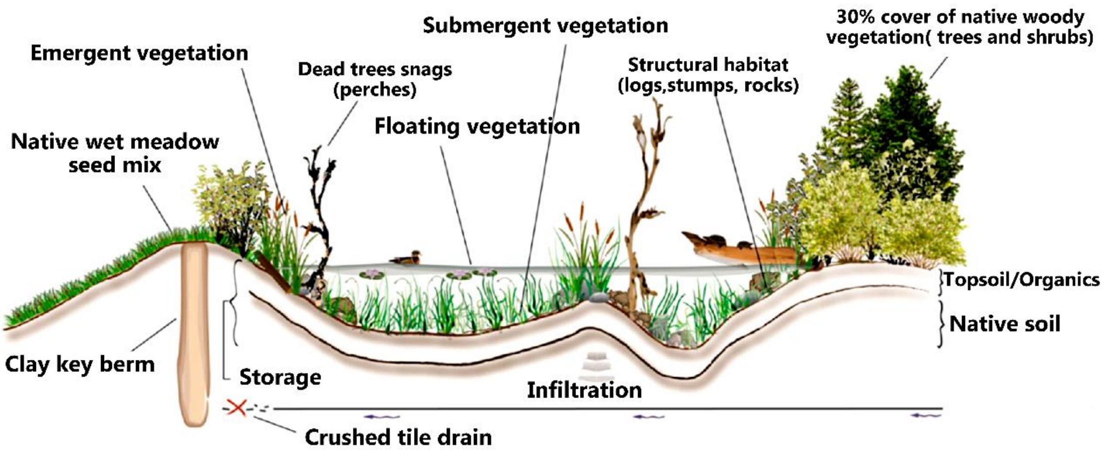

Figure 47 - Design example for a constructed wetland (diagram credit: Toronto & Region Conservation Authority).

In any urban area, the best quality surface runoff is normally associated with roof drainage (for example, Table 21). The quality of roof runoff will not differ significantly from that of local rainfall, although organic matter entrained in the runoff may cause soakaway pits to become chemically reducing. Also, some roof runoff (especially from tin, copper, and lead roofs) can generate significant heavy metal concentrations, sometimes in the hundreds of micrograms per liter range (Harris, 2007). While roof runoff can be an excellent source of irrigation water in urban areas, it is not suited for potable use since it is easily contaminated by fecal matter produced by birds and other animals that frequent rooftops.

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Table 21 - Water quality in urban runoff from various sources in Denmark (from Mikkelsen et al., 1994).

Pollutant	Urban runoff	Highway runoff	Roof runoff	Atmospheric deposition (%)
Suspended Solids (mg/L)	30–100	30–60	5–50	
COD (mg/L)	40–60	25–60	Less	19
Total N (mg/L)	2	1–2	Less	70
Total P (mg/L)	0.5	0.2–0.5	Less	23
Pb (µg/L)	50–150	50–125	10–100	40
Zn (µg/L)	300–500	125–400	100–1000	30
Cd (µg/L)	0.5–3			
Cu (µg/L)	5–40	5–25	10–100	7

By comparison, the most common concern in urban areas relates to runoff from roads and highways (Kobringer et al., 1982; M.M. Dillon, 1990). Highway runoff contains various classes of pollutants (Thompson et al. 1995) that include

- solids (including fine particulates),
- nutrients (e.g., nitrate and ammonia),
- de-icing agents (mostly sodium chloride),
- heavy metals, and
- petroleum hydrocarbons (including polycyclic aromatic hydrocarbons (PAHs).

Two of these groups (heavy metals and PAHs) are a particular concern due to their toxic impacts on receiving waters (Lijklema et al., 1993). The heavy metals in highway runoff typically include lead, zinc, iron, copper, cadmium, chromium, nickel, and manganese (Kobringer et al., 1982) and are all related to automobile operation, abrasion and corrosion of highway structures, or road maintenance activities (Marsalek et al., 1997b; M.M. Dillon, 1990; Sansalone & Buchberger, 1997). The most common source of PAHs is leaking automobile crankcases, with tailpipe emissions being a secondary contributor (Hunter et al. 1979). Unlike heavy metals that tend to be found in the aqueous phase, PAHs tend to concentrate in soils and sediments such that only a small fraction of their total load can be considered bioavailable (Hoffman et al. 1985; Stotz, 1997).

Prior to 2000, the vast majority of studies related to highway runoff contamination were conducted in Europe and the United States (Frazer, 1990). However, one valuable and notable exception was a highly innovative and comprehensive 16-month study of five heavy metals and 14 PAHs in stormwater runoff from the Queen Elizabeth Way (QEW) highway as it crossed the Skyway Bridge in Burlington, Ontario, Canada (Marsalek et al., 1997b). At the site investigated, the QEW highway was paved with asphalt and comprised four lanes in each direction. The bridge was heavily traveled with annual average daily traffic (ADT) (both ways) averaging 100,000 vehicles. Key results of the study, conducted over a 16-month period, are summarized in Table 22 and Table 23.

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Table 22 - Concentrations (C) of heavy metals in highway bridge runoff and urban runoff in mg/L. Concentrations represent event mean concentrations (EMCs) and are shown as means of the whole data set as well as 10th and 90th percentiles (after Marsalek et al., 1997b).

Metal	$C_{10\%}$	C_{mean}[1]	$C_{90\%}$	US Data[2] (mean)	European Data[3] (mean)	MOEE[4]	MOEE[5]	PWQQ[6]
Zn	0.059	0.337	0.775	0.644	0.507	0.19	0.15	0.030
Pb	0.001	0.072	0.166	0.670	0.286	0.046	0.057	0.005
Ni	0.004	0.069	0.177	0.040	0.041	0.012	0.01	0.025
Cu	0.023	0.136	0.277	0.065	0.114	0.16	0.045	0.005
Cd	0.0001	0.015	0.039	0.015	0.006	0.005	0.00094	0.0002

[1] N=53.

[2] Mean of mean concentrations reported in the US literature. N varied from 8 to 29 for various metals (Frazer, 1990).

[3] Mean of mean concentrations reported in the European literature. N varied from 4 to 12 for various metals (Frazer, 1990).

[4] Ontario MOEE (Ontario Ministry of Environment and Energy, Canada), storm sewer outfalls, Etobicoke and Scarborough (Paul Theil & Associates and Beak Consultants, 1995).

[5] Ontario MOEE, storm sewer outfalls, The City of Toronto (Maunder et al.,1995).

[6] Ontario Provincial Water Quality Objectives (Ontario MOEE, 1996).

Table 23 - Concentration of 14 PAHs in highway bridge runoff and urban runoff in ng/L. Concentrations (C) represent event mean concentrations (EMCs) and are shown as means of the whole data set as well as 10th and 90th percentiles (after Marsalek et al., 1997b).

PAH	$C_{10\%}$	C_{mean}[1]	$C_{90\%}$	MOEE[2]	MOEE[3]
Indene	1.3	16.9	42.1		
2-Methylnaphthalene	3.8	74.3	217.5		112
1-Methylnaphthalene	3.0	39.1	101.2		58
Acenaphthylene	1.5	14.7	35.9	18.0	12
Acenaphthene	1.5	24.8	62.5	96.0	
Fluorene	3.4	56.2	141	108	
Phenanthrene	8.3	397	955	550	
Pyrene	31.8	454	1,020	615	186
Fluoranthene	14.2	504	1,320	782	500
Benzo(b)fluoranthene	3.7	23.9	55.8	553.0	119
Benzo(k)fluoranthene	2.3	135	344	570.0	
Benzo(a)pyrene	5.2	186	481	320.0	48
Indenopyrene	0.3	143	322	274.0	
Benzo(ghi)perylene	0.8	222	546	335.0	194

[1] Skyway Bridge runoff (N = 29).

[2] Ontario MOEE, Storm sewer outfalls, Etobicoke, and Scarborough (Paul Theil & Assoc. and Beak Consultants, 1995).

[3] Ontario MOEE, Storm sewer outfalls, The City of Toronto (Maunder et al., 1995).

The following paragraphs highlight some key results of the Skyway Bridge study. The highest mean metal concentrations in whole-water samples were observed for zinc,

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copper, and lead (event mean concentrations of 0.337, 0.136, and 0.072 mg/L, respectively) and were largely comparable to American and European literature values. Lead was the one exception. It was significantly lower than values reported elsewhere and reflected the recent adoption in Canada of unleaded gasoline. Zinc, nickel, and copper in the dissolved phase accounted for 35 to 45 percent of concentrations in whole-water samples.

PAH-event mean concentrations in whole-water samples ranged from 0.015 to 0.5 μg/L for individual compounds. Dissolved phase PAH concentrations accounted for less than 11 percent of whole-water concentrations. Data for runoff sediment (not tabulated here) revealed high mean concentrations of zinc, copper, and lead (997, 314, and 402 μg/g), indicating the sediment to be contaminated for all three metals at the Severe Effect Level (SEL) based on current Ontario Ministry of Environment and Energy guidelines for sediment quality. Sediments exceeding the SEL are considered heavily contaminated and detrimental to the majority of sediment-dwelling organisms (Ontario Ministry of the Environment, Conservation and Parks, 2019). Concentrations of metals in the fine-grained sediment (< 45 μm-size fraction) were greater than in whole-sediment samples, but this enrichment was insignificant in terms of total metal loads.

The study concluded that the uncontrolled discharges of highway (bridge) runoff could significantly impact receiving water quality and may require remediation by stormwater best management practices such as water quality inlets, grassed ditches, vegetation strips, detention ponds, and wetlands.

The Skyway Bridge study focused its attention on heavy metals and PAHs. In cities where they are used, road de-icing salts (Section 4.2.9) can also be a major cause of contaminated highway runoff and may locally elevate salinity concentrations to levels found in brine (Figure 48). While the majority of the salt-laden runoff is directed to urban streams, some inevitably infiltrates, severely impacting groundwater quality (Howard & Haynes, 1993; Howard & Beck, 1993). In some parts of the world, urea and synthetic organic compounds are also used for de-icing purposes, often causing similar impacts on receiving waters including groundwater (e.g., Wejden & Øvstedal, 2006).

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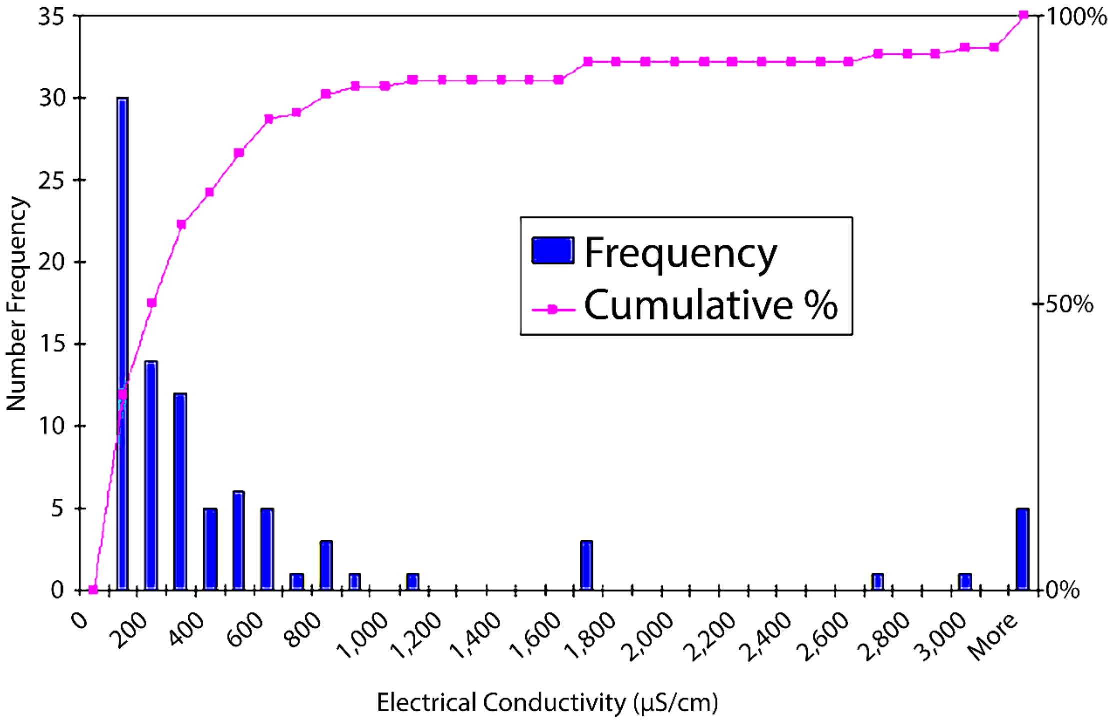

Figure 48 - The electrical conductivity of road drain gully pot waters from the campus of the University of Birmingham, United Kingdom. Samples were collected mainly in winter, spring, and autumn. Maximum conductivity recorded was found to exceed 100,000 µS/cm (characteristic of brine) (from Harris, 2007).

4.2.12 Impacts of Stormwater Runoff on Groundwater Quality

Stormwater runoff can enter the subsurface in one of three ways:

1. naturally, as indirect recharge following its passage across a sufficiently permeable land surface;
2. via engineered structures such as permeable pavements, soakaway pits, and stormwater detention ponds (Pitt et al., 1994); and
3. as artificial recharge involving various types of infiltration basins or recharge wells.

Short-term detention will allow suspended solids/sediments to settle; it may also cause a significant reduction in BOD and COD. In most cases, any remaining solids/sediment will be removed by natural filtration during recharge; the overall effect is to reduce the potential contaminant load on the aquifer. Exceptions may include
recharge into karst systems, where filtration will be minimal; and
where untreated runoff is introduced to the aquifer via recharge wells.

Despite the removal of suspended material, stormwater runoff can be problematic due to contaminants that remain in the dissolved phase. Table 24 shows the potential threat that various groups of urban stormwater pollutants pose to groundwater when their propensities for filtration and retardation are considered (after Zhang et al., 2003; Pitt et al., 1994). In Table 24, high mobility refers to contaminants with a low potential for sorption (attenuation) in the shallow subsurface, while high abundance indicates a likelihood for

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high concentrations and/or high frequency of detection in urban stormwater. A high filterable fraction indicates that a significant proportion of the pollutant associated with particulates can be readily removed by routine sedimentation controls. The values in the table are based on the assumption of a worst-case scenario whereby local sediments are represented by clean sands with a low organic content and minimal potential for attenuating dissolved contaminants. By comparison, sediments rich in clay and organic materials would likely reduce the mobility of many organic contaminants to a significant degree.

Table 24 - Potential for groundwater contamination by urban stormwater pollutants (Zhang et al., 2003; Pitt et al., 1994).

Class	Compound	Groundwater vulnerability			
		Abundance in stormwater runoff	**Fraction filterable**	**Mobility in sandy, low organic soils**	**Surface infiltration with/without pre-treatment**
Nutrients	Nitrates	Low/Mod	High	Mobile	Low/Mod
Pesticides	2, 4 -D	Low	Low	Mobile	Low
	Lindane	Mod	Low	Intermed	Low
	Malathion	Low	Low	Mobile	Low
	Atrazine	Low	Low	Mobile	Low
	Chlordane	Mod	Very low	Intermed	Mod
	Diazinon	Low	Low	Mobile	Low
Hydrocarbons	VOCs	Low	Very High	Mobile	Low
	BTEX	Low	Mod	Mobile	Low
	PAHs	Low	Very Low	Mobile	Low
	MTBE	Low/Mod	Low	Mobile	Low
	Phthalates	Low/Mod	Low	Low	Low
	Phenols	Mod	Low	Intermed	Low
Pathogens	Enterovirus	Present	High	Mobile	High
	Shigella	Present	Mod	Low/Intermed	Low/Mod
	Protozoa	Present	Mod	Low/Intermed	Low/Mod
	Pseudomonas aeruginosa	Very High	Mod	Low/Intermed	Low/Mod
Heavy metals	Nickel	High	Low	Low	Low
	Cadmium	Low	Mod	Low	Low
	Chromium	Mod	Very Low	Intermed	Low/Mod
	Lead	Mod	Very Low	Very Low	Low
	Zinc	High	High	Low	Low
Salts	Chloride	High (seasonal)	High	Mobile	High

In practice, little can be done to protect groundwater from the natural infiltration of polluted stormwater runoff other than to prevent the runoff from being contaminated at the outset. However, much can be done to protect groundwater from polluted runoff when aquifer recharge takes place as a consequence of engineering design. A particular concern is the artificial recharge of groundwater using stormwater, especially when the stormwater is generated in an urban environment. As urban populations grow and

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demand for water increases, more cities have turned towards various means of artificial recharge (recharge enhancement or managed aquifer recharge, known as MAR) as presented by Dillon and Pavelic, 1996; Chocat, 1997; and Pitt and others, 1999. Methods often include the use of permeable pavement and recharge basins to encourage slow vertical drainage to the aquifer but may extend to borehole injection techniques that include schemes such as artificial aquifer storage and recovery (ASR) as presented by Pyne (2005) and Jones and others (1998).

Various water sources can be used for MAR including treated sewage. However, stormwater runoff is a popular choice. In most cases, stormwater is introduced directly to the aquifer with pre-treatment limited only to gravity settling of suspended solids. Where urban stormwater quality is a potential concern, relatively clean water "harvested" from roof surfaces may be the preferred option. The primary purpose of MAR is to recover the recharged water at a later stage, either at the same location (e.g., in ASR schemes) or at some site downflow of the recharge site. The former approach is suitable where the recharge water meets water quality guidelines appropriate for its intended use, while the latter takes advantage of using the aquifer's natural capacity to "polish" the water quality by attenuating any contaminants that may be present.

In some cases, the quality of the water used for recharge is better than the water in the aquifer, such that MAR practices can improve groundwater quality over time. For example, in Perth, Western Australia, Appleyard (1993) showed that the introduction of recharge via infiltration basins reduced natural groundwater salinity and increased dissolved oxygen concentrations in the upper part of the aquifer down-gradient of the site. Concentrations of toxic heavy metals, nutrients, pesticides, and phenolic compounds in groundwater near the basins were low and met Australian drinking water standards. However, phthalates (suspected to be derived from plastic litter) were detected in almost all monitoring wells close to the basins; sediment accumulating in the base of an infiltration basin draining a major roadway contained > 3,500 ppm of lead.

In conclusion, considerable care is required to avoid compromising the quality of groundwater simply for the sake of increasing groundwater reserves (German, 1987). This is especially true where urban stormwater is used for resource augmentation. Ideally, the quality of both the source water and the groundwater needs to be reliably known, and sufficient data should be obtained to allow chemical mixing and transport behavior to be adequately predicted. In some cases, however, decisions are driven more by finance than by science. For example, in Auckland, New Zealand, where stormwater runoff artificially supplies 80 percent of recharge to the shallow fractured aquifer, impacts on groundwater quality (potential and observed) have caused serious debate. In practice, artificial recharge will likely continue because the cost of obtaining alternative fresh water supplies is prohibitively expensive (Smaill, 1994).

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4.3 Urban Groundwater Quality - Compounding Factors

4.3.1 Monitoring Data

A major problem in urban areas throughout the world is the lack of sufficient monitoring data. This issue was raised in Section 2.3.4 with respect to the urban water balance and the risk that additional sources of aquifer recharge could lead to rising water tables flooded basements, tunnels, and electrical utilities. However, the monitoring data problem extends to water quality where the associated risks could be even more devasting.

In essence, urban areas are major sources of groundwater contamination, and yet the vast majority of cities have few water quality monitoring wells, and little is known about how those sources impact groundwater quality until their water eventually emerges in urban springs or as baseflow to urban streams and rivers. In many cases, the emerging water has taken several decades, perhaps centuries, to complete its passage through the aquifer system; by the time evidence for groundwater contamination is found, it is often widespread and beyond cost-effective remediation.

The problem is illustrated by modeling studies conducted by Howard and others in 1993 and 1996 to investigate the environmental fate of chemicals used and stored in and around the Greater Toronto Area, a lakeside urban region in Canada that does not use groundwater and is bereft of monitoring wells. The study was prompted by a mid-1980s investigation of shallow porewaters in the City of Toronto (Pilon & Howard, 1987) that revealed chloride concentrations as high as 14,000 mg/L (two-thirds the salinity of seawater), which was presumably the result of an intensive and growing application of sodium chloride road de-icing chemicals during snowy winter seasons. At the time, road salt application rates were not considered a serious cause for concern; groundwater quality beneath the city was not being monitored, but salt levels in receiving urban streams (the Don and Humber Rivers) and Lake Ontario were only mildly elevated and not considered serious. The area modeled is shown on Figure 49. The modeling study invoked FLOWPATH (Franz & Guiguer, 1990) to test two hypotheses:

1. that slow groundwater travel times could explain why salinity levels in the rives and Lake Ontario remained fairly low despite high salt application rates, or
2. that serious long-term degradation of urban streams (and perhaps Lake Ontario) could be anticipated as salt accumulating in the aquifer system over time eventually emerged.

Groundwater travel times determined by particle-tracking are shown for the entire modeled area in Figure 50 and for the urban core in Figure 51.

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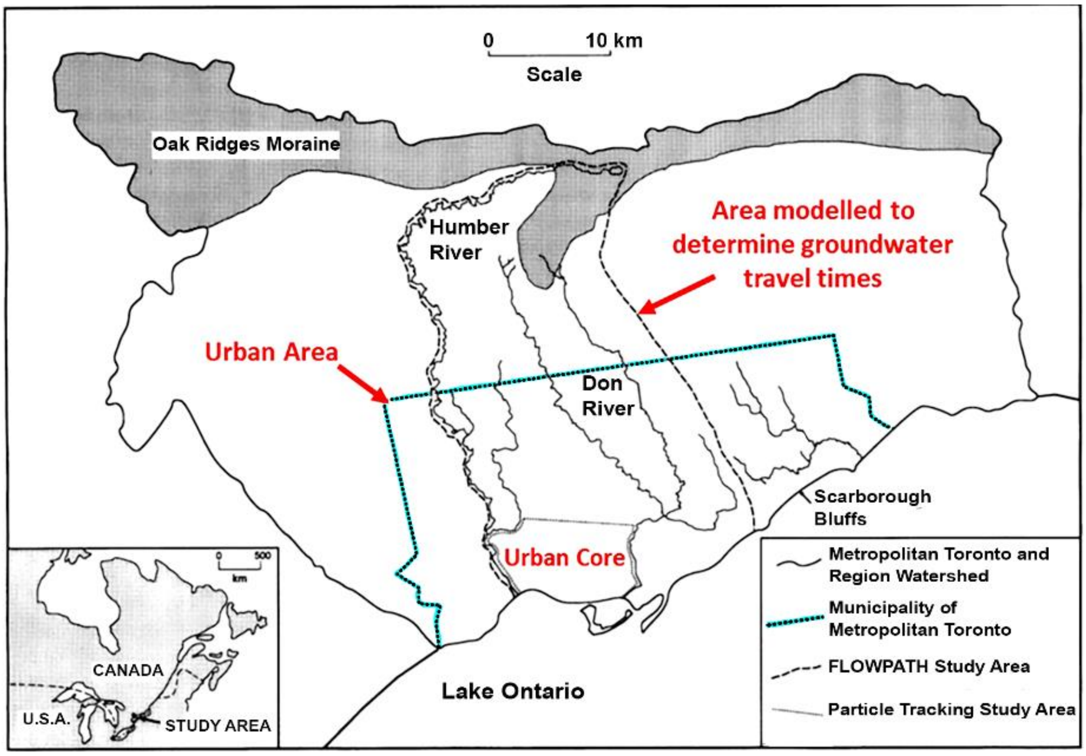

Figure 49 - The Metropolitan Toronto and Region Watershed, showing the urban area, the urban core (88 km^2), and the area that was modeled (700 km^2) to estimate groundwater travel times. Approximately 450 km^2 of the model (nearer the lake) is heavily urbanized (modified after Howard et al., 1993).

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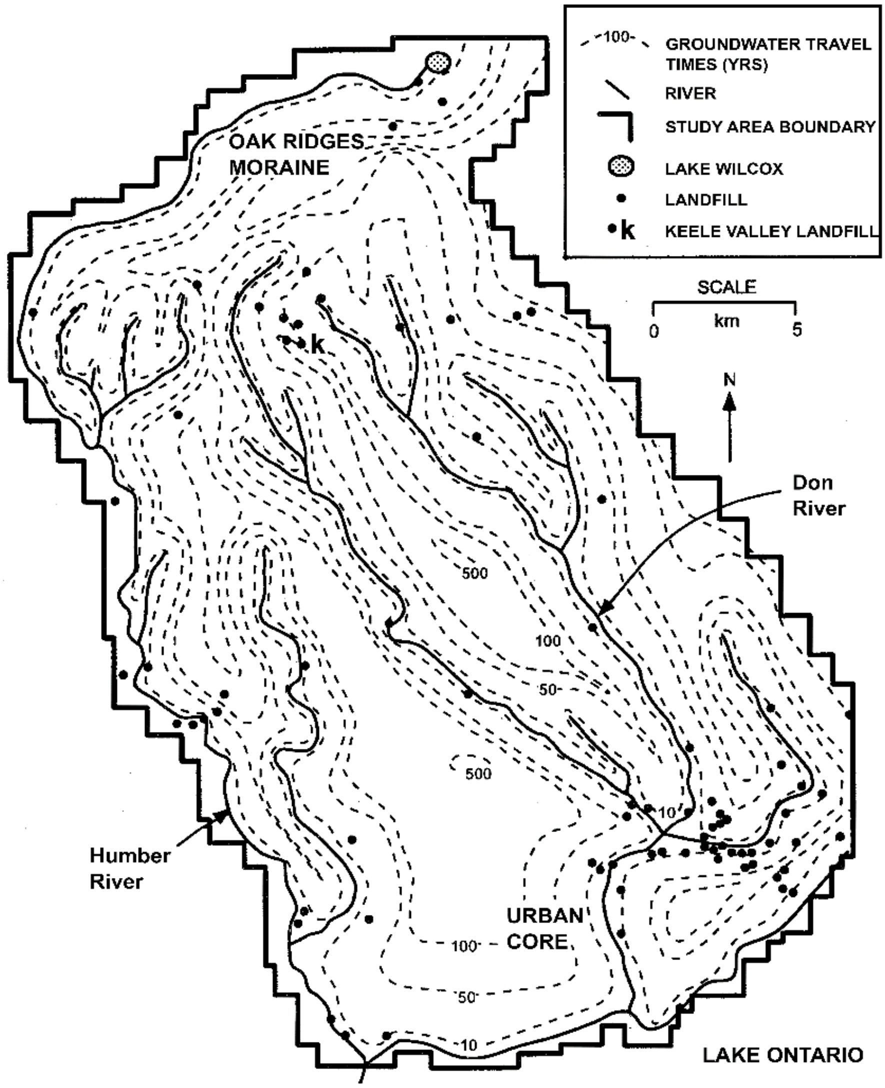

Figure 50 - The modeled area extends from the Oak Ridges Moraine in the north to Lake Ontario in the south. The southern two-thirds (around 450 km^2) is heavily urbanized. Contaminant transport isochrones (travel times) shown assume chemically conservative behavior (retardation factor = 1). Also included are the locations of 82 landfills (only one with a liner and leachate collection system: Keele Valley). None of the landfills are currently in use, and the majority have been built over (from Howard et al., 1996).

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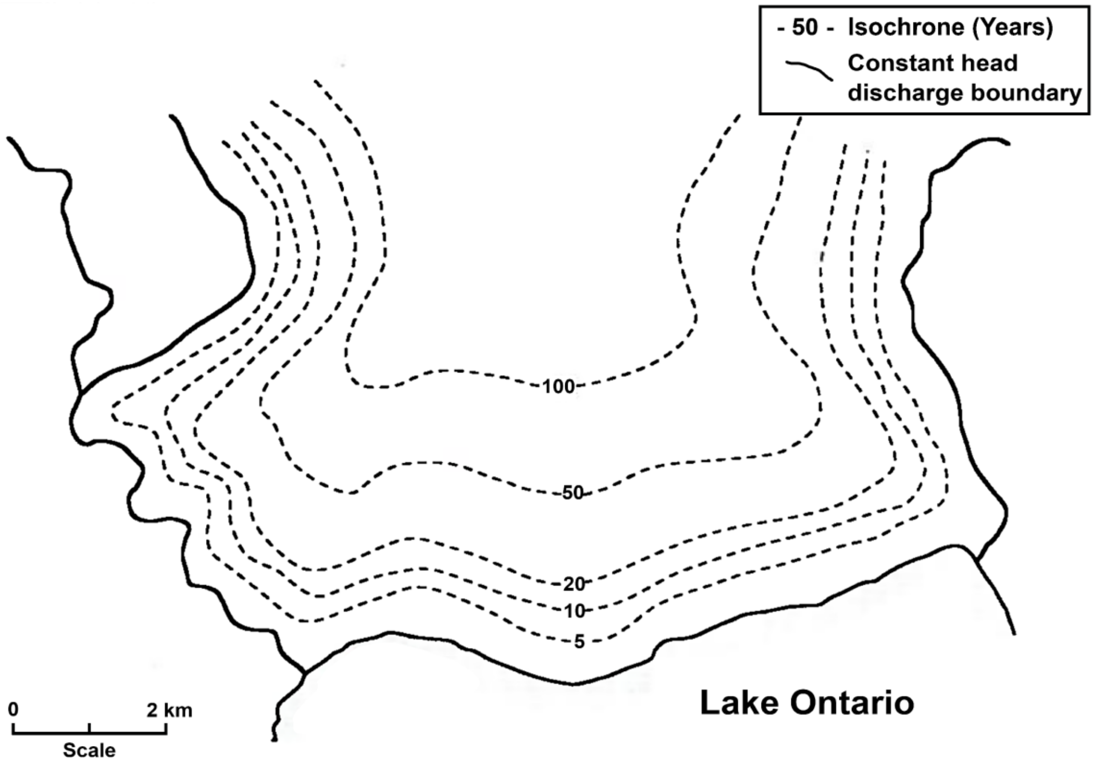

Figure 51 - Contaminant transport isochrones (travel times) shown for the City of Toronto urban core as shown in Figure 48. Travel times assume chemically conservative behavior (retardation factor = 1) (from Howard et al., 1993).

The travel time isochrones in Figure 50 and Figure 51 show chemically conservative contaminants (retardation factor = 1) that reach the water table within a kilometer or two of local rivers and Lake Ontario can be expected to emerge within five to ten years. However, contaminants released more centrally may take several hundred years to complete their journey. Recognizing that much of the city's growth has taken place during the past 50 to 60 years, the majority or urban contaminants associated with this growth have yet to make their mark on city streams and the lake. A compounding problem is that many urban contaminants do not behave conservatively (retardation factors significantly > 1) and are considerably slowed by chemical adsorption. Such compounds will take longer to reach their full impact on the streams and lake.

Figure 52 shows a cumulative area curve for travel-time isochrones in the case of a chemically conservative contaminant such as chloride (Howard et al., 1993). It is based on the travel-time isochrones for the City of Toronto's urban core shown in Figure 51. In the case of road salt—assuming source input (applied salt) remains unchanged over time—it will take 200 years to reach a chemical steady state (salt input = salt output). That is, due to prolonged groundwater travel times, it will take two centuries for water quality in receiving streams and the lake to fully register the damage that has been done to the groundwater.

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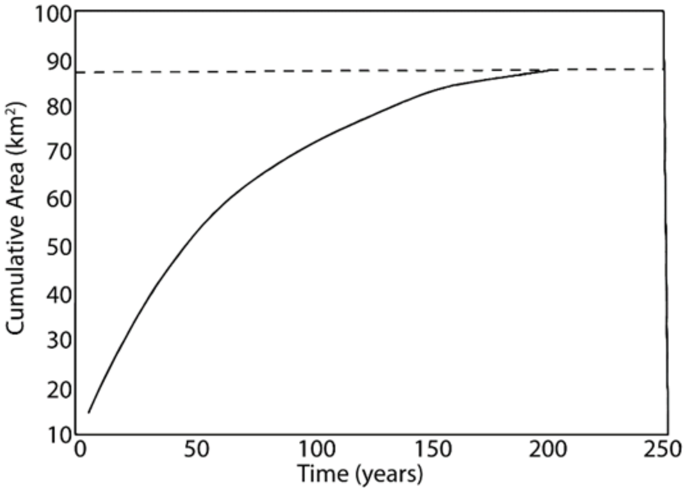

Figure 52 - Cumulative area curve for travel-time isochrones in the case of a chemically conservative contaminant such as chloride. Assuming source input remains unchanged, chemical steady state (input load = output load) is approached after 200 years (from Howard et al., 1993).

Even when chemical steady state has been reached, the salinity of the water emerging from the aquifer will not provide a true indication of the salinity levels in the aquifer remote from the discharge points, which, due to decades of salt accumulation in the system (salt in < salt out) will inevitably be higher. It is only through monitoring wells appropriately sited and dedicated to the collection of water quality data that a true measure of urban groundwater quality can be obtained.

Unfortunately, throughout the world, many regulators assess the "health" of their catchments (including urban catchments) based on the water quality observed in receiving streams. They ignore the fact that over 90 percent of the water in the catchment is likely to be groundwater (simply "out of sight" and therefore "out of mind") and an assessment of the true health of the catchment (and the ultimate, long-term health of the river) will not be possible without good data for groundwater quality. This is particularly important in urban areas that do not use groundwater drawn from wells within the city limits and therefore rarely see a need to monitor the condition of the underlying aquifer, either for water quality or for water level.

4.3.2 Influence of Urban Karst on Contaminant Transport

The term urban karst (discussed in Section 3.4, *The Critical Roles of Urban Fill, Urban Karst, and Deep Foundations*) is often used to describe the secondary permeability that develops in association with most forms of urban construction (Sharp et al., 2001; Krothe, 2002; Krothe et al., 2002; Sharp et al., 2003; Garcia-Fresca, 2007). Typically, deep trenches and drains lined with sands are the primary contributors, but physical openings such as cracks in impermeable pavement or washed-out fill material may also play a major role. Urban karst can enhance aquifer recharge and increase groundwater flow rates. It also promotes contaminant transport.

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The influence of urban karst on contaminant transport is demonstrated by a study of road salt in the City of Toronto's Highland Creek watershed between early 2005 and late 2008. An unexpected, gradual decline in chloride concentration is shown in Figure 53.

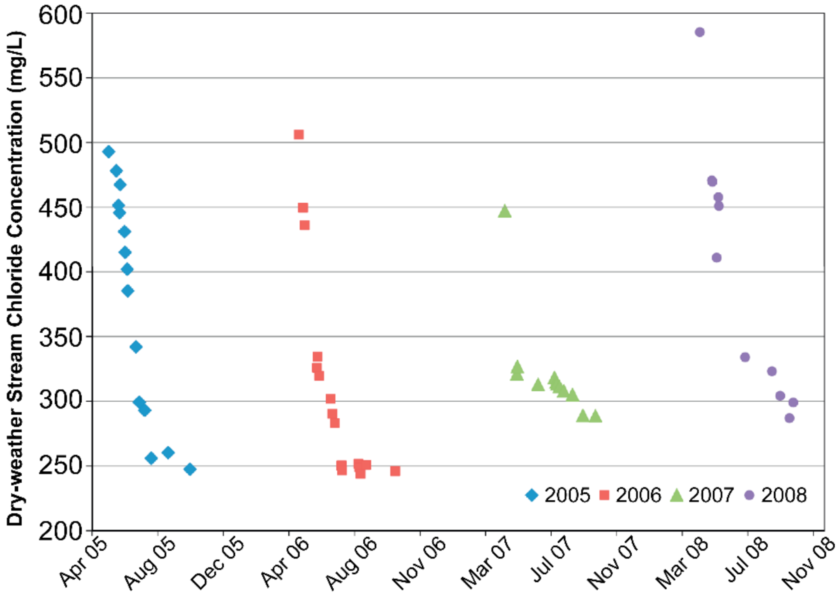

Figure 53 - Chloride concentrations in baseflow at the Highland Creek monitoring station on Morningside Avenue (Toronto, Canada) under non-winter, dry-weather flow conditions, which are generally April through August. During these conditions, we can assume the flow of water in the channel is entirely represented by groundwater discharge, that is, baseflow (after Perera et al., 2013).

Remarkably, the data display a wide range of chloride concentrations observed in the baseflow over the summer and fall months. Concentrations begin around 500 to 600 mg/L in the late spring and gradually fall to around 250 to 300 mg/L. This type of response is inconsistent with a conventional, porous flow model of contaminant transport in which slow dispersive processes act on multiple line sources (salt released to the subsurface from a dense network of urban roads and highways) to, over time, generate consistent quality baseflow to the stream (Howard et al., 1993; Howard & Maier, 2007). Instead, concentrations of chloride in the baseflow begin at relatively high concentrations, presumably influenced initially by the prior season's application of road salt, but this effect gradually dissipates and disappears by the end of summer.

The logical explanation for this unexpected behavior is the presence of a dual-porosity, urban karst aquifer in which a component of the groundwater recharge—strongly elevated in chloride due to the winter season's salting activity—moves relatively rapidly to the stream (in a matter of weeks and months instead of decades) via relatively shallow, preferential flow zones within the aquifer.

A proposed model for the observed behavior is described in Figure 54. In this figure, the magnitude/importance of the preferential flow channels is represented by the thickness of the horizontal and vertical lines.

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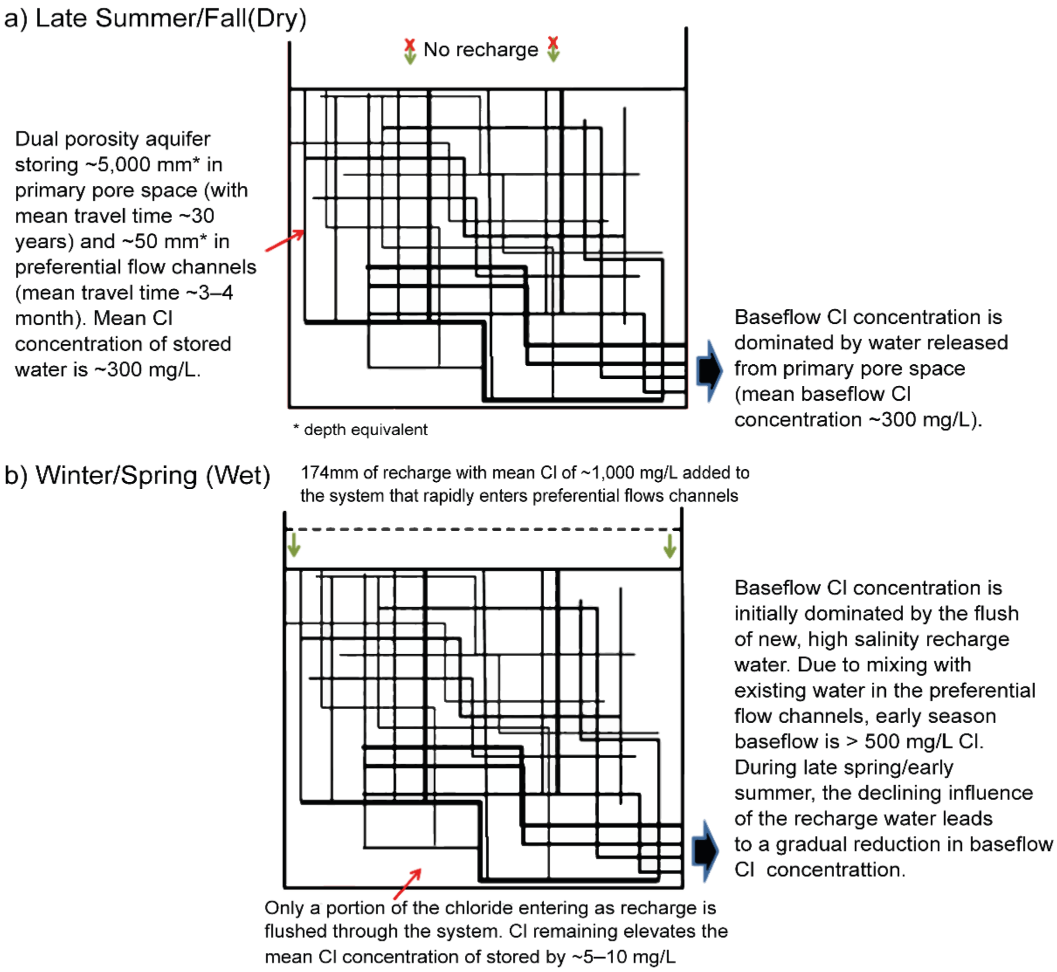

Figure 54 - Dual porosity aquifer model that explains why baseflow chloride concentrations gradually decay during the April–August period from over 500 mg/L to a steady state of around 250 to 300 mg/L. Initially, baseflow concentrations are dominated by the influx of high salinity recharge waters (chloride averaging ≈1,000 mg/L) that mix with water in the preferential flow zones to generate early spring baseflow chloride concentrations above 500 mg/L. Over the summer, recharge virtually ceases, and this allows mixing to occur between water with elevated salinity in the preferential flow channels and the less saline water occupying the primary pore space, increasing that concentration a small amount each year. Mixing causes chloride concentrations in the preferential flow channels to decline and reach steady state with similar changes to the observed baseflow (modified after Perera et al., 2013).

Figure 55 shows the annual chloride mass balance for the Highland Creek watershed as calculated by Perera and others (2013) for the period from 2004 to 2008. It should be noted that to avoid fragmentation of the salt application across calendar years, data are presented for water years that start at the beginning of November and finish at the end of October of the following year. For each year, the total mass of chloride applied to catchment roads and highways is compared with chloride that leaves the watershed in stream flow as short- and long-term outputs.

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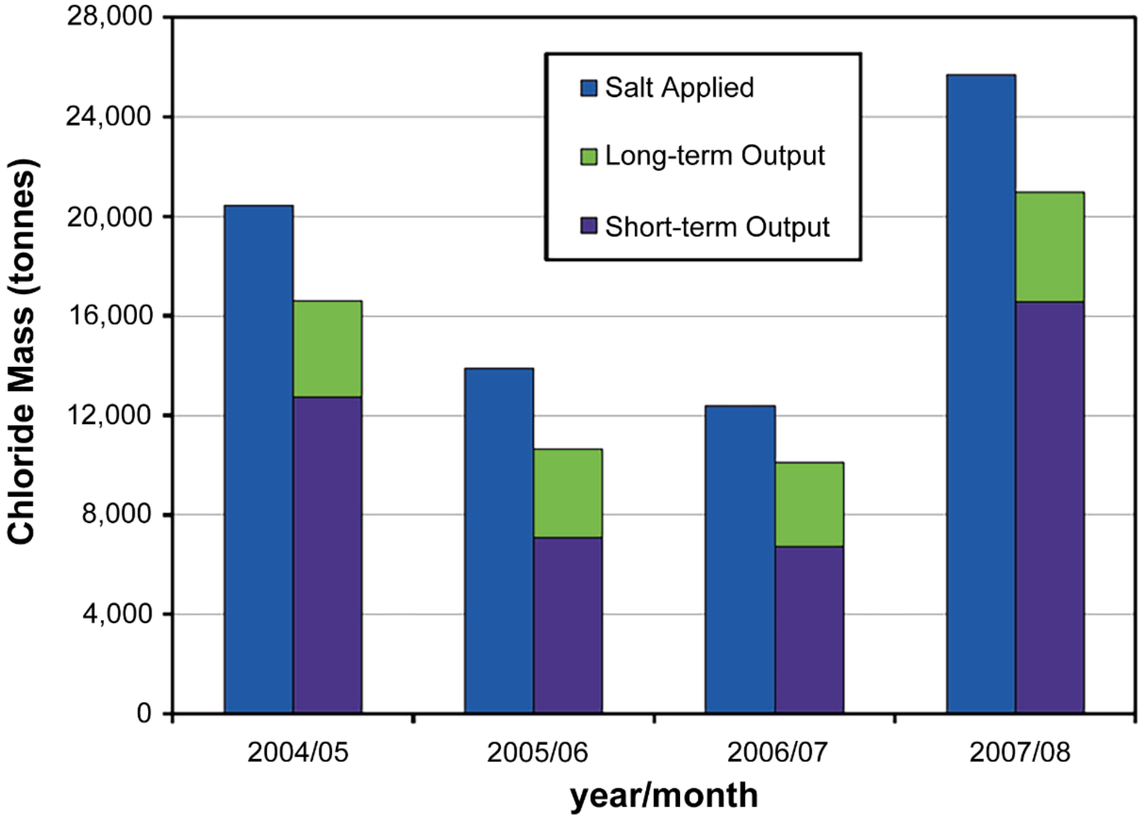

Figure 55 - Annual chloride mass balance for the Highland Creek watershed, Toronto, Canada. Short-term output refers to chloride transported via a combination of surface water runoff (overland flow) and preferential groundwater flow paths associated with urban karst. Approximately 60 percent of the chloride applied to the watershed each salting season gets flushed into Lake Ontario as short-term output, typically within 6 to 12 months of being applied. Long-term output refers to salt that enters the subsurface but does not take advantage of the preferential groundwater flow paths. It enters the stream as baseflow after many decades of travel. For the periods investigated, long-term output represents about 20 percent of the salt applied with the last 20 percent remaining in the groundwater system and increasing concentrations (after Perera et al., 2013).

Particularly significant is the discrepancy between salt applied and total salt output. This indicates a gradual accumulation of salt in the catchment, presumably in the subsurface. The slow accumulation of salt explains why the steady state concentrations of chloride observed at the end of each summer (Figure 53) show a small year-by-year increase. A gradual net accumulation of salt in urban aquifers has also been observed in other studies (Howard & Maier, 2007; Kelly et al., 2008; Novotny et al., 2009; Kincaid & Findlay, 2009; Meriano et al., 2009).

4.4 Exercises Related to Section 4

Exercises related to Section 4 are available at this link.

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5 Major Global Challenges

It is beyond the scope of this urban groundwater overview to document the long list of groundwater challenges being faced by cities throughout the world. However, it is appropriate to draw attention to the more pressing issues of concern—some known for decades, others just emerging—and provide examples for the interested reader. Essentially these challenges fall into three often closely linked categories:

- megacities and peri-urban areas,
- land subsidence due to over-development, and,
- coastal cities and the threat of sea-level rise and seawater intrusion.

5.1 Megacities and Peri-Urban Areas

Throughout the 1900s and into the twenty-first century, rapid and accelerated growth of urban areas continued unabated. In 1950, fewer than 100 cities had a population of 1 million; today, close to 600 cities have a population of 1 million or more and over 30 cities have reached megacity status with populations above 10 million (Table 25).

Table 25 - The world's megacities (based on 2018 United Nations data).

City	Country	Population[1]
Tokyo	Japan	37.400
Delhi	India	28.514
Shanghai	China	25.582
São Paulo	Brazil	21.650
Mexico City	Mexico	21.581
Cairo	Egypt	20.076
Mumbai	India	19.980
Beijing	China	19.618
Dhaka	Bangladesh	19.578
Osaka	Japan	19.281
New York City	United States	18.819
Karachi	Pakistan	15.400
Buenos Aires	Argentina	14.967
Chongqing	China	14.838
Istanbul	Turkey	14.751
Kolkata	India	14.681
Manila	Philippines	13.482
Lagos	Nigeria	13.463
Rio de Janeiro	Brazil	13.293
Tianjin	China	13.215
Kinshasa	Congo	13.171
Guangzhou	China	12.638
Los Angeles	United States	12.458
Moscow	Russia	12.410
Shenzhen	China	11.908
Lahore	Pakistan	11.738
Bangalore	India	11.440
Paris	France	10.901
Bogotá	Colombia	10.574
Jakarta	Indonesia	10.517
Chennai	India	10.456
Lima	Peru	10.391
Bangkok	Thailand	10.156

[1]UN 2018 population estimate in millions.

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Much of the growth has occurred in low-and middle-income countries of Asia and Latin America, creating an enormous burden on the regions' natural resources, the most important being water (Foster et al., 1998). Evidence suggests that dependence on groundwater for water supply is increasing in many Asian and Latin American cities (Foster et al., 2011) due to

- population growth,
- increasing per capita use, and
- reduced security of river-intake sources due to quality degradation and climate change.

The growth in groundwater demand is facilitated by the fairly modest cost of water wells and the fact that supply aquifers lie within a well's length of users (Foster et al., 1998). Unfortunately, there are no systematic and comprehensive data to quantify the growth trend, but Foster and others (2010b) estimate that over 1.5 billion urban dwellers across the world rely on groundwater for their supply.

Urban areas underlain and/or surrounded by high-yielding aquifers have allowed water utilities to increase mains water supply incrementally and at relatively low cost, one well at a time. This often improves the level of mains water-service, reduces water supply price tariffs, and discourages the need for private wells. A common problem is that groundwater resources within city limits are rarely adequate to meet the growing demand for municipal water supply, a natural scarcity sometimes aggravated by competition with water use for irrigated agriculture. Moreover, groundwater quality may also be threatened by inadequately controlled urban-pollution pressures, especially given the close physical association between wastewater handling, disposal or reuse, and the underlying phreatic groundwater (Figure 56). Thus, resource sustainability (both in terms of quantity and quality) often becomes an issue (Foster et al., 2010b).

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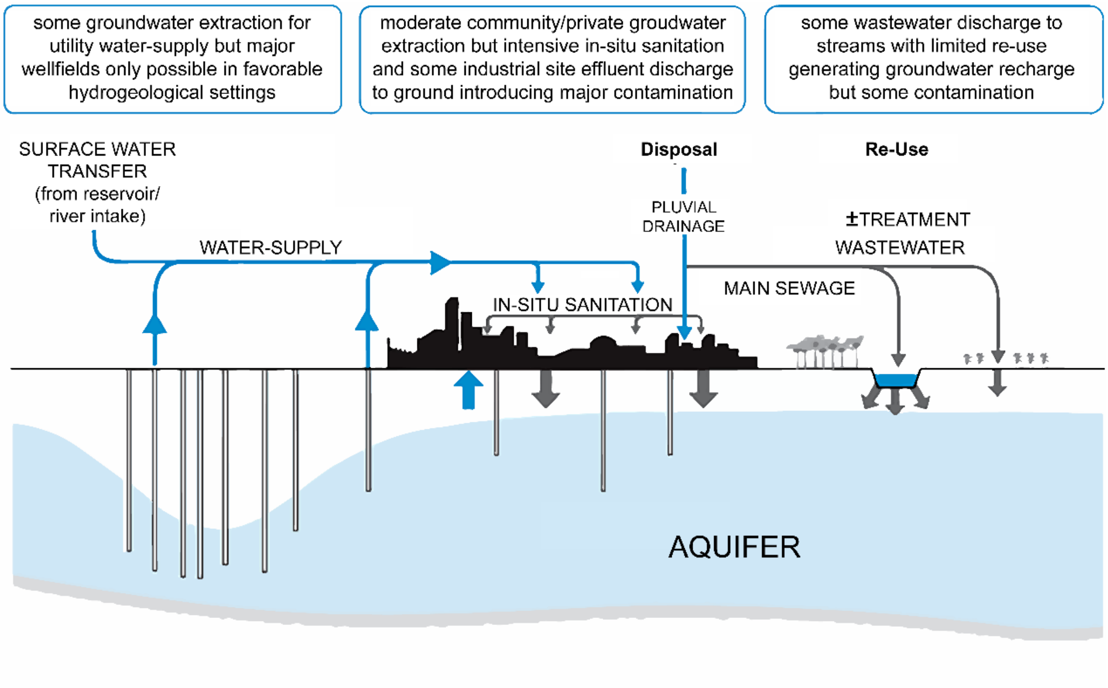

Figure 56 - Unconfined groundwater and its interaction with urban infrastructure and wastewater in cities of many low- and middle-income countries. A combination of local community/private extraction and intensive in-situ sanitation often leads to serious groundwater contamination issues (after Foster et al., 2010b).

The growth in urban groundwater use is not restricted to cities with ready access to high-yielding aquifers (from which major utility water supplies are drawn). It also widely occurs where the utility water supply is imported from considerable distance (sometimes from a surface water source). In such cases, groundwater is often used intensively for private in situ supply, a practice that has often flourished due to inadequate (present or historic) municipal water-service levels and/or high prices for water. Examples of this growth provided by Foster and others (2010b) include

- Brazil, where many cities experienced major private water-well drilling in the late 1980s in response to water supply crises caused by drought, but the water wells continue because they provide a lower-cost water supply;
- Sub-Saharan Africa where, despite much higher unit costs of drilling, water wells (for direct water collection or reticulation to standpipes) became the fastest growing source of urban water supply in the effort to meet mounting demand; and
- Peninsular India, where water well use for urban residential self-supply has become ubiquitous as a coping strategy in the face of very poor utility water services (often < 4 hours every 24 hours) and greatly reduces dependence on expensive water supplied by tankers.

Globally, the urban challenge can be expressed in quite simple terms. If cities are to remain sustainable, water supplies must be sustainable; water supplies, in turn, must be

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supported by adequate sanitation and drainage (Foster et al., 1999). Groundwater will play a critical role in developing sustainable water supplies.

The challenge of supplying rapidly growing cities with water often seems formidable, but confidence is increasing that this challenge can be met. Potential solutions to impending questions over water supply needs will be explored in Section 6, with many of the solutions rooted soundly in the scientific knowledge and understanding described in Section 3, *Impacts of Urbanization on the Urban Water Balance*, and Section 4, *Impacts on Water Quality*.

However, while science and engineering will underpin the future provision of adequate urban water supplies, exploration for new resources and improved technologies cannot provide all the answers. This is demonstrated in Section 5.1.1 and Section 5.1.2 by examining some current urban water issues, for example, conflicts over groundwater in "peri-urban battlefields" involving incessant difficulties associated with distributing water equitably and in adequate amounts to residents of Delhi, India's largest megacity. These sections will show that meeting the urban groundwater supply challenge extends far beyond the provision of safe and sustainable supplies—but requires evidence-based urban groundwater governance. Matters of water governance will be discussed in Section 7, *Urban Groundwater Governance*.

5.1.1 The Peri-Urban Battlefields

The ultimate prize may be a sustainable urban water supply, but the "battlefield" for that supply inevitably lies in peri-urban areas at the rural-urban interface, as shown in Figure 57 (Howard, 2012; Howard, 2013), which are the areas most impacted by a rapid growth in population.

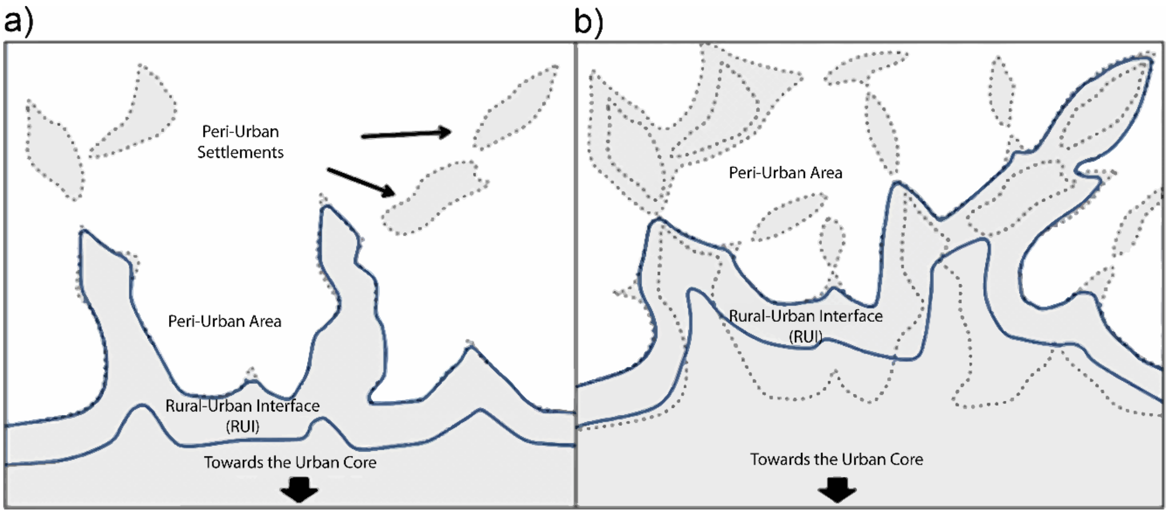

Figure 57 - Two stages of urban expansion: a) and, later, b). In each case, shaded areas are populated. The rural-urban interface (between the two solid lines) and the peri-urban area are shown. It should be noted how peri-urban settlements eventually get absorbed into the urban area and new, peri-urban settlements grow (after Howard, 2012).

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This tension is especially pronounced throughout Asia where strong competition exists between agriculture, industry, and domestic users for scarce water supplies and serious tensions over water can ignite conflict between major consumer groups. With rising concerns over economic development, food security, livelihoods, and a dire need for poverty reduction, the rivalry can be particularly intense between

- the agricultural sector for which irrigation already uses around 70 percent of available reserves, much of it to fuel Asia's green agricultural revolution (Evenson & Gollin, 2003; Giordano & Villholth, 2007; Jones, 2010); and,
- growing towns and cities that house over half the world's population and increasingly seek water to support industry, supply drinking water, and provide adequate sanitation (Chilton, et al., 1997; Chilton, 1999; Howard, 2004, 2007).

Peri-urban conflicts are an inevitable product of rapid population growth and the tensions that develop. Janakarajan and others (2006) suggest that conflict begins when opposition is expressed by at least two categories of actors whose interests are temporarily or fundamentally divergent. They suggest that tensions escalate into conflicts when verbal, legal, or physical confrontations lead to one of the parties implementing a credible threat. Most conflicts involve some combination of economic, environmental, social, or political issue.

Those living in peri-urban areas close to the rural-urban interface are most seriously affected by rapid urban growth. In some cases, growth rates are driven simply by the natural excess of births over deaths, but—in some countries—they can also be strongly influenced by an influx of migrants who are trying to escape rural poverty or are fleeing from political uprisings or an environmental crisis. At other times, rural inhabitants living just beyond the rural-urban interface are simply absorbed into urban principalities as urban boundaries expand and agricultural lands become consumed by housing, industrial estates, and sites for dumping urban waste. Whatever the catalyst for growth, it is the peri-urban lands where the highest rates of growth normally occur. According to Akrofi and Whittal (2011), key characteristics of peri-urban lands are

- high levels of land speculation;
- informal, unplanned settlement growth due to weak municipal authorities and confused, ill-defined mandates of public, private, and civil society leaders;
- rapidly changing land use practices, often incompatible, and with little or no security of tenure; and
- inadequate water services (water supply and sanitation) triggering the emergence of informal service providers.

In many cases, groundwater—often drawn from illegal, unregulated wells—represents the only viable option when a sustainable water supply is required (Foster et al., 2000).

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Since no international agreement exists to define urban settlements in terms of where they begin and where they end, assignments of government responsibility readily become just as obscure. As a result, many peri-urban areas are simply neglected by urban planners and remain unregulated and poorly serviced (Ahmed & Alabaster, 2011). Residents are easily frustrated by their inability to influence allocation of services including water. This is where the seeds of tension and conflict are sown. Peri-urban areas and the rural-urban interface are veritable breeding grounds for conflict, as they bring into co-existence groups with diverse socio-cultural and economic backgrounds, disparate hopes and aspirations, and very little, if any, opportunity to instigate change.

The most severe problems frequently occur in densely populated, peri-urban settlements characterized by foul, sub-standard living conditions with no security of tenure (i.e., slums) as shown in Figure 58 and explained by Black (1994).

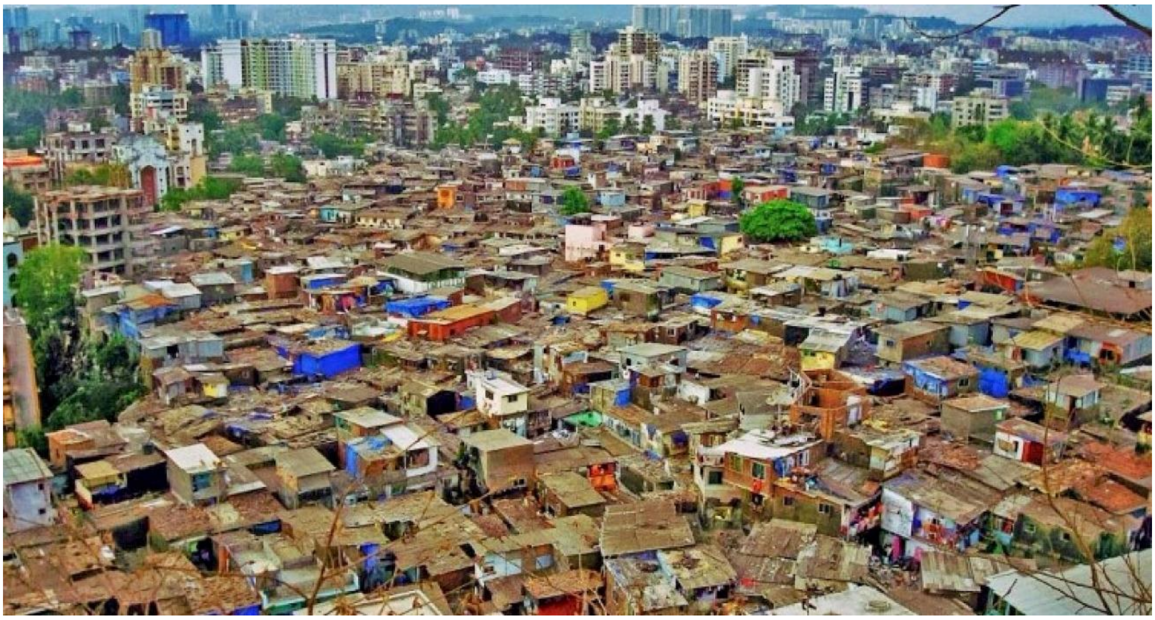

Figure 58 - Peri-urban slums of Mumbai, India. In some cities, two-thirds of the population live in slums. Slums result from a pernicious combination of weak governance, underinvestment in basic infrastructure, poor planning to accommodate growth, infrastructure standards that are unaffordable by the poor, and insufficient public transportation that restricts mobility and access to work. In some parts of the world, rapidly growing urban slums not only impact human health and environmental sustainability but also threaten both national and international security. Most slum dwellers have low social status, poor self-esteem, and no political representation to influence decision making. Many peri-urban slum areas are totally neglected by city planners and completely lack essential services. Degradation of water quality due to the unregulated disposal of industrial, domestic, and human waste is rife. Most large cities enjoy the benefits of at least some water and sanitation infrastructure in their central areas and, in many cases, this is being improved and expanded by private companies or public utility commissions, both of which can expect significant cost recoveries in the form of service tariffs or taxes. Peri-urban slums lack comparable infrastructure and services and are frequently obliged to use water sources that are unsafe, unreliable, and sometimes difficult to access on a regular or continuous basis. Sanitation, where available, tends to be limited to latrines that are often shared with so many others that access is difficult and hygiene maintenance is impractical. Unsafe drinking water and inadequate sanitation have dire implications for human health, particularly the well-being of children and the elderly (photograph: Wikimedia Commons by Nishantd85).

Discord over water is common in the urban environment (e.g., UNESCO, 2006) with conflicts triggered by numerous issues. Based on their studies in India, Janakarajan

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and others (2006) suggested the root causes of conflicts over water are normally associated with

- concerns over quantity with conflicts developing between either sectors or users (e.g., municipality versus industry or private users; urban versus peri-urban or rural);
- quality issues with conflicts arising from the threat of water that is potentially unpotable (e.g., Nagaraj, 2005); and
- regular access to water with conflicts being generated over water rights, price, or simply physical accessibility to a water source.

Few groundwater-dependent cities are able to secure an adequate water supply from within their city limits, but those that have such fortune need to maximize the quantity of the available resource while maintaining adequate water quality. This can be difficult; a vacuum of responsibility for urban groundwater leads, in turn, to a serious lack of accountability. For example, sustainability of groundwater use can be greatly influenced by a wide range of local developmental decisions that are rarely examined in a sufficiently holistic way. The types of decisions that need to be better integrated include (Foster et al., 2010b)

- urbanization and land-use planning (by municipal government offices),
- production and distribution of water supplies (by municipal water-service utilities and public-health departments), and
- installation of sewered sanitation and disposition of liquid effluents and solid wastes (by environmental authorities, public-health departments, and municipal water-service utilities).

Without integrated decision making, responsibility for the sustainability of groundwater supply is divided among a number of organizations, none of which is normally willing to—or, in fact, capable of—taking leadership for coordinated management action.

Most groundwater-dependent cities are ultimately reliant on external aquifers over which they may enjoy little, if any, jurisdiction or influence. With the huge demand for groundwater in rural areas required to meet agricultural needs, an unhealthy competition for the resource is emerging in many towns and cities; people living at the rural-urban interface and in peri-urban areas are at the heart of the conflict. Not surprisingly, there are two diametrically opposed perspectives to this urban-rural issue.

Rural communities believe that the economic and power dynamics of this competition leaves them at a disadvantage because they cannot generate comparable financial returns or are less represented in positions of power (lobby groups, politics etc.), thus are unable to influence water allocation. Studies have shown, for example, that water exported from rural areas to urban centers lead to food insecurity and unemployment (International Fund for Agricultural Development (IFAD), 2001). Farmers recognize that

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contamination of groundwater represents the greatest threat to their livelihoods and that groundwater used in rural areas on the periphery of cities is seriously threatened by polluted urban runoff or leaching of contaminated water from urban pollutant sources (Nagaraj, 2005). Regulation of groundwater is very difficult due to the number of players involved. Non-governmental organizations, foreign government assistance programs, and private companies often act independently and work with different government ministries when implementing their agendas. This lack of coordination prevents responsible resource management and promotes depletion of aquifers. Individual rural users—who typically use their shallow wells for domestic supply and small-scale livelihoods—are the first to be affected by lowered water tables.

From an urban water supply perspective, many cities believe that activities in peri-urban and rural areas pose a serious threat to the sustainability of their supply. For example, over-pumping often invites groundwater salinization, and agriculture can lead to contamination by fertilizers, pesticides, herbicides and, in some cases, wastewater. They recognize, as a best management practice, the need to protect peri-urban wellfields through regulation, for example, with their capture areas declared as ecological or drinking-water protection zones. They find, however, that any attempt to establish procedures and incentives for resource protection often encounters administrative impediments related to fragmented powers of land-use and pollution control. Too often, there is an enormous disconnect between water and land use regulations.

Overall, two important conclusions can be drawn:

1. At the global scale, the economic opportunities afforded by urban growth are seriously tempered by the social challenges growth brings (Stockholm International Water Institute (SIWI), 2011). Most of the world's urban growth occurs in low- and middle-income countries where relatively prosperous, well-serviced urban cores are surrounded by an expanding sprawl of under-serviced suburbs and peri-urban slums. In many cases, these areas support the vast majority of the urban population but are seriously neglected when it comes to proactive land use planning and the provision of adequate sanitation and drinking water services. It has become clear that developed world solutions for dealing with growing cities are not readily transferable to less-developed countries where a serious disconnect occurs between water managers and urban planners. In many cases, there is no urban planning at all. Without the planning of urban space and infrastructure, opportunities to provide adequate water and sanitation services are seriously compromised.
2. Many of the world's groundwater-dependent cities rely to a large extent on peri-urban wellfields. Accordingly, there is a dire need to work with the peri-urban and rural communities to ensure resources are adequately protected and that the needs of both sets of users are adequately met. To some extent, this can be achieved through broad stakeholder participation; however,

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the ultimate challenge will be to develop aquifer management plans that provide for rural-urban co-management. While co-management of urban and rural groundwater is a worthy goal with many potential benefits for all users, this would need significant reform of current institutional arrangements. Important first steps should include increased public awareness, concerted dialogue amongst stakeholders, and data-sharing among agencies with an interest in water management. The starting point for co-management is cooperation.

5.1.2 A Megacity Example of the Urban Groundwater Sustainability Challenge - New Delhi, India

Examples of the urban groundwater sustainability challenge can be found throughout the world. In India, a notable example of urban water supply difficulties is the megacity of New Delhi, located within the National Capital Territory of Delhi. This example gives a brief—but valuable—insight into the types of challenges faced and the complexities that extend beyond solutions based on science isolated from their social and political context.

Throughout India, groundwater supply issues tend to be more problematic in cities sited on the low-storage and often poorly transmissive hard-rock aquifers of peninsular India than in cities and large towns located on the Indo-Gangetic alluvial floodplains to the northeast. For example, water scarcity issues in Lucknow City (2.9 million inhabitants) on the Indo-Gangetic Basin (IGB) flood plain aquifer (shown in Figure 59 and Figure 60) tend to be more associated with the city's aging, leaking, and regionally inefficient water reticulation system than by the availability of groundwater from thick, productive, alluvial sand aquifers (Foster & Choudhary, 2009). The IGB aquifer is also an important source of irrigation water for rural agriculture.

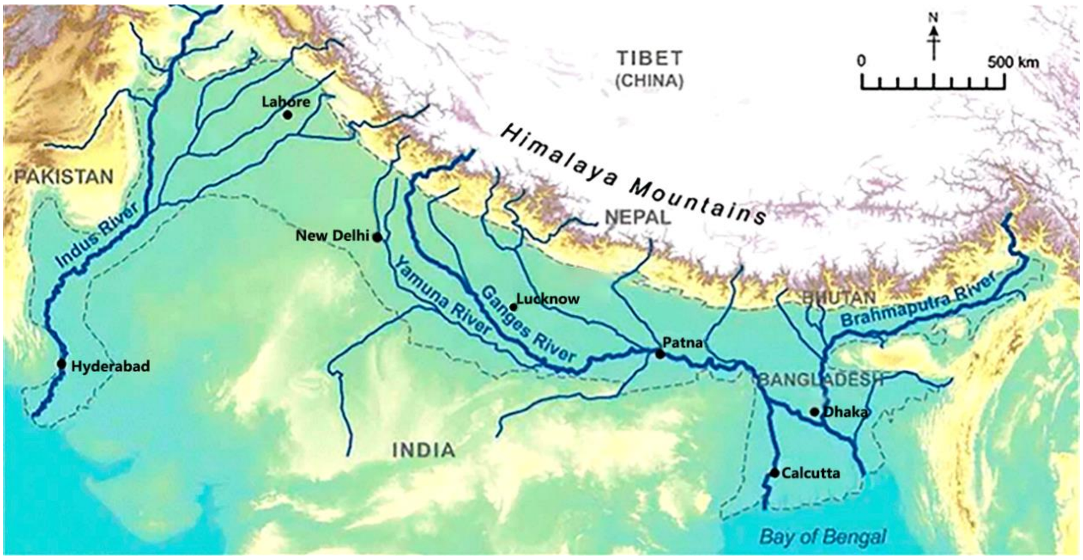

Figure 59 - Location map of the Indo-Gangetic Basin (IGB) alluvial aquifer, shown within the dashed line and with topography as background. The cities of Lucknow and New Delhi are shown (modified after Bonsor et al., 2017)

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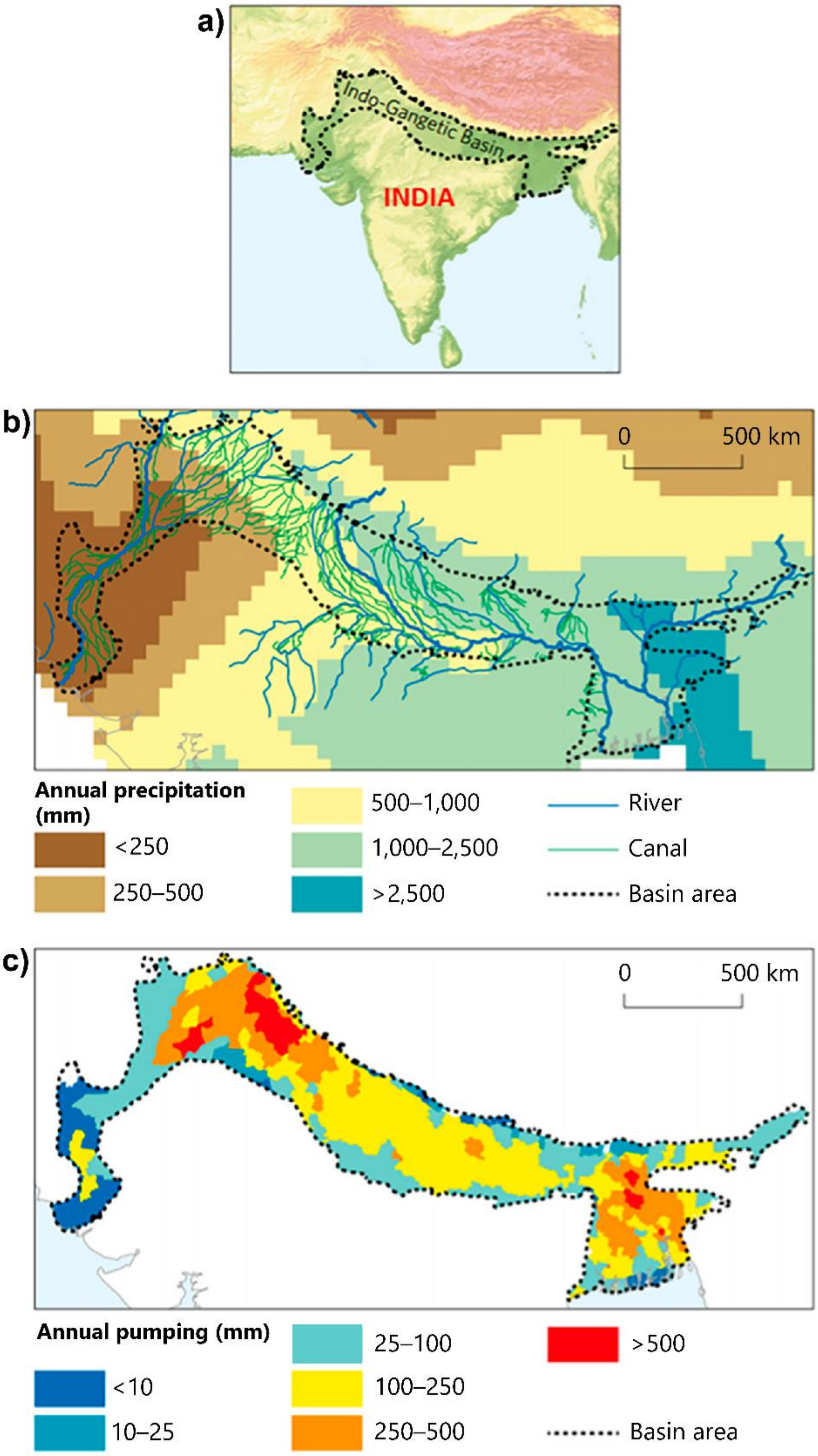

Figure 60 - The location, hydrology, and pumping from the IGB aquifer system: a) Location of the IGB in the Indian subcontinent; b) Mean annual precipitation from 1950 to 2010 and rivers and major canal distribution; c) Estimated mean annual groundwater abstraction in 2010, showing the high groundwater abstraction in northwest India, northern Pakistan, and central and northern Bangladesh (modified after MacDonald et al., 2016).

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The National Capital Territory of Delhi (the location of New Delhi is shown in Figure 59) also sits on the highly productive Indo-Gangetic Basin alluvial aquifer, and yet suffers from an unreliable water supply that is inequitably distributed and far below international standards. For example, according to recent reports, residents of the influential, wealthy Nangloi Jat area receive around 270 L/person/day from the municipality, while residents in poorer villages a few miles from Nangloi Jat—lacking political power and wealth—receive less than 5 L/person/day. Janakarajan and others (2006) suggest that the public water supply undertaking (the Delhi Jal Board, DJB) is simply incapable of meeting the city's water and wastewater needs. A recent report by the Comptroller and Auditor General of India, the official auditor of India's public sector, estimates that about four million Delhi residents, mostly in poorer communities, lack a piped water supply and must rely exclusively on relatively expensive tanker water (Figure 61).

Figure 61 - Delhi Jal Board's water tanker trucks. The effects of water supply mismanagement are mostly felt by the urban poor in city slums where supplies average just 27 L/person/day (Llorente, 2002). In contrast, Delhi's upper middle-class readily pay around \$50 for a delivery of 6,000 L of water to top up their home's depleted water tank, via private companies that are not licensed by the government. The Delhi Human Development Report 2013, released by the city's government, reveals that hours of queuing at water points is a common occurrence in the slums of Delhi and that brawls around water tanker trucks are on the rise (photography by the Times of India)

In Delhi, as throughout India, the natural reaction of consumers to the rationing of water supplies, inefficient delivery of municipal services, and lax regulations has been to develop compensatory strategies. These strategies have been described as decentralized governance structures and can be either formal or informal.

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With formal strategies, private operators (independent of the Delhi Jal Board) sell water via water tankers. Due to the weak regulatory framework, the quality of this water cannot be guaranteed; some opportunistic companies resell public water or fill their trucks with untreated groundwater from illegal wells. While the more reputable private operators would endorse stricter regulations, they receive little support from the government. They struggle to provide services which are safe, reliable, and affordable.

In response, highly informal coping strategies are developed by the poor and rich alike. The poor are more likely to steal water directly from the public distribution network via illegal connections, while higher income households will most likely install roof-top storage tanks and/or drill tube wells—usually unlicensed and therefore unregulated—for in-situ supply. In some areas, the use of unlicensed, low-cost water wells for residential self-supply has risen dramatically and has threatened to deplete the underlying aquifer. In the longer term, neither the formal (regulated) nor informal (coping) strategies can be considered sustainable from social, economic, and environmental standpoints.

All the decentralized water supply solutions suffer a relatively high cost, especially given the fact that water is essentially free. One proposed solution deserving serious consideration would be to institutionalize and, in effect, sanction "self-help" local community initiatives for a sustainable water supply. Such an approach would be beneficial in several ways.

- It would internalize costs and accounting and help maintain much-needed transparency.
- Local residents would be able to ensure decentralized water supply installations are adequately maintained.
- Rights of access would be guaranteed.
- It would encourage a more responsible and effective management of the resource including early detection of system leaks and improved management of demand.

However, such an arrangement would need considerable institutional and regulatory improvements, including the establishment of mechanisms for consultation, negotiation, and decision making. Janakarajan and others (2006) recommended that the first goal should be to simplify the existing institutional framework and redefine jurisdictional responsibilities to improve cooperation between the various decision-making levels, thus avoiding duplication of tasks and limiting discretionary powers.

The second goal would be to establish a broad-based regulatory framework, bringing together stakeholders at all social levels and giving them the power to make democratic decisions regarding water. Last, but by no means least, they recommend major governance reforms that would halt and even reverse the existing technocratic, top-down approach to the provision of water services.

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5.2 Seawater Intrusion

In some parts of the world, the impacts of urbanization on groundwater have been documented for a century or more. These impacts have usually occurred in fast-growing cities where the use of groundwater has significantly exceeded natural rates of aquifer replenishment, an activity often referred to as overdraft, overexploitation, overdevelopment, or groundwater mining. While the intensive use of groundwater in this way can generate considerable social and economic benefits, particularly in the short term, it will lower the regional potentiometric surface and lead to reduced well yields and increased pumping costs. Recent examples include São Paulo, Brazil (Diniz et al., 1997) and Ljubljana, Slovenia (Mikulic, 1997).

Reduced groundwater heads can also induce poor quality water to enter deeper parts of the aquifer from rivers and polluted shallow aquifer systems (e.g., Ahmed et al., 1999) and sometimes—more seriously—it can lead to land subsidence and/or the ingress of saline water from deeper geological formations or, more commonly, the sea. The intrusion of seawater in aquifers beneath coastal cities is explored in this section; the problem of land subsidence is examined in Section 5.3.

Close to 45 percent of the world's population lives within 150 km of the coast, many in coastal cities that benefit immensely from global trade and the flow of goods through their ports. Over half the world's megacities are located on the coast (Table 26).

Table 26 - Megacity examples of coastal cities.

Coastal City	Country
Bangkok	Thailand
Buenos Aires	Argentina
Chennai	India
Guangzhou	China
Istanbul	Turkey
Jakarta	Indonesia
Karachi	Pakistan
Lagos	Nigeria
Lima	Peru
Los Angeles	USA
Metro Manila	Philippines
Mumbai	India
New York City	USA
Osaka	Japan
Rio de Janeiro	Brazil
Shanghai	China
Shenzhen	China
Tianjin	China
Tokyo	Japan

In coastal areas, aquifers are threatened by seawater intrusion. The intrusion of seawater into coastal aquifers is a natural consequence of the density contrast between fresh and saline water (Ghyben, 1888; Herzberg, 1901; Carlston, 1963; Ghassemi et al., 1996)

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as illustrated in Figure 62. Good stewardship of coastal aquifers involves maintaining the integrity of the saline water wedge and keeping the saline water a safe distance from supply wells. This task is difficult at the best of times and will become more challenging under conditions of climate change and sea level rise (Webb & Howard, 2011). The saline water wedge, shown in Figure 62a, is normally separated from the freshwater body by a narrow transition zone of variable density. In some cases, the wedge can extend naturally for many kilometers inland (Bear, 1972; Raudkivi & Callander, 1976). Providing conditions remain undisturbed, it will remain stable, its position being largely defined by the freshwater potential and local hydraulic gradient.

However, when the aquifer is perturbed by freshwater pumping, by sea level change, or by changing aquifer recharge rates, the saline body will gradually move until a new equilibrium condition is established. Serious problems can emerge when saline water from the saline wedge is drawn into pumping wells and degrades water quality. Typically, this occurs at individual wells where sustained pumping lowers the freshwater potential in the immediate vicinity of the well and causes saline water to be drawn upwards to the well, a phenomenon known as upconing, as shown in Figure 62b. Often this type of problem is very localized and can be rectified by distributing production amongst several wells—each pumping at relatively low rates but cumulatively providing the necessary yield.

A similar but potentially more serious problem occurs when a coastal aquifer is overdeveloped (freshwater pumping rates exceed natural aquifer recharge rates) on a regional scale. This results in a lowering of the freshwater potential throughout the area and a progressive and extensive invasion of the aquifer by seawater, as shown in Figure 62c. In some heavily exploited aquifers (e.g., Howard, 1987), the inflow of seawater may represent a significant component of the aquifer's flow budget.

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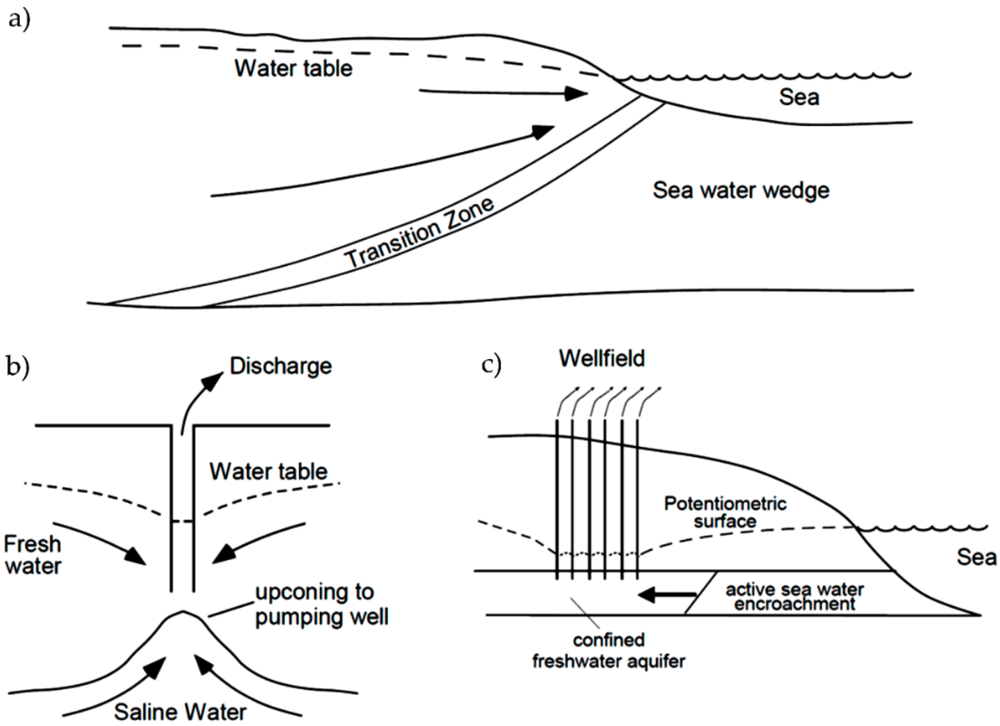

Figure 62 - In a coastal area, a) a saline water wedge is normally separated from the freshwater body by a narrow transition zone of variable density. When the aquifer is perturbed by freshwater pumping, sea level change, or changing aquifer recharge rates, the saline body will gradually move inland until a new equilibrium is established. b) Sustained pumping of individual wells can lower the freshwater potential and draw saline water upward, a phenomenon known as upconing. c) A similar but potentially more serious problem occurs when a coastal aquifer is overdeveloped, that is freshwater pumping rates exceed natural aquifer recharge rates) on a regional scale, leading to progressive, extensive invasion of the aquifer by seawater (from Ghassemi et al., 1996).

Intrusion of saline groundwater is a common problem in coastal cities throughout the world. Well known megacity examples include Los Angeles in the USA, Manila in the Philippines, and Jakarta in Indonesia, but any coastal city that pumps groundwater in significant quantities can be affected; there are perhaps hundreds of such cities globally.

The megacity of Los Angeles has a history of saltwater intrusion (Edwards & Evans, 2002) dating back to the early twentieth century. Prior to 1920, groundwater withdrawal was smaller than the aquifer's natural recharge; however, as the population grew, withdrawals increased and the potentiometric surface fell below sea level. It was not until the 1950s that seawater intrusion became a serious concern; in response, freshwater injection wells were installed as barriers to halt the seawater advance (Figure 63). These barriers were not always as effective as planned, primarily due to local hydrogeological complexities. Today, around one-third of the water supply for coastal areas of Greater Los Angeles comes from local groundwater sources. Seawater intrusion remains a carefully managed threat.

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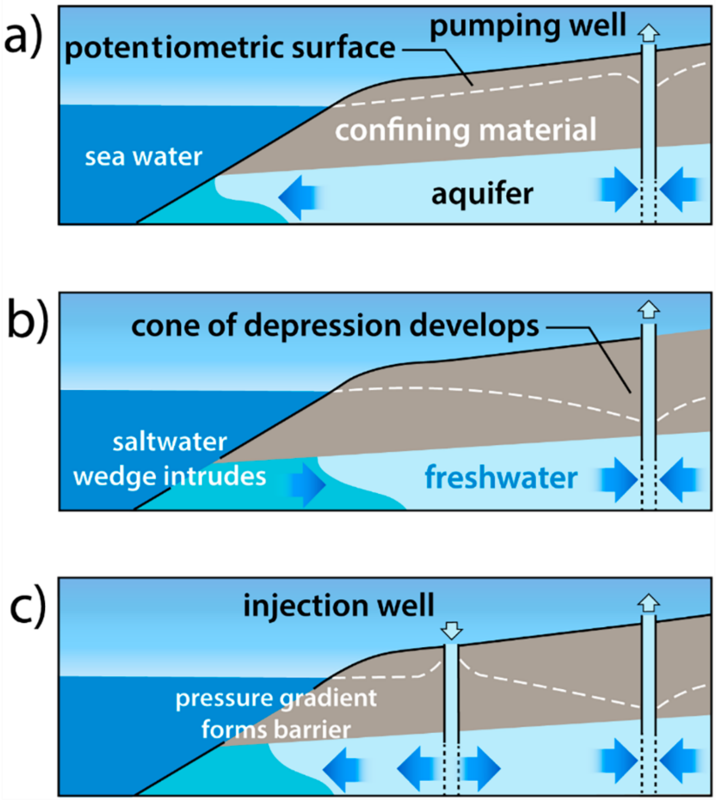

Figure 63 - To stop the advance of seawater, sets of closely spaced wells were drilled in Los Angeles to inject high-quality freshwater into the aquifer, thus creating hydraulic pressure ridges or barriers (USGS, 2018).

Similar to Los Angeles, the megacities of Manila, Philippines, and Jakarta, Indonesia, have suffered the consequences of overdeveloping their coastal aquifers. In Manila, pumping of groundwater lowered the potentiometric surface locally to between 70 and 80 m below sea level—in some cases at a rate of 5 to 12 m per year. As a result, saline water from Manila Bay extended inland as much as 5 km; samples drawn from wells in coastal areas exhibited chloride concentrations as high as 17,000 mg/L (almost 90 percent seawater).

By comparison, Jakarta's pumping rate of 650,000 m^3 per day caused groundwater levels to decline at rates of between 1 and 3 m per year to reach 20 to 40 m below sea level. As reported by Foster and others (1999), efforts in these cities to reduce groundwater pumping in favor of imported surface water have largely failed. There has been no difficulty closing down municipally-operated wells, but it has proven impossible to control a very large and escalating number of shallow, privately operated groundwater sources that are mostly unregulated, untreated, and unmonitored.

5.2.1 Chennai, India–A Case Study in Salt Water Intrusion

The coastal megacity of Chennai, India (Figure 64), is a particularly interesting case study, as it further demonstrates how good water governance can be just as important as

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sound science when it comes to resolving urban water supply problems. In terms of water, Chennai is one of the many seriously troubled cities in India (Janakarajan, 2005).

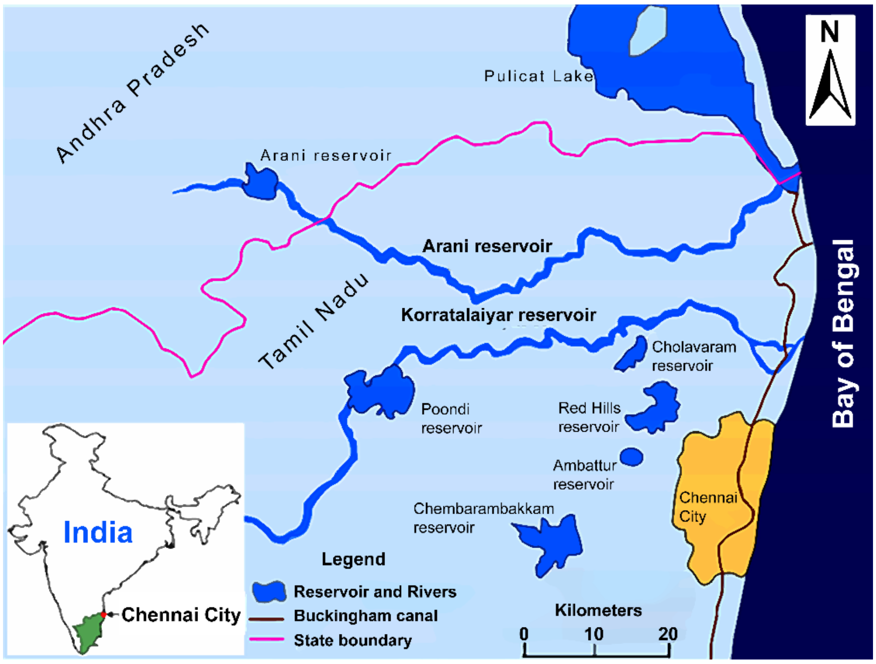

Figure 64 - The coastal city of Chennai (modified after Brunner et al., 2014).

The Chennai Metropolitan Water Supply and Sewerage Board (MWB) supplies less than 50 percent of the population's estimated water needs and, as in most Indian cities, groundwater from shallow tube-wells helps to fill the demand gap (Zérah, 2000). Unfortunately, the city's aquifers are seriously stressed; the situation has become unsustainable. In several coastal areas, water levels have declined to such low levels that seawater is actively intruding and, just to the north of the city, seawater has migrated inland a remarkable 16 km.

During the past four decades, Chennai has relied increasingly on the import of water from public wells and agricultural wells located in peri-urban villages. This has created serious social tensions (Janakarajan et al., 2006). The pumping of groundwater from beneath common lands in peri-urban villages began in 1969 when, to solve a water crisis. The MWB installed ten wells in a village just outside Chennai and brought water into the city via a pipeline. The MWB also insisted that farmers in surrounding villages sell the water they pumped from their irrigation wells, demands that were met with mixed reaction (Gambiez & Lacour, 2003). While some farmers complied, others steadfastly refused. Some found the water sales business to be immensely more rewarding than farming, and this led to a very significant reduction in irrigated farmland, seriously disrupting the social and economic dynamics of the region.

When Chennai's major sources of water supply started to decline in the 1980s, passage of the Chennai Metropolitan Area Ground Water (Regulation) Act gave the MWB full licensing power to control private well construction and establish limits on pumping.

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The Act's primary intention was to ensure that groundwater was used exclusively for domestic needs and prevent the common practice of trucking groundwater to private markets.

Twenty-five years later, it became apparent that the MWB had been the main violator of the Act, being responsible for much of the groundwater overexploitation in peri-urban villages that the legislation was supposed to deter. In fact, the MWB persisted in this practice, expanding its catchment area to include peri-urban areas lying 50 km or more from the city. The MWB also appeared to operate its tanker trucks without a license while many private tanker-truck companies complained their permit applications were being ignored.

Social unrest can escalate rapidly and, under these conditions, it becomes inevitable that regulations get flouted, even among citizens who normally respect the law. It led to a situation where many water trucks were operated illegally and, in terms of permits to drill water wells, the legislation was widely disregarded. Many industries pumped water in direct contravention of the Act, knowing that enforcement of regulations would be weak to non-existent with minimal likelihood of prosecution.

At present, no viable solution to Chennai's severe water supply issues has been found. Various megaprojects have been proposed involving major interbasin transfers of water, but the costs are prohibitive. There has been some success in creating a meaningful dialogue on the subject through the formation of a Multi-Stakeholder Platform (MSP) that initially involved a 65-member committee of farmers (both water sellers and non-water sellers), landless agricultural laborers, women's self-help groups, non-governmental organizations, researchers, lawyers, urban water consumers, and a few government officials, from both urban and peri-urban areas. More members were added to broaden stakeholder representation at subsequent committee meetings.

The MSP has tackled several important issues, including declining water tables, declining agricultural activities, emerging livelihood problems, seawater intrusion, water quality degradation, water and soil pollution, aggregate mining, and residents' growing unrest. However, progress is slow; the challenge will be to find resolutions once the MSP fully engages the MWB and similar government agencies. Meanwhile, as illustrated by an article in the Times of India (March 20, 2019), shown in Figure 65, the water crisis continues unabated, with many wells drying up or going saline.

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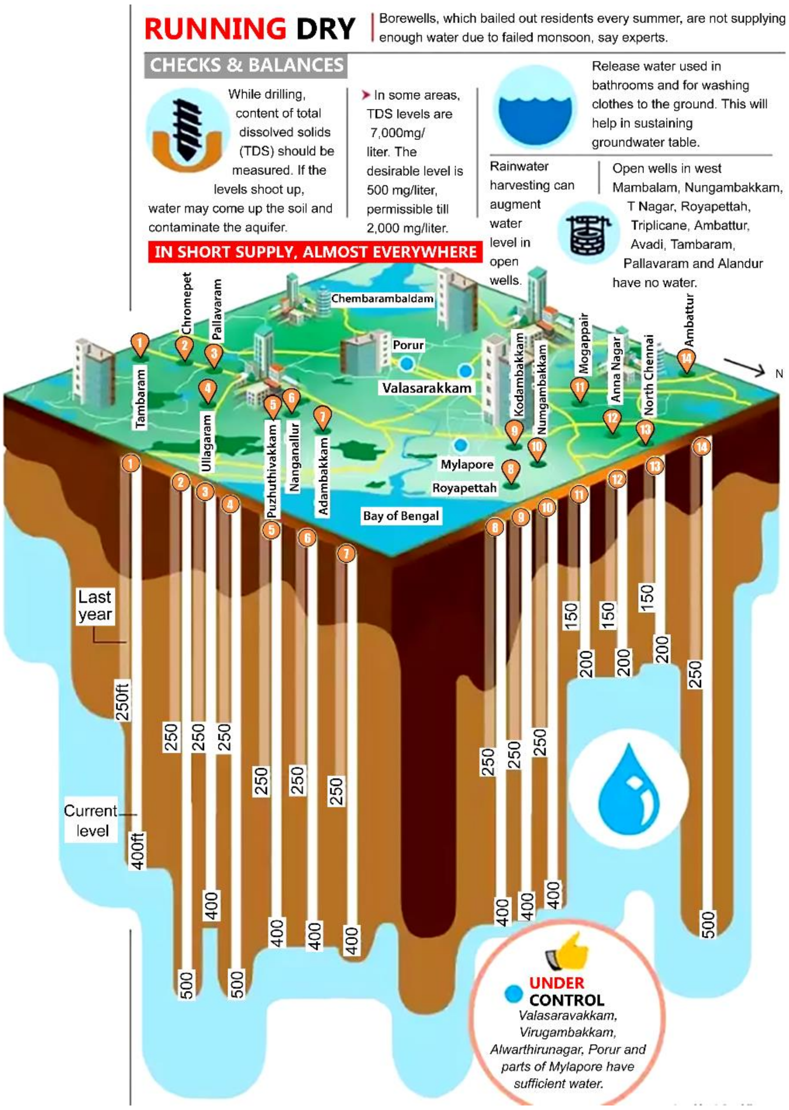

Figure 65 - A recent The Times of India newspaper article↗ (March 20, 2019) documents the continuing water crisis in Chennai (well depths are shown in feet). In some areas, water levels, recently at 50 to 60 m below ground level, are reported to have fallen by another 40 m. Locally, wells are being drilled to depths of over 150 m to find water, and there are concerns that saline water frequently encountered at these depths could pollute the shallow phreatic aquifer by cross-contamination (adapted from The Times of India, 2019).

Fortunately, not all the world's coastal city aquifers are suffering as severely as Chennai, with two of the world's megacities—Tokyo (Japan) and Bangkok (Thailand)—reporting considerable success with their respective battles against seawater intrusion. In both cases, these cities faced the dual challenge of seawater intrusion and land subsidence but managed to develop solutions capable of resolving both issues. The case studies are briefly described in the next section (5.3), which focuses primarily on land subsidence.

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5.3 Land Subsidence

A lowering of the land surface, or land subsidence, is commonly associated with the excessive exploitation of Earth's groundwater, oil, gas, and mineral reserves (Holzer & Johnson, 1985; Singh, 1992; Hu et al., 2004; Xue et al., 2005) and is known to affect well over 100 cities globally. Urban examples of land subsidence reported to have resulted from the pumping of groundwater have been reported in Tokyo, Japan (Tokunaga, 2008; Hayashi, 2008; Hayashi et al., 2009); Jakarta and Samarang, Indonesia (Chaussard et al., 2013); Rafsanjan, Iran (Mousavi et al., 2001; Rahnama & Moafi, 2009); Venice, Italy (Teatini et al., 2012); Mexico City, Mexico (Osmanoğlu et al., 2011; Yan et al., 2012; Chaussard et al., 2014); Houston-Galveston, Texas, USA (Buckley et al., 2003); and Bangkok, Thailand (Phien-wej et al., 2006). In China, land subsidence has been reported in Shanghai (Hu, 2009; Ma et al., 2018), Beijing (Ng et al., 2012), and Tianjin (Yi et al., 2011). Table 27 shows measured rates of land subsidence (local maxima) for selected locations, many heavily urbanized.

Table 27 - Measured subsidence rates for selected locations (modified after Ma et al., 2018). Rates represent the local maximum measured rate for the specified location (modified from Galloway & Burbey, 2011).

Location	Rate (mm/a)	Period	Measurement method	Source
Bangkok, Thailand	30	2006	Differential interferometry	Phien-wej et al. (2006)
Bologna, Italy	40	2002–2006	Differential interferometry	Bonsignore et al. (2010)
Changzhou, China	10	2002	Differential interferometry	Wang et al. (2009)
Coachella Valley, USA	70	2003–2009	Differential interferometry	Sneed (2010)
Datong, China	20	2004–2008	Differential interferometry	Zhao et al. (2011)
Houston-Galveston, USA	40	1996–1998	Differential interferometry	Buckley et al. (2003)
Jakarta, Indonesia	250	1997–2008	Differential interferometry	Abidin et al. (2009)
Kolkata, India	6	1992–1998	Differential interferometry	Chatterjee et al. (2006)
Mashhad Valley, Iran	280	2003–2005	Differential interferometry	Motagh et al. (2007)
Mexico City, Mexico	300	2004–2006	Differential interferometry	Osmanoğlu et al. (2011)
Murcia, Spain	35	2008–2009	Differential interferometry	Herrera et al. (2010)
Saga Plain, Japan	160	1994	Differential interferometry	Miura et al. (1995)
Semarang, Indonesia	80	2007–2009	Differential interferometry	Lubis et al. (2011)
Tehran Basin, Iran	205–250	2004–2008	Differential interferometry	Dehghani et al. (2013)
Tokyo, Japan	40	1988–1997	Differential interferometry	Hayashi et al. (2009)
Toluca Valley, Mexico	90	2003–2008	Differential interferometry	Calderhead et al. (2012)
Yunlin, China	100	2002–2007	Differential interferometry	Hung et al. (2010)
Beijing City, China	115	2003–2009	Differential interferometry	Chaussard et al. (2013)
Guangrao, China	65	2008–2009	Leveling	Liu & Huang (2013)
Thessaloniki plain, Greece	45	1995–2001	Differential interferometry	Raspini et al. (2014)

Land subsidence can cause severe structural damage that results in high maintenance costs for roads, railways, dykes, pipelines, and buildings. Recent alluvial or basin-fill aquifer systems are particularly susceptible to subsidence, as unconsolidated materials are easily compacted when excessive pumping lowers pore water pressure and

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increases values of effective stress. With global climate change, subsiding cities in coastal areas are additionally threatened by an increased risk of flooding and tsunami damage. In this section, I review specific examples: Tokyo, Japan, and Bangkok, Thailand.

Land subsidence is thought to have been first observed in Mexico City in 1891 (Ortega-Guerrero et al., 1999). It has been studied scientifically for almost a century. The land subsidence problem attracted wide international attention in the early part of the twentieth century (Pratt & Johnson, 1926; Adams, 1938; Jolly, 1939), but its relationship to fluid withdrawal (normally water, oil, or gas) was not fully understood until much later for several reasons.

- The effects were initially subtle, and detection required the establishment of a precise geodetic network with the passage of enough time for repeated surveys to reveal land level changes.
- Rates of subsidence remained very low until the rate of fluid withdrawal grew to keep pace with expanding populations and increased demand.
- The subsidence caused damage that resulted in lawsuits; the obvious response from pumpers was that they were not responsible.

There appears to be some confusion as to who first recognized the relationship between fluid extraction and subsidence. Minor (1925) and Pratt and Johnson (1926) seem to be the most popular choices with their observations of the Goose Creek oilfield near Houston in Texas, USA. Other contenders include Meinzer and Hard (1925) who noted that a reduction in pressure in the Dakota Sandstone artesian aquifer was accompanied by aquifer compression. It was this observation that led to Meinzer's classic publication (Meinzer, 1928), where he presented that water drawn from storage in a confined aquifer was released through compression of the aquifer skeleton and expansion of the water. This subsequently led to the development of the theory of elastic and inelastic storage coefficients by Theis in 1935. However, it does not appear to have been until Tolman wrote his book in 1937 (Figure 66) that the connection between aquifer compaction and land subsidence was explicitly described.

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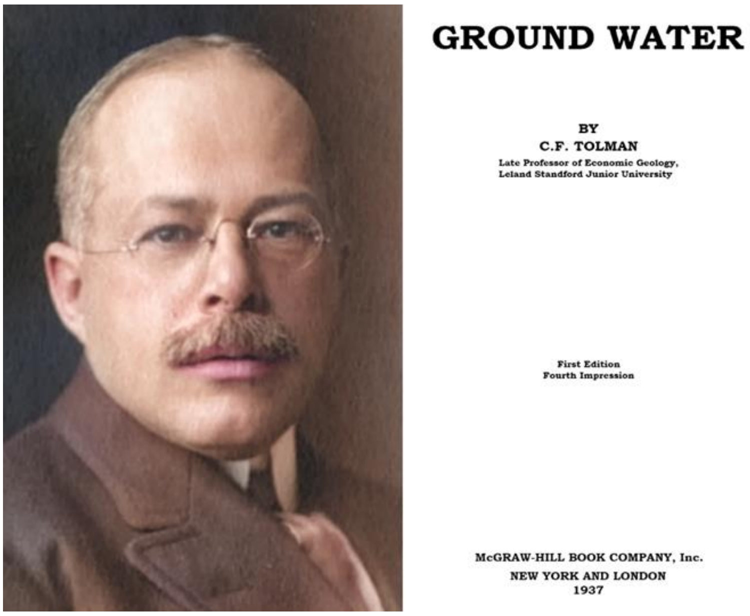

GROUND WATER

BY
C.F. TOLMAN
Late Professor of Economic Geology,
Leland Standford Junior University

First Edition
Fourth Impression

McGRAW-HILL BOOK COMPANY, Inc.
NEW YORK AND LONDON
1937

Figure 66 - The title page of *Ground Water*, the first general treatise published in English describing the new science of groundwater hydrology. It was published in 1937 and authored by Cyrus Fisher Tolman (known to his students as "Chief") of Stanford University (photograph: Stanford University Archives).

Terzaghi (1943) was responsible for further scientific developments before the mantle was taken up in the 1950s and 1960s by the USGS—for example, the work of Poland (1961) and Poland and Davis (1969). In 1965, land subsidence was highlighted in the United Nations Educational, Scientific and Cultural Organization (UNESCO) program of the International Hydrological Decade. Much of the work conducted around this time focused on the physical mechanisms of subsidence, the relative roles of elastic and inelastic compaction, and the potential for controlling the problem while meeting water supply targets.

Land subsidence due to intensive groundwater demand is also well documented in other large cities. Many of these cities (for example, Houston, Jakarta, Shanghai, Venice, Calcutta, Taipei, Tokyo, and Bangkok) are located in coastal areas where aquifers are prone to seawater intrusion (as discussed earlier); land subsidence often compounds the problem by subjecting parts of the city to flooding and above-surface invasion by the sea. Three case studies—Mexico City, Mexico; Tokyo, Japan; and Bangkok, Thailand—are presented here.

5.3.1 Mexico City

Land subsidence represents one of the earliest large-scale manifestations of excessive groundwater use in urban areas, and Mexico City—one of the world's most populated megacities—is a defining urban example.

Mexico City lies above the Basin of Mexico at a mean elevation of about 2,240 m above sea level. Pumping from the basin began in 1847 when the first well was drilled (Ortega & Farvolden, 1989). At that time, the aquifer was under flowing artesian conditions with potentiometric levels approximately 2.7 m above the land surface. The aquifer proved to be an excellent source of water and, by 1899, over a thousand water wells had been

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drilled. Despite the aquifer's popularity and the increasing demands being placed upon it, serious land subsidence problems did not begin to emerge until the late 1920s (Sanchez-Diaz & Gutierrez-Ojeda, 1997; Carrera-Hernandez & Gaskin, 2007). By the late 1930s, the land surface was subsiding at a rate of 4.6 cm/a; a decade later, the subsidence rate had reached 16 cm/a.

Only then was the link between pumping and land subsidence starting to be recognized. Although some wells in the center of the city were closed, subsidence rates reached 40 cm per year in 1959 before slowly declining (Hunt, 1990; Howard, 1992). Publications from around these times (e.g., Poland & Davis, 1969) reported that the land surface had dropped by over 9 m in some areas, sometimes leaving well casings protruding, like telegraph poles, high above the ground surface (Figure 67). The subsidence caused severe disruption of underground water mains and sewer pipes and substantial damage to roads and buildings. It also caused major alterations to surface drainage conditions.

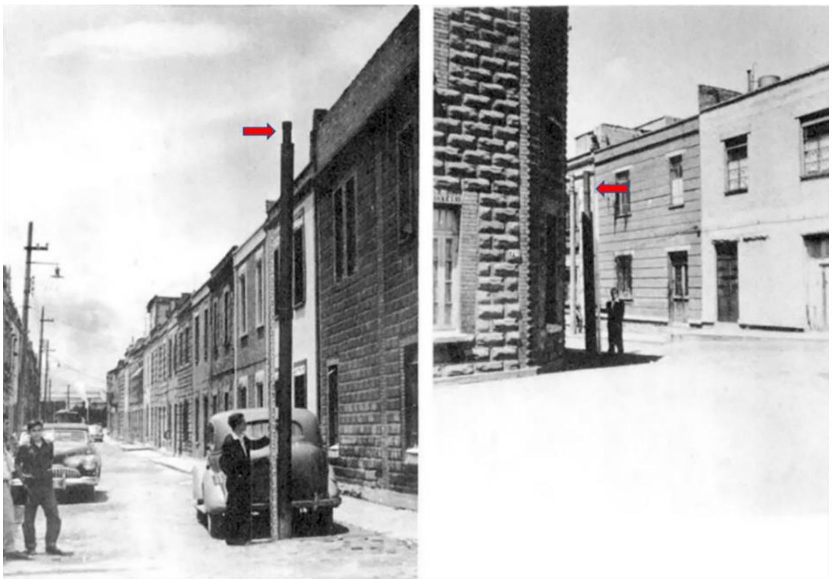

Figure 67 - Protrusion of well casings (arrowed) in the northern part of Mexico City. The wells were drilled around 1923, probably 100 m deep and the photos were taken some 30 years later in 1954. The casing to the left protrudes 5.45 m; the casing to the right protrudes 4.50 m. (from Poland & Davis, 1969) (photography by Ing R. Marsal).

Half a century later, subsidence rates have been reduced considerably (Figure 68) and the problem appears to be under control.

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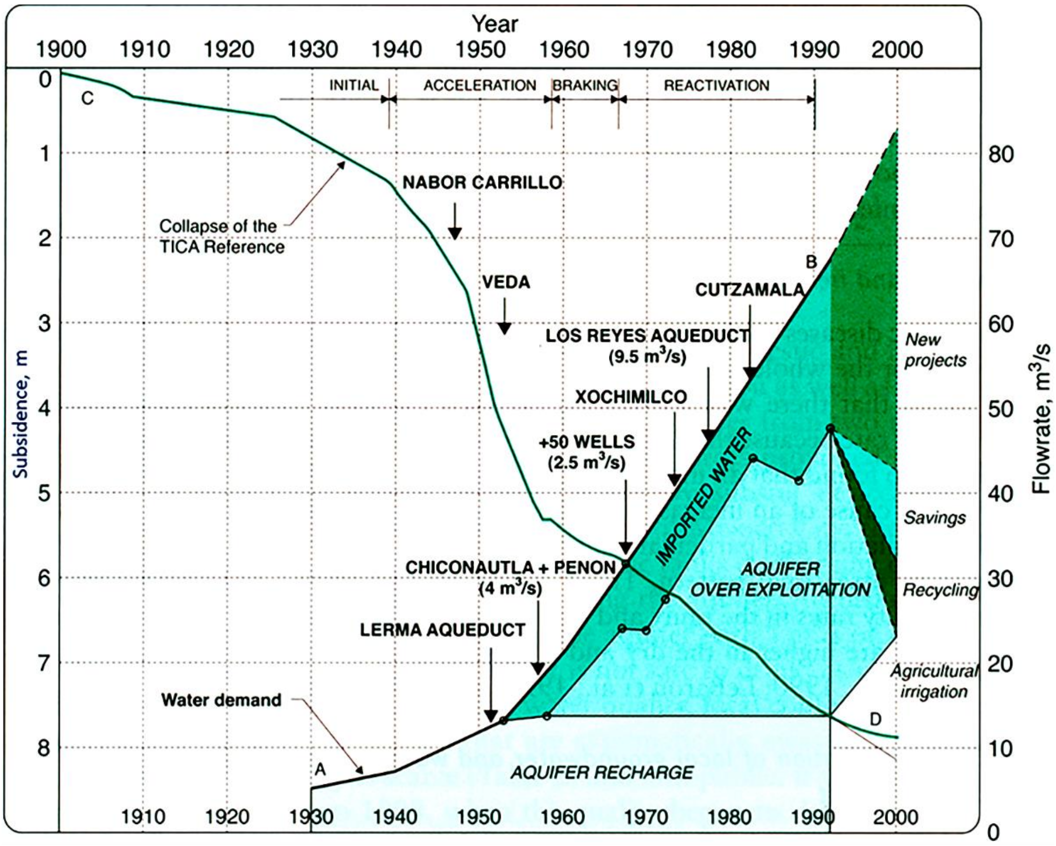

Figure 68 - Mexico City subsidence between 1900 and 2000 in relation to aquifer development. The subsidence problem appears to be under control since pumping rates began to be strictly reduced to avoid over-pumping, alternative water supplies were imported following a socioeconomic analysis, and the city introduced conjunctive management of groundwater resources with surface water. The period from 1990 to 2000 displays projected flow rates for new thrusts related to irrigation, recycling, conservation savings and new imported water projects. For the period 1990 to 2000, the magnitude of recharge was not available when the diagram was created and subsidence is shown as an extended straight line (thin black line) along with the anticipated projection (green line labeled D) suggesting stabilization (modified after Jiménez, 2009; source: Santoyo et al., 2005).

While reliable data are difficult to obtain, the Mexico City Metropolitan Area (MCMA) (population: 21.6 million) appears to derive around 70 percent of its water from the severely overexploited local aquifer system (over 45 m^3/s are pumped for municipal use compared to a natural recharge of < 20 m^3/s). The remaining water needs are imported and include (Figure 69)

- ~15 m^3/s obtained from the Cutzamala river basin to the west, a transfer that is energy-intensive as it requires a vertical lift of over 1,000 m;
- ~5 m^3/s of groundwater contentiously imported from the Lerma basin, a distance of about 60 km to the west of the city;
- ~ 9 m^3/s reclaimed wastewater via the Los Reyes Aqueduct; and
- ~ 7.7 m^3/s reclaimed wastewater.

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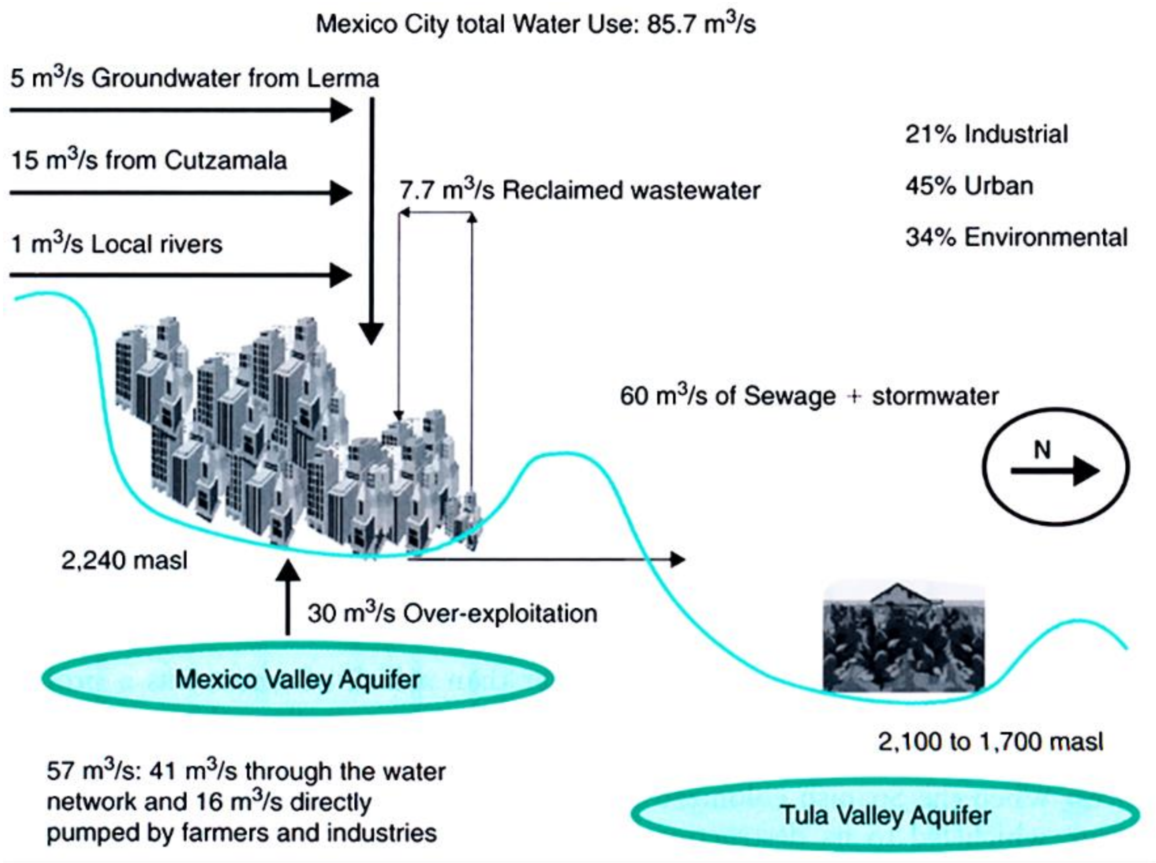

Figure 69 - Water budget for Mexico City during the early part of the twenty-first century (modified after Jiménez, 2009).

Despite considerable successes, the situation remains tenuous. The average city resident enjoys 300 L of water each day, which far exceeds the internationally accepted standard of 100 L to cover essential household needs. In reality, the 300 L "average" likely includes the 30 to 40 percent of this water that is lost via leaky reticulation systems. It also conceals the fact that affluent individuals in richer parts of the city use as much as 600 L/d, while the poor use just 20 L/d.

Although the MCMA enjoys one of the highest coverage levels of water supply and sewerage in the country (Castro, 2006), discontent with water services continues to simmer in some areas, particularly over the equitable use of groundwater, major flooding risks, poor water quality, inefficient water use, limited wastewater treatment, and health concerns about the reuse of untreated wastewater in agriculture. An important issue that has not been resolved concerns the compensation of communities that were resettled due to the construction of the Cutzamala river basin project (Tortajada, 2006).

Today, the challenges for Mexico City remain enormous, as serious interruptions in water supply would be viewed as a national crisis that could destabilize the federal government. Overcoming these challenges is made particularly difficult by the highly fragmented institutional arrangements for water management with various—sometimes overlapping—responsibilities distributed across all levels of government. Reform of water governance is long overdue.

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5.3.2 Tokyo, Japan

Tokyo, the most heavily populated coastal city in the world, is a well-known and much studied example of land subsidence. The city is situated in the southwestern part of the Kanto Plain, an area of approximately 16,000 km^2 that lies just above sea level. The plain is underlain by thick accumulations of unconsolidated sediment that gives rise to a highly productive aquifer system. Unfortunately, the sediments are readily compressible.

Land subsidence due to the intensive use of groundwater was first observed in the 1910s. While the demand for groundwater was reduced in the early post-World War II years, subsidence eventually resumed at an accelerating rate in response to the revival of industry that increased demand for groundwater. As a result, water levels declined rapidly. During the 1960s, production approached 1.5 million m^3 per day; some parts of the city reported subsidence of over 4 m (Figure 70), with the land reaching as much as 1 m below mean sea level. This raised serious alarm in a highly populated area prone to storm surges and typhoons and at risk of tsunamis.

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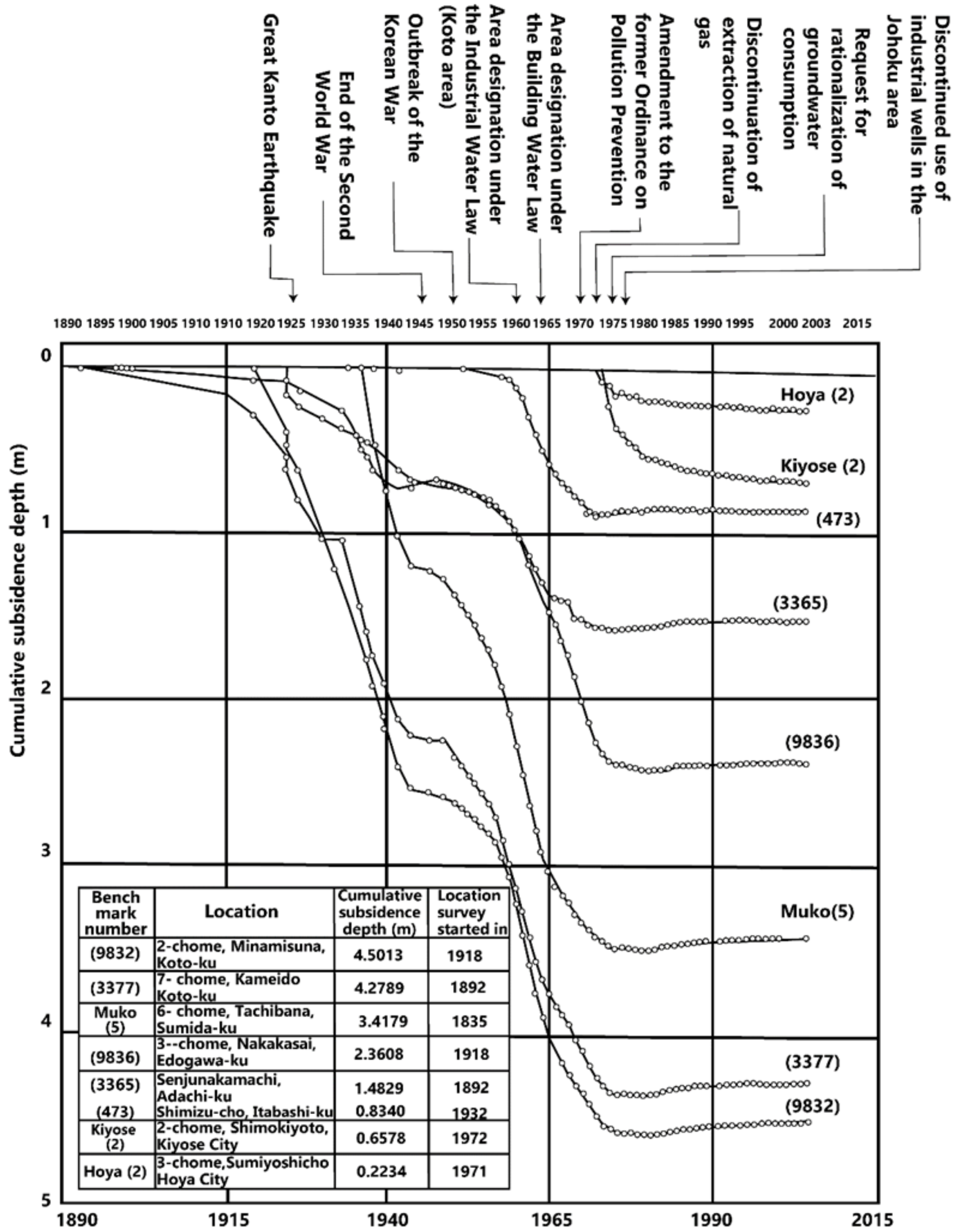

Figure 70 - Cumulative subsidence at major benchmarks in the Tokyo region from 1890 to 2003 (modified after Sato et al., 2006; source: Tokyo Metropolitan Research Institute for Civil Engineering Technology, 2004).

Various countermeasures introduced during the 1960s included raising riverbanks and construction of a sea barrier. A program was also introduced for major reductions in groundwater withdrawal. Today, the problem appears to have been largely resolved. Reductions in pumping promoted a rapid rise in groundwater levels (Figure 71) that, as indicated by Figure 70, essentially halted subsidence. In 2003, up to 550,000 m^3 of groundwater were still being pumped daily for public water supply and other uses, but the situation remained stable (Sato et al., 2006). Tokyo is no longer seriously regarded as a groundwater-dependent megacity.

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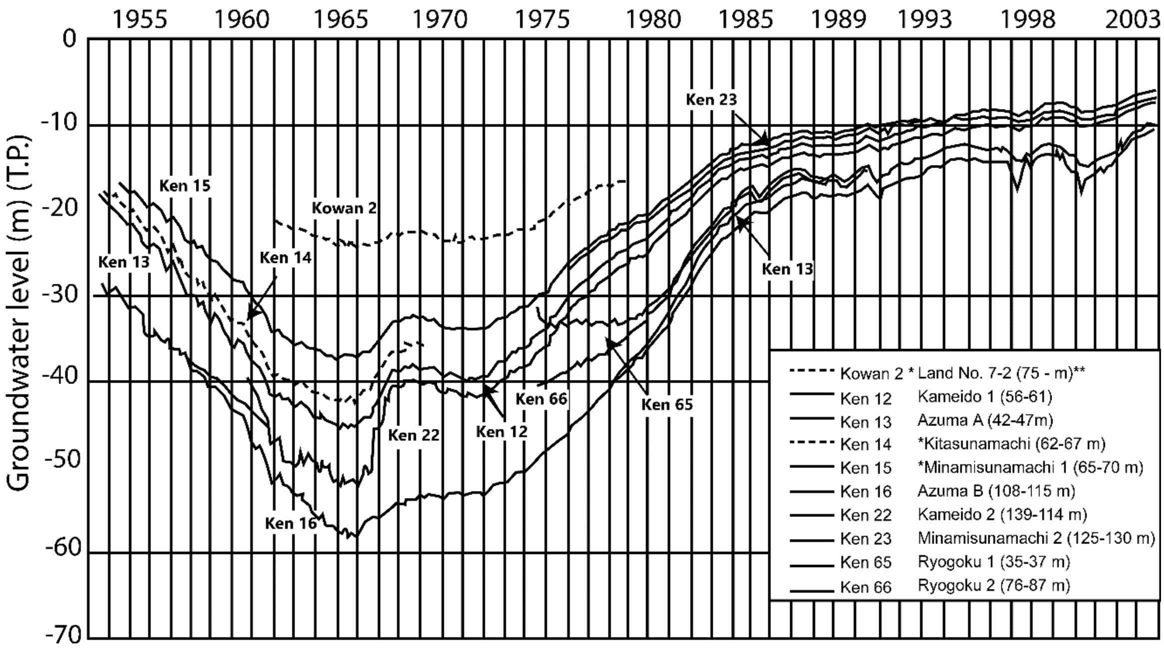

Figure 71 - Variation in groundwater levels in wells in the districts of Koto-ku and Sumida-ku between the early 1950s and 2003. A single asterisk indicates the former site of a well and the double asterisks refer to parentheses which contain values indicating screened intervals (modified after Sato et al., 2006; source: Tokyo Metropolitan Research Institute for Civil Engineering Technology, 2004).

5.3.3 Bangkok, Thailand

Bangkok, the capital of Thailand, has suffered similar problems to those in Tokyo, but its cutback on groundwater has been less extreme. Greater Bangkok occupies much of the Lower Chao Phrayh Basin, which is underlain by a thick accumulation (≈500 m) of inter-bedded alluvial and marine sediments of Pliocene-Pleistocene-Holocene geological age that support eight semi-confined aquifer units, each separated by relatively thin aquitards. The composite aquifer system is mostly confined by the uppermost Holocene Bangkok Clay, an exception being the Middle Chao Phrayh Basin to the north where the clay is thin or absent and the system is recharged (Buapeng & Foster, 2008) as shown in Figure 72.

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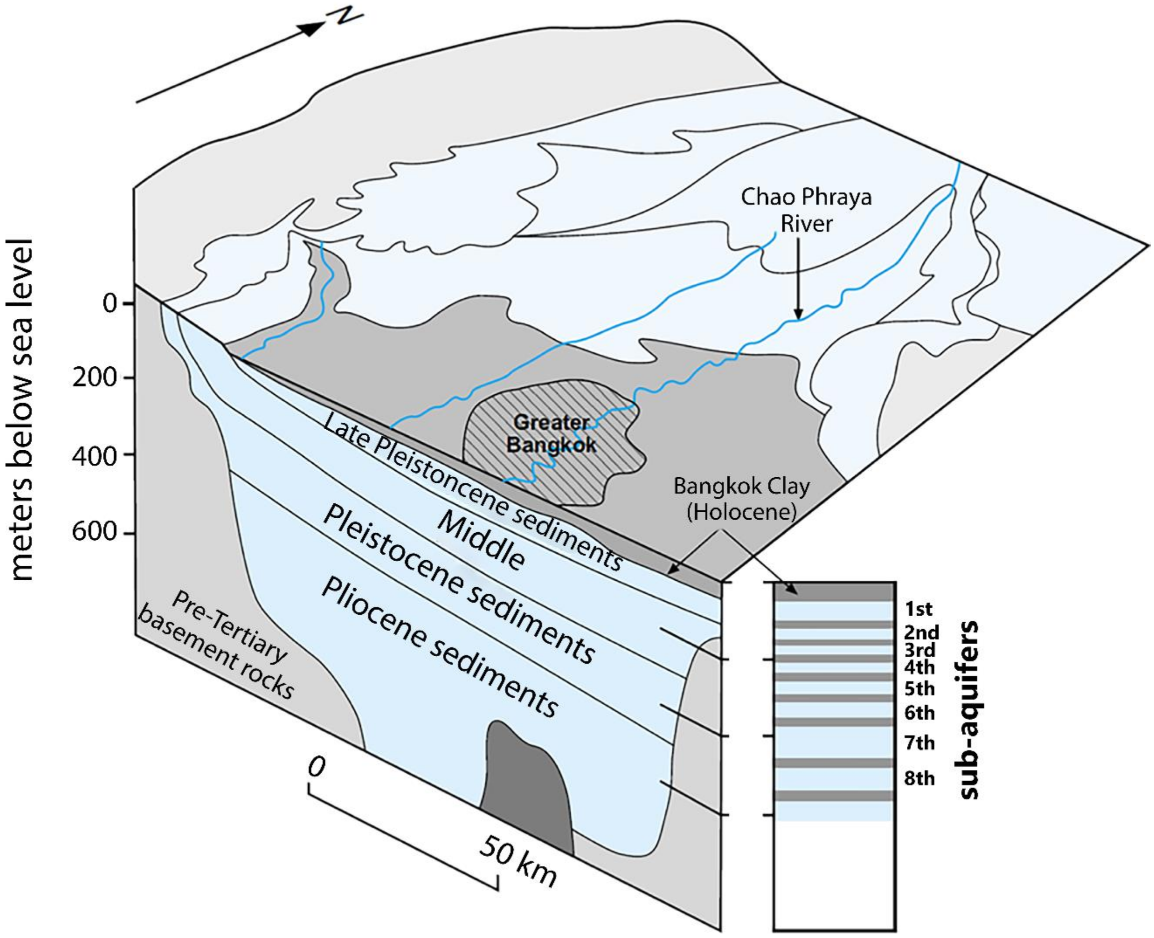

Figure 72 - Hydrogeological block diagram showing structure of the Middle and Lower Chao Phraya alluvial aquifer system in relation to Greater Bangkok (from Buapeng & Foster, 2008).

Widespread exploitation of groundwater for urban water supply began in the 1950s. By 1980, the aquifer system was producing 500 ML/d, mostly by the Metropolitan Waterworks Authority (MWA) but with private industrial abstraction becoming increasingly significant. The most productive zones were the second, third, and fourth sub-aquifers at a depth of around 100 to 250 m below ground level. As a result of pumping at these rates, groundwater levels declined significantly, reaching 40 m or more below sea level by 1985 (Figure 73).

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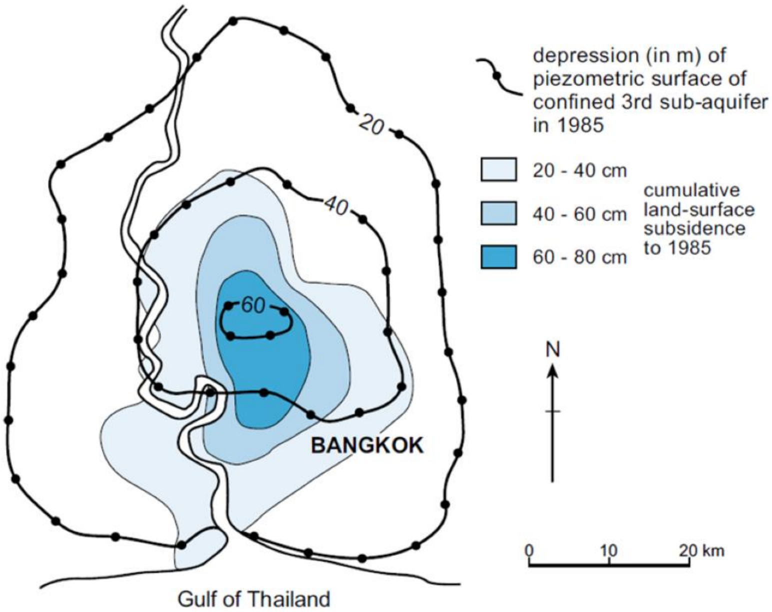

Figure 73 - Land subsidence related to groundwater abstraction in Greater Bangkok in 1985 (from Buapeng & Foster, 2008).

The land subsidence was accompanied by significant damage to the urban infrastructure and increased flooding risk during tidal surges and heavy rainfall events (Figure 74). Moreover, the depression of the potentiometric surface to below sea-level increased concerns about sea-water intrusion. Notably, chloride concentrations rose above 1,500 mg/L in the center of the city.

Figure 74 - Due to the city's low elevation and the effects of land subsidence, flooding of downtown Bangkok remains a constant problem (photography by Ken Howard).

The public water authority responded to the problem by progressively closing its pumping wells beginning in 1985 (Buapeng & Foster, 2008). Unfortunately, despite the elimination of production by the late 1990s, an increase in tariffs for domestic, commercial, and industrial mains water supply (imported from distant sources) sparked an explosion of private water well drilling that included

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- relatively shallow wells, typically 100 mm in diameter and up to 150 m deep (targeting the second and third sub-aquifers) each yielding up to 1 ML/d for large apartment blocks and some commercial users; and
- large, deep boreholes (200 to 300 mm in diameter and perhaps 500 m deep), providing up to 10 ML/d for industrial and major commercial users.

In the face of deteriorating environmental problems, the government increased its efforts to constrain pumping by

- identifying critical areas where water well drilling would be totally forbidden,
- adopting the legal power to seal water wells in areas with mains water supply coverage, and
- licensing wells and charging for groundwater according to metered (or estimated) pumping rates.

At first, water prices were set at nominal levels that provided little incentive to reduce pumping but did, at least, establish the administrative framework and provide a useful database. Later, prices were increased and structured in such a way as to ensure that the greatest financial burden was borne by industrial and commercial users in the critical areas. Public awareness campaigns were introduced and the well sealing program was vigorously pursued. Very gradually, the situation was brought under control (Figure 75 and Figure 76).

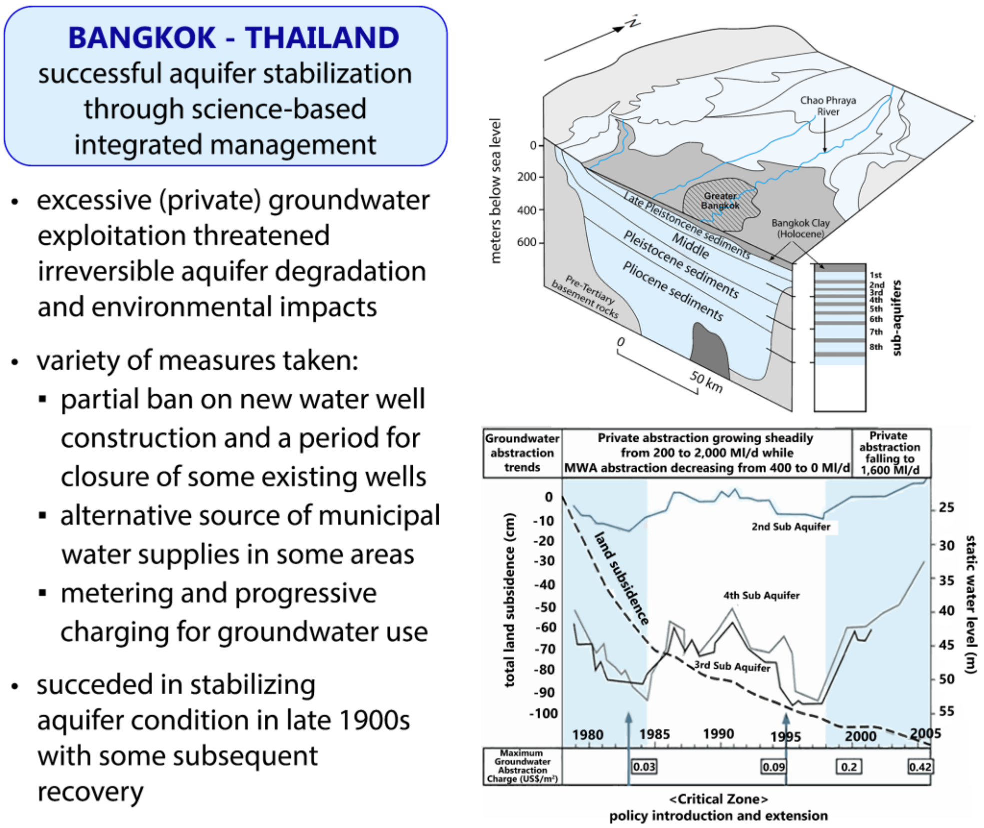

Figure 75 - By the late 1990s, the system was eventually stabilized through science-based groundwater management (Buapeng & Foster, 2008).

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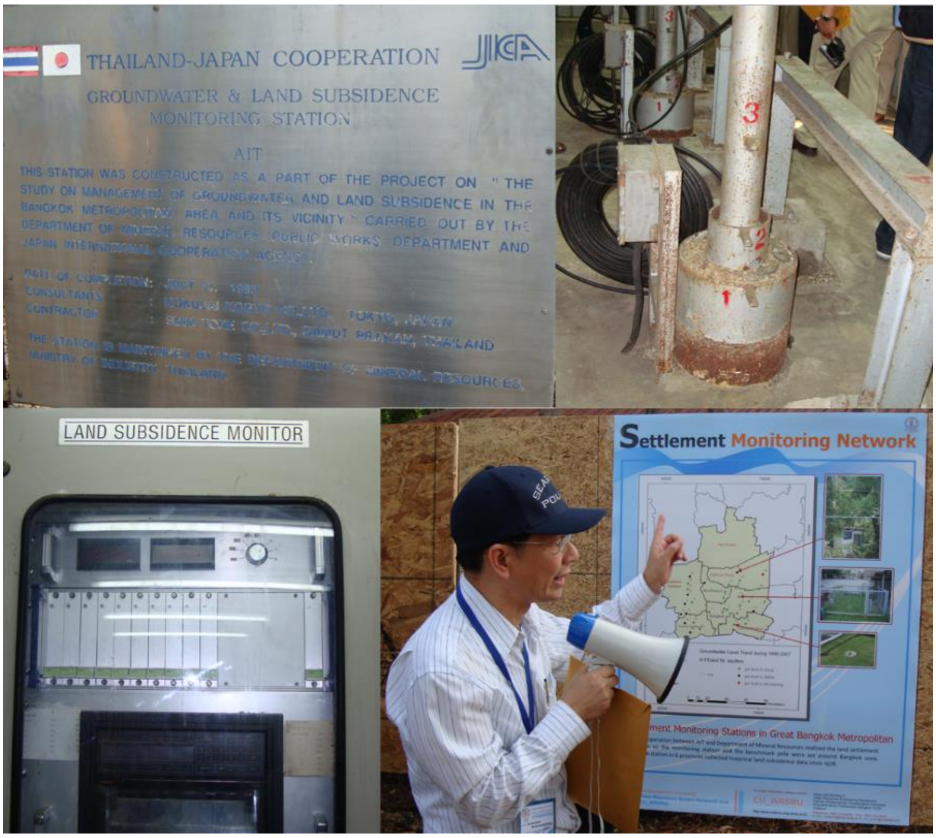

Figure 76 - In Thailand, the task of combating land subsidence has relied heavily on data provided by the country's settlement monitoring network (photography by Ken Howard).

By 2008, Greater Bangkok had just over 4,000 licensed water wells providing about 15 percent of the total water supply. Licenses are required for all wells more than 15 m in depth (i.e., that draw supplies from the main freshwater aquifers); around 58 percent of the licensed production is for industrial use. Many of the largest industrial water-users have been driven out of Greater Bangkok by the high water tariffs.

The other primary user group includes private urban dwellings and large apartment blocks that do not have ready access to mains water supply. In some districts, conflicts arose when extension of the mains water supply created substantially higher costs for water. The issues were settled by allowing water well users to retain limited use of their wells for the duration of their existing license, allowing them to maintain their wells as a back-up supply for 15 years, provided they were adequately metered and open to inspection.

As a success story, the Greater Bangkok groundwater management program provides an excellent example of adaptive management and how progressive control of groundwater pumping can bring stability to a seriously over-developed aquifer faced with severe and potentially irreversible environmental degradation. It demonstrates how consistent and persistent application of regulatory measures (licensing and charging) targeted to objectively-defined priority areas, supported by good data and properly

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explained to stakeholders, can reverse trends in groundwater resource decline and environmental damage—and with minimal discontent.

5.4 Further Reading

Section 5, *Major Global Challenges*, has only touched upon some of the many urban groundwater issues throughout the world—true life experiences from which much can be learned. As these case studies show, many of these practical, real-world issues do not find their way into the readily searchable scientific literature and do not get the attention they frequently merit.

One such body of work that deserves special consideration in any book section that purports to deal with urban groundwater relates to the work undertaken by GW-MATE—the Groundwater Management Advisory Team—a multi-disciplinary team of groundwater specialists working long-term for the World Bank with special funding principally from the Netherlands Government and supplemented by the United Kingdom and Denmark (Figure 77). Under the direction of Stephen Foster and active between 2001 and 2012, GW-MATE produced many documents related to groundwater worldwide, with a significant number dealing specifically with urban groundwater issues.

Figure 77 - The Groundwater Management Advisory Team (GW-MATE) (from "What is GW·MATE?↗" n.d. by the Netherlands Bank Water Protection Program).

Today, the GW-MATE website is no longer operative. However, the publications of GW-MATE have been archived on the website of the International Groundwater Resources Assessment Centre (IGRAC↗). The majority of the documents are categorized as:

- GW-MATE Briefing Notes,
- GW-MATE Case Profiles,
- GW-MATE Book Contributions, and
- GW-MATE Strategic Overviews.

Other materials were reorganized as shown in Table 28.

Table 28 - GW-MATE reorganized materials available at IGRAC website.

Reworked material (e.g., journal articles and editorials)	
Foster (2020)	Global Policy Overview of Groundwater in Urban Development—A Tale of 10 Cities!
Foster et al. (2011)	Groundwater use in developing cities: Policy issues arising from current trends
Foster & Hirata (2011)	Groundwater use for urban development: Enhancing benefits and reducing risks
Foster et al. (2018)	Urban groundwater uses in tropical Africa—A key factor in enhancing water security?
GW-MATE case profiles with a significant component of urban groundwater and/or rapid population growth	
Argentina	Integrated approaches to groundwater resources conservation in the Mendoza aquifers
	Mitigation of groundwater drainage problems in the Buenos Aires Conurbation
Brazil	GW use in Metropolitan Fortaleza: Evaluating its strategic importance and potential hazard
Brazil, Uruguay, Paraguay, Argentina	The Guarani Aquifer Initiative
China	Towards a sustainable GW resource use for irrigated groundwater and surface water management
India	A hydrogeologic and socio-economic evaluation of community based GWM: The case of Hivre Bazaar
	Addressing GW depletion in the weathered granitic basement aquifer of Andhra Pradesh
	Confronting the groundwater management challenge in the deccan traps country of Maharashtra
	Groundwater use in Aurangabad
	Lucknow City Groundwater Resource use and strategic planning needs
Kenya	The role of groundwater in the water supply of Greater Nairobi
Mexico	The 'Cotas': progress with stakeholder participation in GW management in Guanajuato
Paraguay	Actual and potential regulatory issues relating to groundwater use in Gran Asuncion
Thailand	Controlling GW abstraction and related environmental degradation in metropolitan Bangkok
	Strengthening capacity in groundwater resources management
Venezuela	Yacambu, Quibor: A project for integrated groundwater and surface water management
Yemen	Rationalizing groundwater resources utilization in the Sana'a basin
GW-MATE series of strategic overviews: Key urban groundwater documents	
Groundwater Governance—Conceptual framework for assessment of provisions and needs	
Conjunctive Use of Groundwater and Surface Water—From spontaneous coping strategy to adaptive resource management	
Urban Groundwater Use Policy—Balancing the benefits and risks in developing nations	

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Finally, readers interested in the hydrogeology of specific global cities should be aware of John Chilton's book *Groundwater in the Urban Environment* (Chilton, 1999), which includes numerous selected city profiles. A list of the cities profiled are included in Table 29.

Table 29 - Cities profiled in *Groundwater in the Urban Environment, Volume 2* (Chilton, 1999).

Aguascalientes	Mexico
Amsterdam	Netherlands
Antwerp	Belgium
Bangalore City	India
Bangkok	Thailand
Belgrade	Serbia
Berlin	Germany
Bucharest	Romania
Calgary	Canada
Caracas	Venezuela
Central Indiana	USA
City of Nablus	Palestine
Dhaka	Bangladesh
Dobrich	Bulgaria
Engels	Russia
Espoo Region	Finland
Florence	Italy
Gaza Strip	Palestine
Goteborg	Sweden
Hull	UK
Irkutsk	Russia
Jakarta and Bandung	Indonesia
Lusaka	Zambia
Madras	India
Nottingham	UK
Oklahoma City	USA
Perth	Australia
Rio Cuarto City, Cordoba	Argentina
Sana'a	Yemen
São Paulo	Brazil
St. Petersburg	Russia
Tangshan City	PR China
Tomsk	Russia
Trelew City	Argentina
West Midlands	UK
Wigan	UK
Zibo City	China

5.5 Exercises Related to Section 5

Exercises related to Section 5 are available at this link↴.

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6 Solutions to the Urban Sustainability Challenge

6.1 The Challenge

The urban sustainability challenge can be described very simply: If cities are to remain sustainable, they require sustainable water supplies. This will be an enormous undertaking, given the explosive rate of growth being experienced in many of the world's large urban areas. The challenge is imminent! In March 2018 (Figure 78), the capacity of Theewaterskloof, Cape Town, South Africa's largest reservoir, reached a record low 11 percent, with the city about to become the world's first major city to run out of water. Day Zero was anxiously anticipated to occur in April 2018, just one month later.

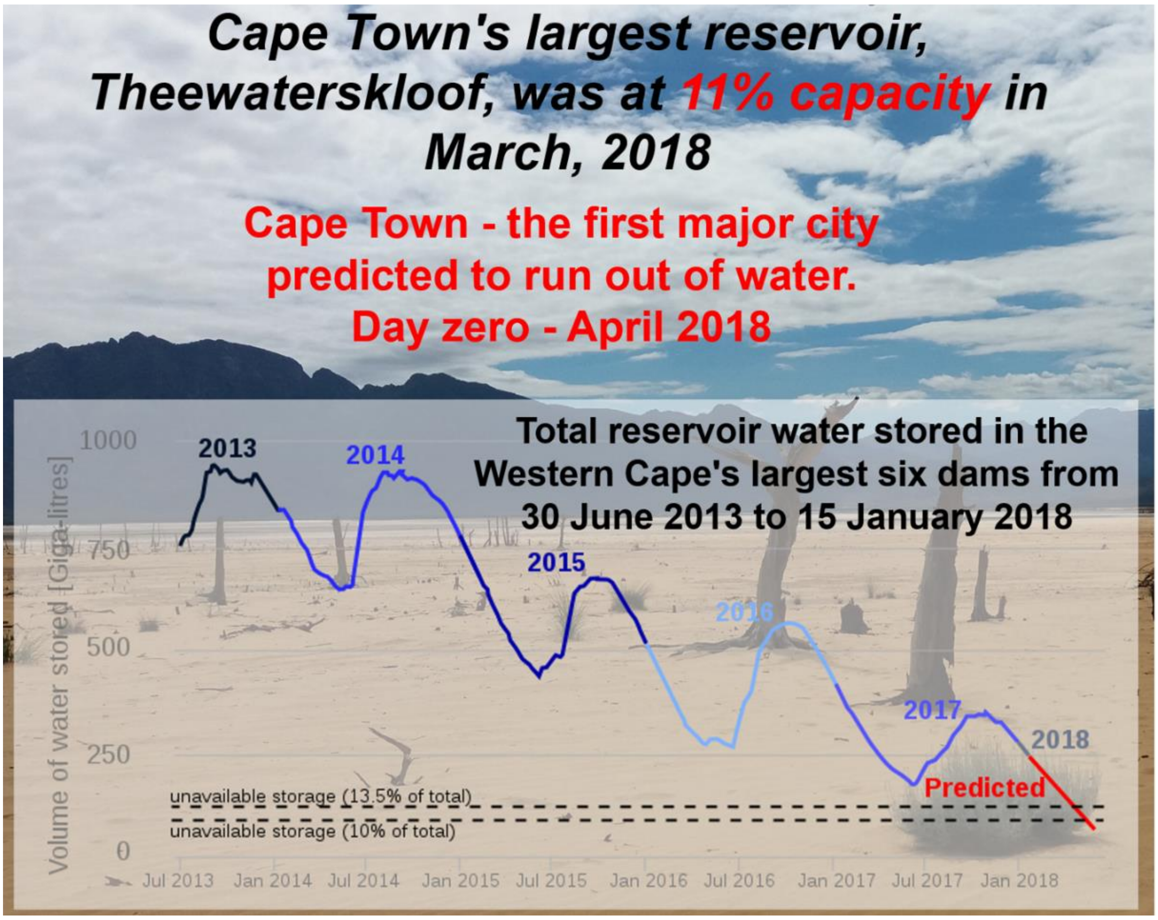

Figure 78 - Cape Town, South Africa, was expected to become the first major city in the world to run out of water, until an unexpected run of wet weather provided at least temporary reprise (graph adapted from Wilson, 2023; photo adapted from Eid & Øhsleybø, 2020).

Due to stringent conservation efforts and fortuitous rainfall events, Cape Town narrowly avoided its potentially momentous fate. However, the situation remains tenuous and water supplies remain insecure. Cape Town is not alone. Numerous other cities face potentially serious water supply shortages (Figure 79).

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Figure 79 - Cape Town, South Africa, is not the only city facing serious water shortages. Numerous other large cities face imminent water supply crises (Source: GW-Project Original, 2023).

Meeting the urban challenge will be complex and difficult. Moreover, it will be complicated by many global uncertainties, not least of which will be the long-term implications of climate change. If there's one certainty, however, it is that groundwater resources will play a pivotal role when it comes to providing solutions. Groundwater may be out of sight and out of mind, but it represents over 97 percent of the world's fresh, readily available water, is widely distributed, and remains relatively well protected from surface sources of pollution. Moreover, it tends to be resilient to climate change, at least in the short term, and can be introduced as needed—just one well at a time—with minimal upfront infrastructure costs. Fortunately, the science of urban groundwater is well advanced, innovative, and science-based; spin-off technologies abound, and there are encouraging signs that the urban sustainability challenge can be overcome.

In this section, some of the more promising solutions to the global urban water crisis are explored. While many of them have already been introduced at least locally—and with considerable success—implementing them globally will remain difficult, especially in countries with limited technical capacity, a poorly trained workforce, and inadequate financial resources for new technologies and associated infrastructure. Such countries also tend to lack urban water governance (as discussed in Section 7, *Urban Groundwater Governance*), which is an essential requirement for achieving solutions to urban water problems.

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6.2 Meeting the Challenge: The Essential Elements

The challenge of meeting the world's increasing demand for urban water supplies is immense. However, the task has been distilled by Sharp (1997) into three essential elements:

- increasing urban water supply,
- reducing water demand (demand management), and
- using available water more efficiently.

Each of these elements is explored in more detail in the sections that follow.

A key consideration in examining all three elements is that water use in cities of middle- and high-income countries regularly exceeds 300 to 400 liters/person/day and, while the normal expectation is that all this water should meet drinking water quality standards, only a few liters are consumed by humans. A few liters more are required to assist with personal hygiene. Most of the remainder is used by city industries or for watering lawns, irrigating parkland, washing cars, flushing toilets, and laundering clothes. If alternative water sources of lower quality water could be directed to meet even a portion of these needs, significantly more potable water would be available to meet the very basic human demand for safe drinking water.

6.2.1 Increasing Urban Water Supply

Increasing the availability of potable water supplies can be achieved in many ways and could be less of a challenge than it may first appear.

The development of new groundwater sources represents a viable opportunity for many developing cities, and technological improvements can further increase drinking water supplies by providing treatment and improving potability. Resource mining (intensive use) has proven to be an effective means of providing additional water, at least for the short term, and recharge management (artificial recharge) is an effective means of augmenting the supply.

New Groundwater Resources and Resource Mining

Finding new groundwater resources may not be an option for all cities facing serious overdraft problems. However, for many cities, it is a potential solution that is too often ignored in favor of more convenient surface water sources. Certainly, the importation of surface water may provide considerable benefits in the short term—but can prove detrimental in the longer term, especially in cities where sewage waste is handled internally (e.g., through septic systems) and not exported in comparable amounts.

Barrett and others (1997) have suggested that urban groundwater is an underutilized resource in the United Kingdom and would certainly prove beneficial in cities afflicted by rising water levels due to leaking reticulation networks. It appears to be a viable option for many cities in Russia, where Zektser and Yazvin (2002) argue that available groundwater resources are often ignored. They estimate that total groundwater

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abstraction in the country, including mine drainage, accounts for just 3.2 percent of the potential safe groundwater yield.

New groundwater supplies may also provide a solution in cities where existing groundwater resources are under stress and groundwater levels are declining. For example, hydrogeologists working in the heavily populated Valley of Mexico, have argued underexploited groundwater reserves remain that, if developed, could significantly improve supply issues in Mexico City. Also, recent work by Ferguson and others (2021) suggests that crustal groundwater volumes (to a depth of 10 km) are significantly greater than once believed, recognizing, of course, that most of these deep reserves are likely to be saline and would require considerable treatment. This would be a significant impediment to their use in low- and middle-income countries.

Time and further study will determine whether optimism for finding new groundwater supplies is justified. However, in the meantime, it should be appreciated that limiting the development of groundwater to the extent that aquifers are naturally replenished, while favored by many as a responsible approach, has never been obligatory. Over-development of available groundwater as a deliberately conceived policy does not always deserve the criticism it attracts. On the contrary, over-exploitation/intensive use of groundwater (also known as resource mining) can promote economic growth and prosperity, while allowing expensive investments (such as dams, long distance pipelines, and desalination facilities) to be postponed. It can generate considerable benefits if consciously planned, properly assessed, and realistically costed, and groundwater production is carefully monitored. A feasible plan is required for alternative water supplies when the groundwater resources are eventually exhausted.

History tells us there is not a prosperous nation in the world that has not benefited at some time from mining groundwater although, in fairness, this is mostly due to an ignorance of the hydrogeology and potential risks than through a judiciously evaluated and carefully planned production strategy. Arie S. Issar is professor emeritus at the J. Blaustein Institute for Desert Research, Ben Gurion University of the Negev, in Israel. Issar has always argued that water has no value unless it is used, and that would apply just as equally to old, non-renewable fossil groundwater underlying North Africa and the Middle East, for example, as it does to renewable groundwater reserves underlying more temperate regions of the world.

Offshore Reserves

Recognizing that most the world's growing cities lie in coastal areas, an interesting development is the work Post and others (2013) published in *Nature* that highlights the global presence of offshore fresh and brackish groundwater reserves called vast meteoric groundwater reserves (VMGR), shown in Figure 80. Examples include Indian River Bay, Delaware, USA, where fresh water can be found in a confined sandy aquifer up to 1 km offshore (Krantz et al., 2004) and the carbonate aquifer system along the eastern seaboard

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of Florida, USA, where fresh water can be found in boreholes up to 100 km from the coast (Johnston, 1983). Using a range of observational data, they estimate that the Earth's continental shelves currently store around 5×10^5 km^3 of groundwater with total dissolved solids (TDS) concentrations of less than 10,000 mg/L

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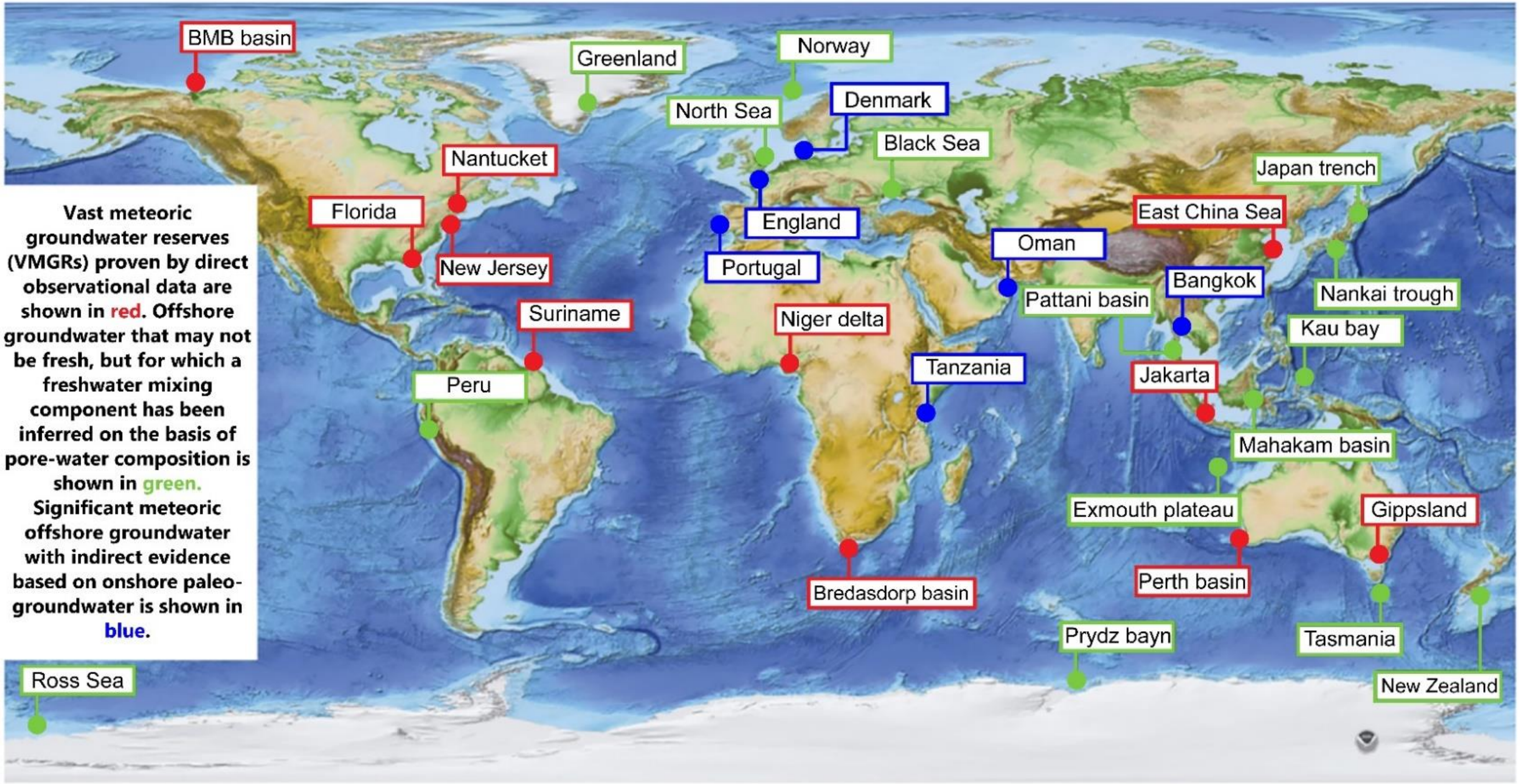

Figure 80 - World map showing established occurrences of fresh and brackish offshore groundwater (modified after Post et al., 2013).

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The offshore reserves are paleo-groundwaters that were emplaced tens, if not hundreds of thousands, of years ago. As such, their development would be considered a form of mining and face the same ethical dilemmas as the depletion of non-renewable fossil groundwater resources found onshore. Their development would also be a technical challenge given that the water essentially underlies a large body of seawater that is denser and would readily replace it or readily mix with it given favorable hydraulic conditions. As a result, offshore groundwater reserves, while promising, would never be capable of resolving the impending global water crisis. However, it would provide an option that could be integrated with other options in an overall long-term water supply strategy.

Aquifers Beneath Lands Reclaimed from the Sea

Another interesting water supply option relates to aquifers created beneath lands reclaimed from the sea. Humans have been reclaiming land from lakes, rivers, and oceans for centuries, usually to create more areas for habitation, industry, and agriculture. The simplest and most common method is infilling, which entails the filling up of an otherwise underwater area with rocks and sediment. Often the sediment is obtained by dredging shallow offshore areas, which performs the dual function of adding usable land while maintaining shipping lanes.

Many hundreds of square kilometers of land are reclaimed every year. China has reclaimed around 12,000 km^2, while the Netherlands and South Korea have added 7,000 km^2 and 1,500 km^2, respectively. The tiny island state of Singapore (Figure 81) has increased its land area by around 24 percent since independence in 1965, adding 135 km^2 to reach a total of 719 km^2. By 2030, the government wants the area of Singapore to increase by a further 7 to 8 percent.

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Figure 81 - Map of Singapore showing the amount of land reclaimed since 1950 (modified after Wells et al., 2019).

Reclaimed land provides considerable benefits in terms of economic development, but it often comes at a serious environmental cost. Mangroves and marine ecosystems are most seriously affected. On the positive side, land reclaimed using good quality sand offers opportunities for the development of artificial aquifers that can store water that would otherwise be lost as runoff to the sea.

The potential for using reclaimed land as a groundwater reservoir is being explored on Jurong Island, Singapore, by a research group that includes the National University of Singapore, Deltares, Department of Soil and Groundwater in the Netherlands, Utrecht University, and the Public Utilities Board (PUB), Singapore (Saha et al., 2016). A field site has been set up, and the hydrogeology and groundwater dynamics of the island (shown on Figure 81) are being studied with comprehensive field data and computer models.

Saltwater intrusion and subsidence risks have been evaluated and are considered manageable. Recharge mechanisms and various extraction strategies were explored using computer simulations and produced very encouraging results. The aquifer system would require active management to avoid seawater intrusion and maintain the integrity of the freshwater lens but yields of fresh water (1 gm/L chloride or less) were deemed to be a viable source of water for industry on the island. In effect, the study demonstrates that a reclaimed island, designed initially for industrial use alone, can serve a dual purpose: space for industry and a groundwater reservoir that can supply that industry. The study serves as an excellent example of what can be achieved with future reclaimed island aquifers.

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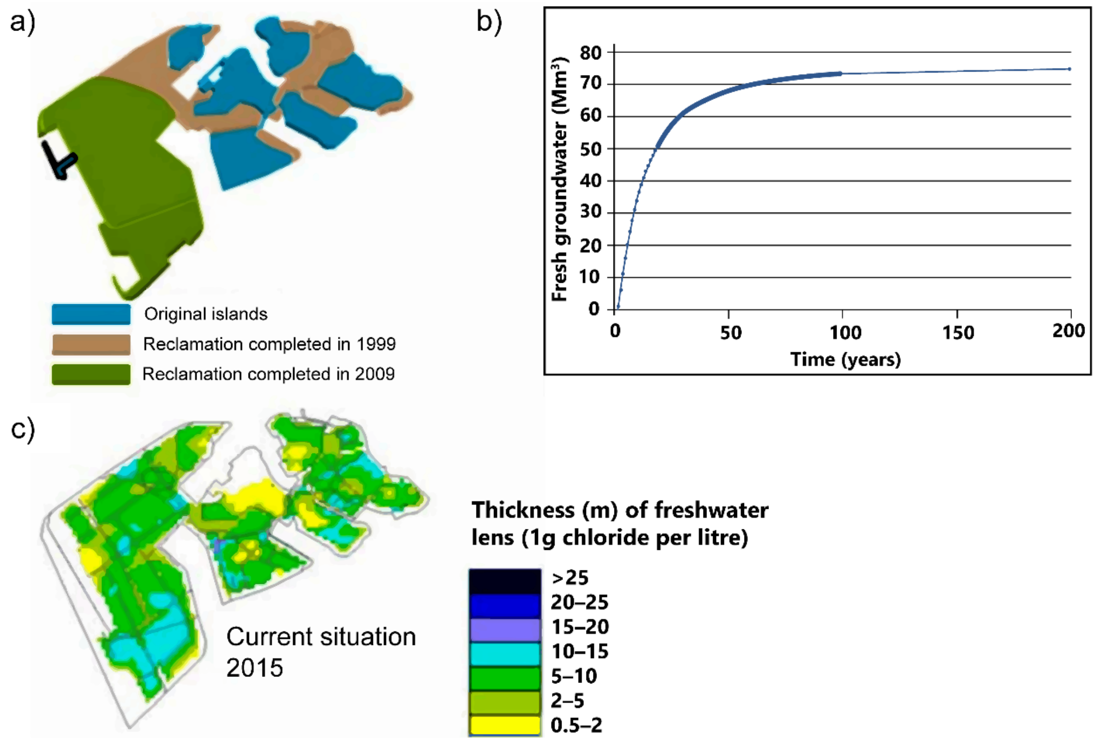

Figure 82 - a) Development of Jurong Island over time; b) Full development of the fresh groundwater lens (1 gm/L chloride) may take well over 50 years (initial groundwater salinity was 18 gm/L); c) Thickness of the freshwater lens on the island in 2015 (locally 10 to 15 m; figure modified after Saha et al., 2016).

Recharge Management Using Artificial Recharge (MAR)

Most overexploited aquifers can benefit considerably from resource augmentation by human intervention in the water cycle. This is known as artificial recharge or managed aquifer recharge (MAR) (Dillon, 2020). The approach is particularly beneficial in urban areas (Howard et al., 2000), where large volumes of additional water are generated as a result of significantly reduced evapotranspiration losses. MAR can utilize this water to augment the groundwater resource while, at the same time, reducing stormwater runoff and the risks of flooding and erosion that may otherwise result.

Typically, resource augmentation is achieved by diverting stormwater runoff into constructed infiltration facilities that release water to the aquifer via gravity drainage (Jacenkow, 1984; Asano, 1986; Li et al., 1987; Watkins, 1997). A comprehensive description of MAR technologies is beyond the scope of this book, but most are fundamentally simple, and some are described by Braune and Sumaya (2021). Often, they comprise little more than a shallow excavated basin into which urban stormwater is introduced (Figure 83). The design challenge is to make certain the aquifer materials are sufficiently permeable, thus ensure the water mound that naturally develops beneath the basin does not reach the base of the basin and thereby become a limiting constraint on infiltration rates (Figure 84).

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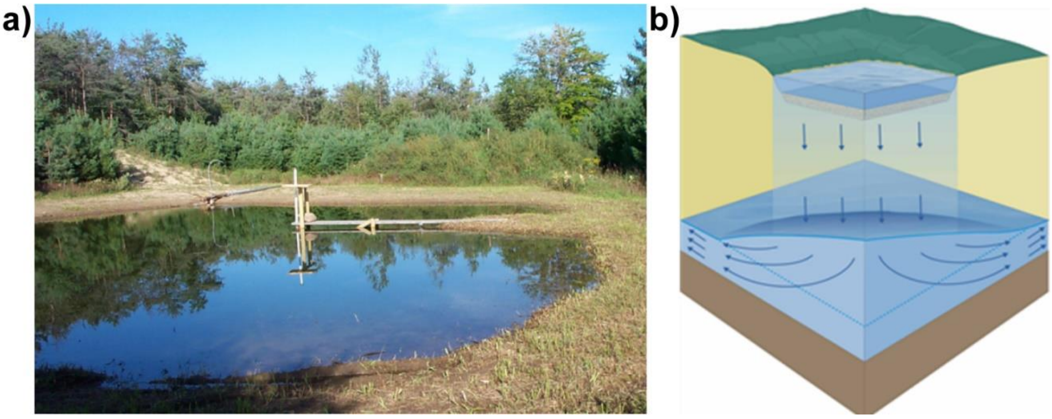

Figure 83 - Rapid infiltration basins (RIBs) have been tested successfully across parts of the Oak Ridges Moraine aquifer in southern Ontario, Canada. a) A recharge pond. b) Schematic of flow from the pond to the underlying water table with a mound developing below the pond (from Howard et al., 2000, 2007; photography by Ken Howard).

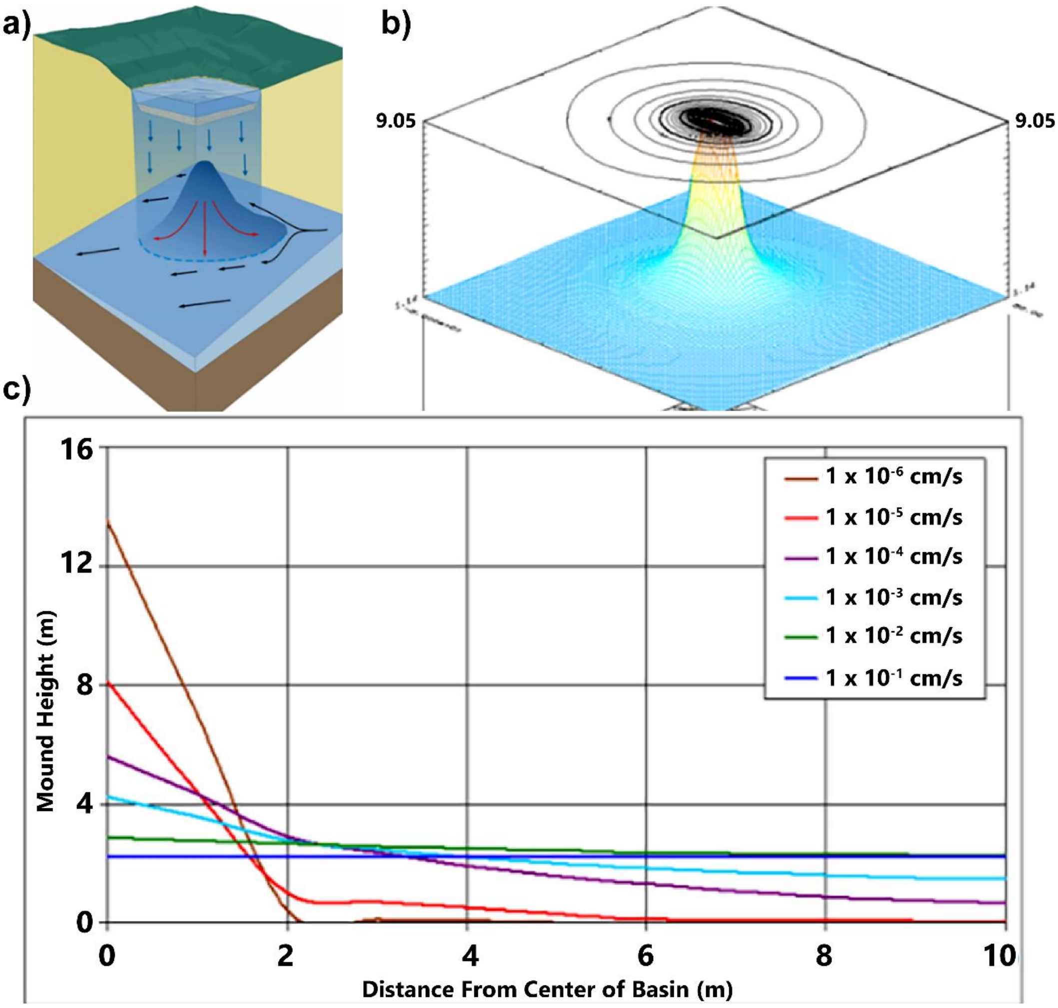

Figure 84 - The height of the recharge mound that develops beneath the basin is a limiting feature. a) Schematic of flow through a recharge mound and into the underlying aquifer. b) Three-dimensional rendering of the mound surface with water elevation projected to the surface as contour lines of equal head. c) Permeable sediments are required to ensure the recharge mound is not so high that it limits infiltration rate (Howard et al, 2007).

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When layers of low permeability materials such as glacial tills are present, infiltration columns can be installed (Figure 85) that allow infiltrated water to bypass these layers. Studies along the Oak Ridges Moraine in Ontario, Canada, (Howard et al., 2000, 2007) have shown these rapid infiltration columns (RICs) function efficiently, even during winter when ice develops across the basin (Figure 86).

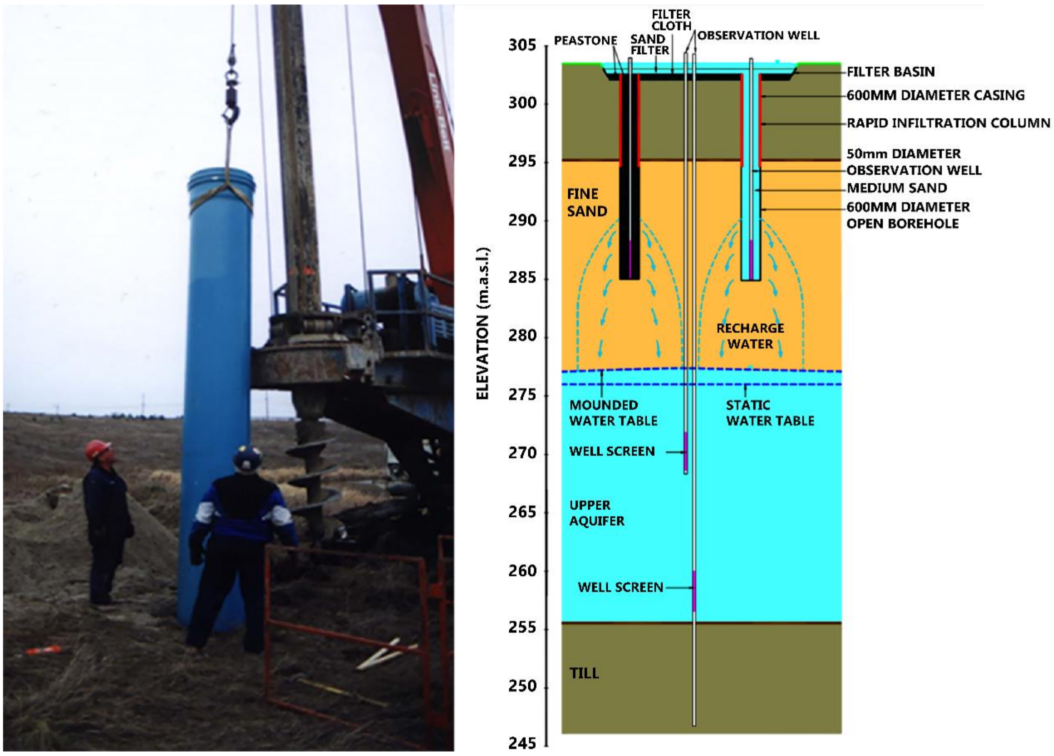

Figure 85 - Rapid infiltration columns (RICs) can be used to allow infiltrated water to bypass low permeability materials such as glacial tills (photography by Ken Howard).

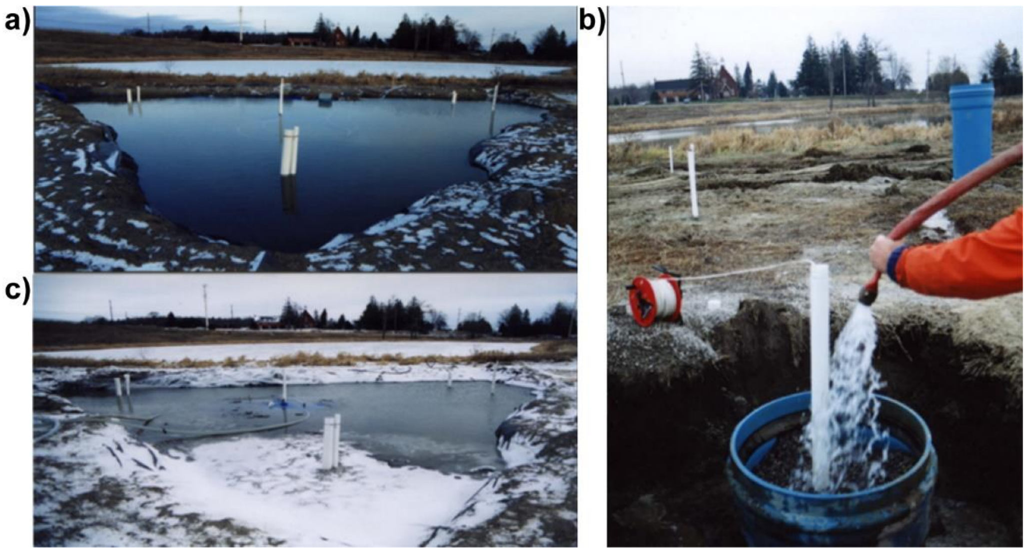

Figure 86 - Rapid infiltration columns (RICs) work well even during winter when ice develops across the basin. a) Unfrozen recharge pond b) water is being added to infiltration column to test its ability to accommodate recharge c) Frozen recharge pond (photography by Ken Howard).

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As an alternative approach, pumping wells can be used to induce groundwater recharge from rivers and streams that carry stormwater. This method (bank filtration) is popular in Europe. Problems are rare but can include clogging of pore spaces with fine sediment, thus reducing infiltration efficiency, and aquifer contamination (Pitt et al., 1996).

Today, the chemical, physical, and biological processes of artificial recharge are well understood and methodologies are well advanced. Pre-treatment of infiltrated water, for example, using oil/water separators, can solve some of the more common pollution issues. Further, the simplest way to avoid clogging of pore spaces is to encourage vegetation growth in the basin, so the network of plant roots maintains adequate soil permeability (Figure 87).

Figure 87 - Recharge basin for the infiltration of stormwater on Long Island, New York, USA. The basin requires very little maintenance since the roots of vegetation maintain sufficient soil permeability (photography by Ken Howard).

Finally, the water used for the artificial recharge of aquifers is not limited to stormwater runoff. Currently available technologies allow wastewater to be treated to drinking water quality standards and—while many government agencies are hesitant to allow this water to be used directly for supply—it is an ideal candidate for the "polishing effects" of artificial recharge. Many strategies are available but most use infiltration lagoons (Figure 88). One option discussed in 1971 by Schicht (Figure 89) and tested in El Paso, Texas, USA (Sharp, 1997) involves the injection of tertiary-treated sewage directly into the aquifer.

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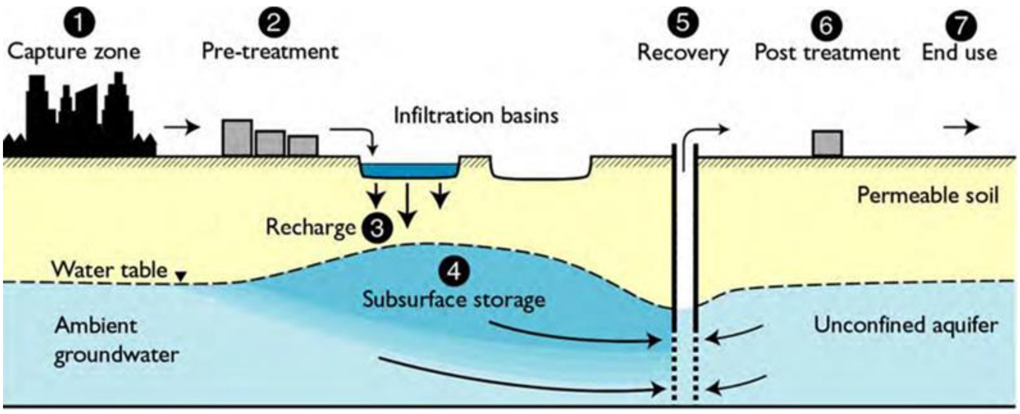
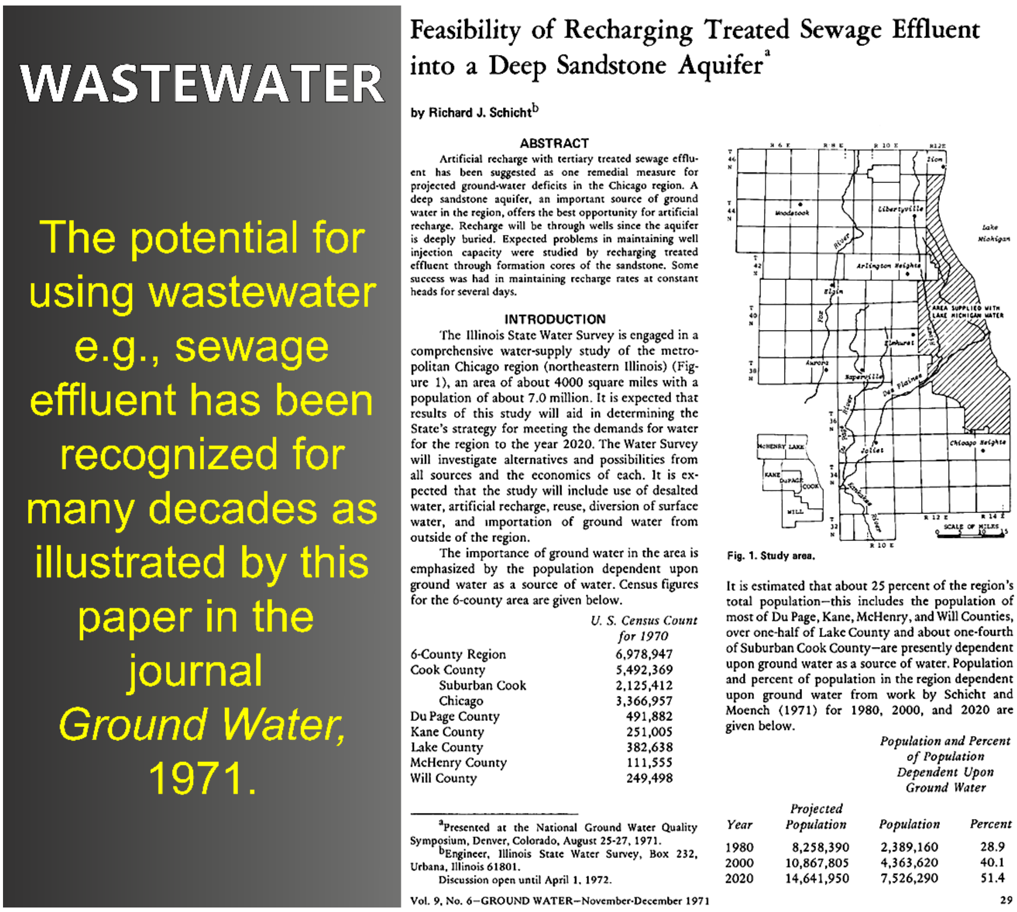


Figure 88 - Schematic showing how wastewater captured from a city can be pre-treated and infiltrated for subsequent recovery and post treatment prior to its eventual use (,from Natural Resource Management Ministerial Council, Environment Protection and Heritage Council, & National Health and Medical Research Council, 2009).



Feasibility of Recharging Treated Sewage Effluent into a Deep Sandstone Aquifer[a]

by Richard J. Schicht[b]

ABSTRACT

Artificial recharge with tertiary treated sewage effluent has been suggested as one remedial measure for projected ground-water deficits in the Chicago region. A deep sandstone aquifer, an important source of ground water in the region, offers the best opportunity for artificial recharge. Recharge will be through wells since the aquifer is deeply buried. Expected problems in maintaining well injection capacity were studied by recharging treated effluent through formation cores of the sandstone. Some success was had in maintaining recharge rates at constant heads for several days.

INTRODUCTION

The Illinois State Water Survey is engaged in a comprehensive water-supply study of the metropolitan Chicago region (northeastern Illinois) (Figure 1), an area of about 4000 square miles with a population of about 7.0 million. It is expected that results of this study will aid in determining the State's strategy for meeting the demands for water for the region to the year 2020. The Water Survey will investigate alternatives and possibilities from all sources and the economics of each. It is expected that the study will include use of desalted water, artificial recharge, reuse, diversion of surface water, and importation of ground water from outside of the region.

The importance of ground water in the area is emphasized by the population dependent upon ground water as a source of water. Census figures for the 6-county area are given below.

	U. S. Census Count for 1970
6-County Region	6,978,947
Cook County	5,492,369
Suburban Cook	2,125,412
Chicago	3,366,957
Du Page County	491,882
Kane County	251,005
Lake County	382,638
McHenry County	111,555
Will County	249,498

[a]Presented at the National Ground Water Quality Symposium, Denver, Colorado, August 25-27, 1971.
[b]Engineer, Illinois State Water Survey, Box 232, Urbana, Illinois 61801.
Discussion open until April 1, 1972.

Fig. 1. Study area.

It is estimated that about 25 percent of the region's total population—this includes the population of most of Du Page, Kane, McHenry, and Will Counties, over one-half of Lake County and about one-fourth of Suburban Cook County—are presently dependent upon ground water as a source of water. Population and percent of population in the region dependent upon ground water from work by Schicht and Moench (1971) for 1980, 2000, and 2020 are given below.

Year	*Projected Population*	*Population and Percent of Population Dependent Upon Ground Water* — *Population*	*Percent*
1980	8,258,390	2,389,160	28.9
2000	10,867,805	4,363,620	40.1
2020	14,641,950	7,526,290	51.4

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Figure 89 - Research paper authored by Schicht (1971) shows that interest in the use of sewage effluent for recharging aquifers dates back half a century (from Natural Resource Management Ministerial Council, Environment Protection and Heritage Council, & National Health and Medical Research Council, 2009).

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Treatment and Desalination

When all else fails, a shortage of good quality water can often be solved by some form of water treatment. Water that is unfit for consumption due to the presence of bacteria can be readily made potable using a wide variety of methods; chlorination, ozonation, reverse osmosis, and the use of ultraviolet radiation are among the most common (Supong et al., 2017). In areas where the water supply is seriously contaminated, small-scale water treatment units can be installed in individual houses and apartments (Limaye, 1997) and used to produce the few liters of water required daily for potable use.

Bacterial contamination is most frequently encountered in surface water but is also associated with springs and shallow wells. A more common problem with deeper groundwater is high levels of total dissolved solids (TDS), that is, elevated water salinity. Fortunately, modern desalination technologies can be used to produce potable water from water of virtually any salinity. While energy requirements for desalination can prove expensive, advances in technology and improvements in efficiency have significantly reduced costs in recent years. As a result, desalination now represents an economically viable means of supplying good quality potable water for the urban masses, especially when it is used to treat slightly to moderately saline (brackish) water—that is, not seawater, which requires considerably more time and energy—and provide sufficient water for drinking water requirements only, recognizing that the vast majority of water used in urban areas does not need to meet potable water quality standards.

In fact, the universal availability of appropriately treated water targeted for potable use exclusively would likely revolutionize the way groundwater is managed, protected, and utilized in cities. Marginally saline groundwater resources that would once have been regarded as unsuitable for development and use due to concerns over meeting potability standards would suddenly gain popularity as the most cost-effective source of water for non-potable purposes. It would mean, for example, that rigorous land use restrictions imposed to protect and guarantee the potable quality of underlying groundwater resources would become virtually unnecessary.

Finally, an alternative means of making beneficial use of slightly "inferior" quality groundwater is to blend it with good-quality water in such proportions that the water meets water quality guidelines that are appropriate for its intended use. Ideally, this would require the establishment of an external wellfield, conveniently located and well protected from sources of contamination, that would provide a reliable source of high-quality groundwater. Depending on the quality of the inferior water with which the high-quality is mixed, desalination and/or treatment may not be required.

6.2.2 Reducing Water Demand

In some parts of the world, finding and/or establishing new groundwater resources, improving available groundwater resources using MAR, and treating/desalinizing poor quality water are not viable options. As populations rise and

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water needs increase, the only alternative is to introduce measures that can reduce or at least temper demand.

Reducing or tempering the demand for water can be achieved through a wide variety of measures. According to Sharp (1997), they typically include

- water conservation (e.g., finding ways to use less water to perform the same task);
- controls on accessibility such as limit on the number, depth, and yields of wells through issuance of well construction and development permits and limits on the availability of municipal supplies to certain periods of the day; and
- cost structuring such as water metering and tariffs.

Regardless of the approach taken, experience in some parts of the world has shown user education should always be the starting point—an informed and knowledgeable public can become an accepting public. Education in good water management practices and the critical need for such practices need to be targeted at all levels of government, industry, and the population at large, recognizing that the commitment and cooperation of millions of city dwellers are required if water problems are to be alleviated.

Water Conservation

Water conservation measures can be implemented at all stages in the distribution network. At the consumer level, low-flow plumbing fixtures (showerheads, toilets, and faucets) have proved to be highly successful at reducing water demand in high- and middle-income countries. Similarly, bans on washing cars and watering lawns, imposed by municipalities during times of water stress, often enjoy high levels of compliance—albeit likely due to the very public nature of what would be viewed as anti-social activities. For example, the City of Los Angeles reduced its water use by 30 percent during a drought in 1991 by requiring residents to conserve water and, despite a significant population increase, managed to maintain a 1970s level of water use well into the new millennium.

Conservation measures can also be implemented at the municipal level. For example, rates of groundwater pumping can be reduced significantly in many cities if leakage from pressurized water distribution networks (commonly in the range of 10 to 30 percent, as discussed in Section 3) is significantly reduced or eliminated (Jones, 1997). Unfortunately, this requires a significant investment in system maintenance and, in many cases, requires replacement of water mains. When funds for maintenance are unavailable, a lowering of mains water pressures can provide some respite. However, there may be limits to which this can be done. Throughout much of the United Kingdom, there is a legal obligation to supply a water pressure of at least 7 meters head ≈0.7 bar) at the boundary stop tap.

In 1994, the National Rivers Authority in the United Kingdom demanded that water companies achieve economic levels of leakage and metering before it would allow any new well pumping permits to be issued. In addition, the consumer protection agency

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(the Office of Water Services) obliged water companies to publish data on leakage annually. Partly in response, South West Water, one of the worst offenders, implemented a program aimed at reducing rates of leakage from 32 percent to just 20 percent over a period of six years. As impressive as this may sound, such a level of achievement pales in comparison to Bielefeld, Germany, which in the early 2000s reported losses of only 5 percent while engaging 32 teams of workers to replace leaking pipes at a rate of 2percent per year—four times the average in the United Kingdom.

Controls on Water Accessibility

In many low-income countries, the per capita usage of water is already very low; there are few opportunities for significant savings to be made at the domestic level by adopting water conservation practices unless major incentives for reducing water use are introduced. A very moderate reduction can be achieved by limiting household accessibility to municipal water to just a few hours each morning and evening, as is commonly enforced in India (Limaye, 1997). However, this does little, if anything, to regulate usage during those times water is available.

At the communal and municipal level, demands on the aquifer can also be reduced by limiting the amount of water that can be pumped. According to Morris and others (1997), this is best achieved by imposing stringent controls on the construction of new water wells (e.g., by rationing the issuance of drilling permits) as opposed to simply restricting pumping rates via water-taking permits for wells already in operation. Lack of compliance is the primary problem. Many argue that it is pointless to regulate water usage if laws are not adequately enforced and violators are not prosecuted. According to Limaye (1997), the greater the number of rules and regulations, then the greater the level of illegal activity including corruption. In effect, current problems with urban groundwater management will not be resolved until governments agree to work with groundwater users and simply refrain from trying to regulate and control them. In the poorest countries, the needs and interests of stakeholders, including community users, are widely ignored.

Water Pricing - Cost Structuring

One of the most effective means of exerting control over demand is the financial disincentive that results from increased water tariffs. As discussed by Morris and others (1997), this can be achieved at the wellhead by imposing realistic charges for raw water by taking account of one or more of the following:

- the full costs incurred by the regulatory body for administering resource development and for evaluating, monitoring, and managing the groundwater resource;
- the potential cost of providing alternative raw water supplies to users in the event the source fails and goes out of commission; and
- the full cost of environmental impacts that will likely ensue due to the taking of groundwater.

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Pricing at the wellhead provides a strong incentive for reducing both demand and water-mains leakage. It may do little, however, to encourage water conservation at the consumer level unless the charges imposed at wellhead can be passed on to these users equitably according to the actual amounts used. This requires individual metering, which is a proven means of reducing wastage. For example, in Germany, water meters have controlled demand effectively for decades. In an experiment in Yorkshire, United Kingdom, during the 1990s, the introduction of meters in 700 homes saw a 29 percent reduction in water bills. A 20 percent reduction was observed during a much larger United Kingdom study conducted on the Isle of Wight.

Unfortunately, many of the world's largest cities presently lack the infrastructure (e.g., household meters and data collection facilities) required to exert significant control over water use by pricing at the domestic level. Even where domestic metering is present, the administrative burden of collecting fees is often challenging, and recovery rates tend to be low. Metering may also prove counter-productive if tariffs are greater than the user can realistically afford. Excessive water costs will tend to promote illegal activity or simply lead to poor sanitation and impacts to human health.

6.2.3 Using Available Water More Efficiently

Eventually, there become limits to which urban supply can be increased and urban water demand reduced. Delivering sustainable supplies of groundwater to rapidly growing cities will be unattainable unless presently available water is used much more efficiently. This means impacts on groundwater quality must be minimized and available water reserves need to be managed to maximize yields with minimal wastage.

Groundwater that bypasses a wellfield and enters a stream or the sea is not necessarily "wasted." Groundwater resource management is always a tradeoff. Every single drop of water removed from an aquifer by pumping is water that, if left undisturbed, would be destined to go somewhere else and perform some other hydrologic function such as sustain stream flows and/or provide support for groundwater dependent ecosystems. In all cases, the potential environmental benefits of the groundwater need to be weighed carefully against the social and economic benefits derived by pumping and removing it. Difficult decisions need to be made (Figure 90).

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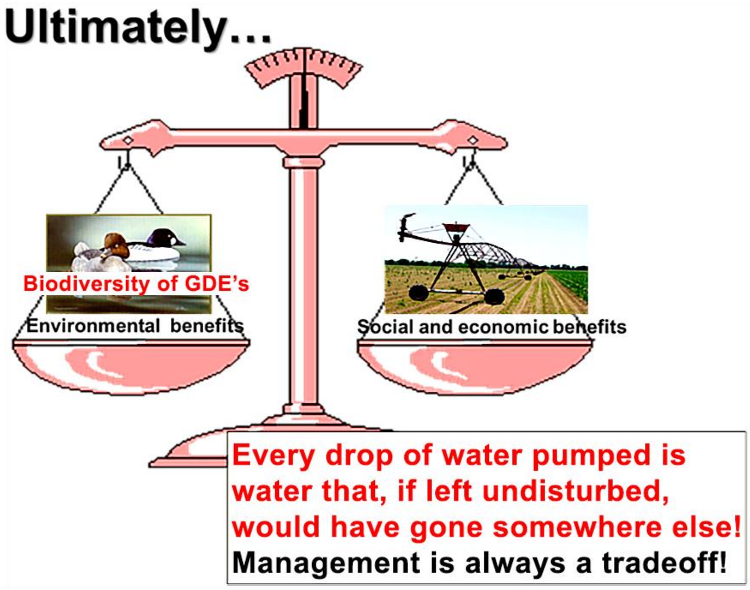

Figure 90 - Aquifer management is always a tradeoff. Decision making needs to balance the social and economic benefits of pumping with the potential environmental benefits of allowing the groundwater to fulfill its normal hydrologic function (from personal files of Ken Howard).

Groundwater resource protection[3] and sound aquifer-management decision making are key to improving water use efficiency, but that is far more easily said than done. In the future, demand for safe water supplies will continue to be met from both groundwater and surface water sources. It is therefore essential that groundwater management and protection strategies consider the entire water cycle and incorporate opportunities for optimizing the combined development of ground and surface water using the principles of conjunctive use (Paling, 1984). In effect, there is an urgent need to maximize potential benefits by integrating ground and surface water resources into a single management plan that also incorporates water quality protection as an essential element of that plan.

Sound management of urban water resources must be underpinned by good science that includes a full, quantitative understanding of the resource setting. It is the role of the scientist to address perceived problems and formulate solutions that can be used by those managing the resource (Barber, 1997). Best management practices must be allowed to evolve by gradually assimilating developments in science and technology. Fortunately, the science of urban hydrogeology, while young, is already well advanced and provides a strong foundation for the development of appropriate resource management strategies. These strategies, in turn, can be most effective when supported by two fundamental pillars:

1. groundwater resource protection that, when used effectively, can significantly reduce the extent to which potential sources of groundwater contamination can

3 While the term *groundwater resource protection* is considered by some to include both water quality and water quantity components, its use here is limited to water quality. Used in this way, groundwater resource protection would be an integral part of an overall resource management plan that includes both quality and quantity.

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degrade and compromise the viability and value of the groundwater resource; and

2. resource modeling that utilizes numerical simulation models to optimize the development of the groundwater resource.

Resource modeling can be useful in many ways. It can help

- quantify aquifer recharge and other water budget components,
- establish the aquifer's sustainable yield,
- assess potential impacts of pumping on receiving water bodies and groundwater dependent ecosystems,
- identify the best sites for installing wells and predict their likely yields,
- identify options for conjunctive use of surface water and groundwater,
- allow the testing and evaluation of alternative wellfield pumping strategies, and
- enable potential threats to the resource (quantity and quality) to be assessed.

6.3 Groundwater Resource Protection

Urban aquifers will never be sustainable unless efforts are made to control activities which, through their nature or intensity, may threaten groundwater quality. The purpose of groundwater resource protection is to maximize the availability of fresh groundwater by minimizing the extent to which groundwater may become degraded.

One valuable option is to adopt new technologies that significantly reduce the extent to which potential sources of contamination are released into the environment. In urban areas, for example, municipal landfills and storage tanks containing petrochemicals such as gasoline are well established threats to local aquifers with numerous cases of groundwater pollution reported. Today, modern landfill design that includes low-permeability liners, drainage blankets, leachate collection and treatment facilities, as well as strict controls over water levels and water/leachate flow directions using pumping and gravity drainage tunnels can provide significant groundwater protection (Figure 91).

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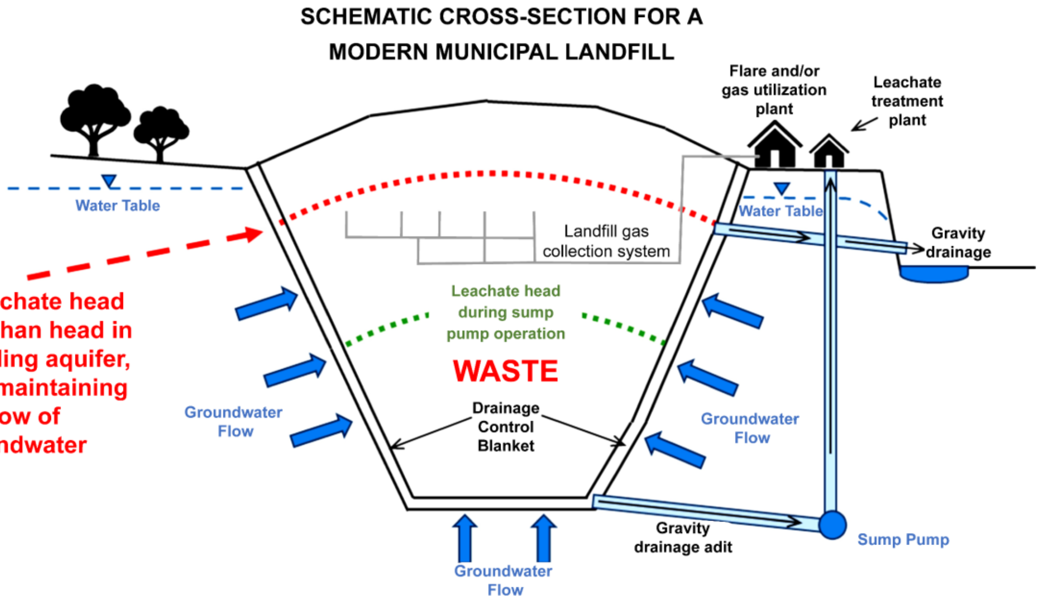

Figure 91 - Modern municipal landfills afford considerable groundwater protection by featuring low-permeability liners, drainage blankets, leachate treatment, and controls over water levels and water/leachate flow directions using pumping and gravity drainage tunnels (figure by Ken Howard).

Similarly, gasoline tanks that include double walls and continuous leak monitoring represent a considerable improvement over bygone years (Figure 92).

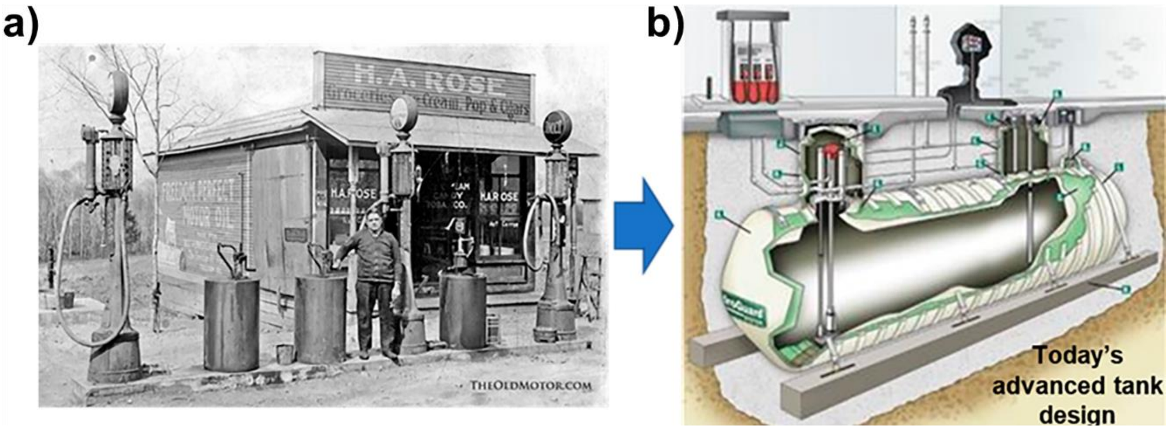

Figure 92 - Relative to a) early gasoline tank design, b) modern gasoline tank design includes double walls and continuous leak monitoring, thereby significantly reducing contamination risk in the event of tank-wall failure (a) from theoldmotor.com↗; b) from Containment Solutions, Inc., 2021).

Unfortunately, these progressive technologies come with a cost, and uptake tends to be slow, particularly in low- and middle-income countries that can ill afford the necessary financial investments. In the meantime, approaches to groundwater protection tend to focus their attention on controlling/regulating potential sources of contamination in areas that pose the greatest threat to groundwater. To this effect, the methodologies adopted normally fall into one of two categories (Howard, 1987):

1. application of standards of practice, and
2. application of standards of performance.

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The standards of practice approach to groundwater protection is, by far, the more common. It involves the development of procedural guidelines, regulations, and/or directions that, when followed, will significantly reduce the likelihood that groundwater will be degraded by potential sources of contamination. As described in some detail below, many methods are available—some being more rigorous and scientifically sound than others. In general, standards of practice are highly prescriptive, easy to follow, and can be readily enforced. The downside is that standards of practice make no guarantees about the eventual outcome and, in fact, say nothing whatsoever about the expected water quality.

The standards of performance approach, in contrast, explicitly acknowledges that groundwater quality degradation will be an inevitable consequence of many activities, and stipulates very clearly the extent to which degradation would be deemed acceptable, that is, it sets a water quality target, usually at a specific location, thereby putting the onus on those proposing any activity that can change groundwater quality (e.g., building a highway, urban sub-division, or landfill) to meet that target and maintain it for all time. While this provides the proponent of the activity with considerable flexibility with regards to the design of the activity (including choice of available technologies), it obliges the proponent to perform all the work necessary—including thorough hydrogeological investigations—to ensure targets can be met. The proponent is also responsible for installing appropriate monitoring systems and developing contingency programs that could be implemented if problems arise.

6.3.1 Standards of Practice

Approaches to groundwater resource protection using standards of practice are many and varied. The simplest involve establishing minimum separation distances (a setback or buffer zone) between a potential source of pollution and an aquifer or well. The actual distance selected is often largely arbitrary and takes no consideration of the local hydrogeology including aquifer parameters and prevailing groundwater flow directions. For example, the province of British Columbia in Canada requires that the footprint of municipal landfills be sited a minimum distance of 300 m from a water supply well or water supply intake and a minimum 500 m from municipal or other high-capacity water supply wells. Similarly, the neighboring province of Alberta requires a 450-meter setback from a landfill for any water well providing water for human consumption (Alberta, 2023).

No scientific justification is given for any of these distances, but they are easy to follow and simple to regulate. Significantly, a recent study conducted in China (Xiang et al., 2019) examined landfill isolation distances in the context of pollution risk, landfill design, aquifer characteristics, and parameter uncertainty, to find that the required setbacks varied greatly, and ranged from 106 m to 5.46 km in sand aquifers, 292 m to 13.5 km in gravel aquifers, and 2.4 to 58.7 km in coarse gravel aquifers.

Commonly recommended separation distances of a water well to septic tanks, livestock yards, gasoline tanks, pesticide and fertilizer sources, and manure stacks are

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shown in Figure 93. Table 30 shows the required minimum horizontal separation distances (setbacks) between wells and sewage systems in Ontario, Canada. Many other jurisdictions have similar regulations. Typically, a minimum separation or setback of 10 m is considered sufficient when wells or springs are located near sources of fecal contamination. However, some regulations suggest 5 m is sufficient; others recommend 30 m.

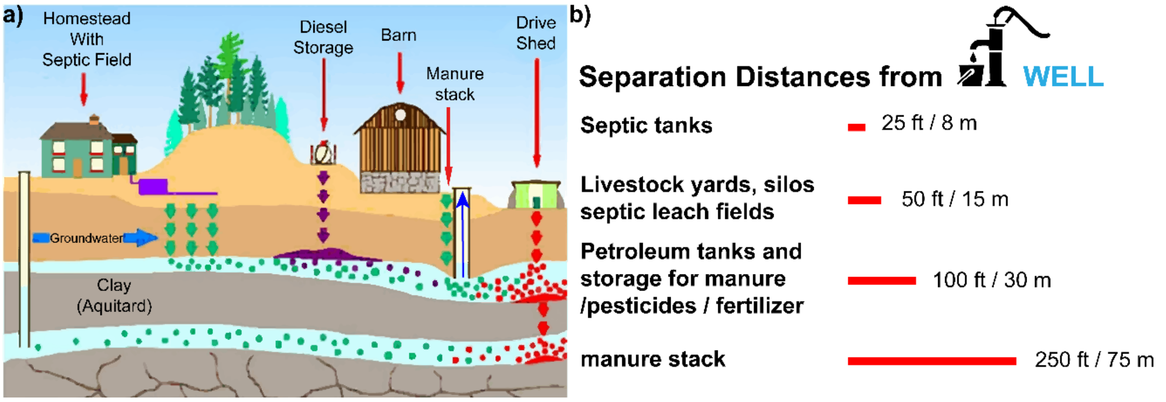

Figure 93 - The simplest standards of practice approach involves establishing minimum separation distances (setbacks) between a potential contaminant source and the water supply such as a well or spring (from a) Ontario Ministry of Environment, Conservation, and Parks, 2023; b) figure by Ken Howard).

Table 30 - Ontario, Canada: Minimum horizontal separation distances (setbacks) between wells and existing sewage systems or other sources of contaminants (adapted from Ontario Ministry of Environment and Energy, 2019).

Sewage system	Well with watertight casing to a depth of ≥ 6 m (19.7 ft)		Any other well	
Earth pit privy	15 m	(50 ft)	30 m	(100 ft)
Privy vault, pail privy	10 m	(33 ft)	15 m	(50 ft)
Greywater system	10 m	(33 ft)	15 m	(50 ft)
Cesspool	30 m	(100 ft)	60 m	(200 ft)
Treatment units (such as a septic tank)	15 m	(50 ft)	15 m	(50 ft)
Distribution pipe in a leaching or filter bed	15 m	(50 ft)	30 m	(100 ft)
Holding tank	15 m	(50 ft)	15 m	(50 ft)
Other sources of contaminants	15 m	(50 ft)	30 m	(100 ft)

While it is easy to criticize the standards of practice approach, especially when applied in such a simplistic way, any regulation is often better than none. This is particularly true in many low- and middle-income countries where sources of drinking water (wells and springs) often lie in very close proximity to sources of fecal bacteria (latrines and toilets), as shown in Figure 94 and Figure 95.

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Figure 94 - Example of groundwater pollution in Lusaka, Zambia, where the pit latrine in the background is contaminating the shallow well in the foreground with pathogens and nitrate (photography by Kennedy Mayumbelo).

Figure 95 - A study in northern India found evidence of fecal contamination in 18 percent of the water sources; 91 percent of the contaminated water sources were situated within 10 m of a recently installed toilet (Pulla, 2016).

Densely populated urban areas are especially at risk. The basic practice of ensuring that wells and springs are located a minimum of 10 m from sources of fecal contamination (ideally, a minimum of 30 m where conditions permit) has brought immense health benefits for both urban and rural populations throughout the world. Springs can

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additionally be protected by providing a catchment wall, delivery pipe and adequate drainage (Figure 96).

Figure 96 - Emuyokha Spring is a “so-called” protected water source in Ijinha Community, western Kenya (from The Water Project, 2023).

A more sophisticated standards of practice approach to groundwater protection involves the assessment of aquifer vulnerability (or perceived threat), thus providing a basis for controlling or regulating potential sources of contamination. The concept of aquifer vulnerability was first introduced by Margat (1968) in consideration of the risk to groundwater that may be imposed by surface and near-surface pollutant sources. Since that time, there have been countless attempts to provide a definition for aquifer vulnerability (Vrba & Zaporozec, 1994), and numerous methods have been devised for assessing vulnerability in a reasonably consistent, quantitative, and, ultimately, useful manner (Civita, 1993).

Some workers have argued that aquifer vulnerability should be measured purely in terms of the natural, fundamental properties of the groundwater system, characteristics that can be easily incorporated into maps; others (e.g., Foster, 1987) have argued that aquifer vulnerability is best considered in terms of pollution risk and should additionally consider the characteristics of the contaminant. Palmquist (1991), for example, correctly noted that in the absence of a contaminant source, even the most susceptible aquifer cannot be considered “threatened” or “at risk.”

Today, the term aquifer vulnerability tends to be defined in terms of the intrinsic properties of the groundwater system, consistent with the recommendation of Vrba and Zaporozec (1994). Used in this way, the term is sometimes referred to as intrinsic vulnerability to distinguish it from specific vulnerability, a term that is generally preferred for situations where the role and behavior of a specific contaminant is explicitly

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incorporated in the analysis. As an example, Figure 97 shows a specific groundwater nitrate vulnerability map for England and Wales.

Figure 97 - A specific groundwater nitrate vulnerability map for England and Wales that incorporates land use patterns (thus fertilizer and manure applications) for the cropping year 1994/95). The authors prefer to call this map a groundwater nitrate risk map, saving the term specific vulnerability for maps they developed using a uniform rate of nitrate application (100 kg N/ha). Aquifer vulnerability classes range from 1 (most vulnerable) to 9 (least vulnerable) and tend to be most strongly influenced by the hydrogeological properties of the types of rock present (modified after Lake et al., 2003).

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The assessment of aquifer vulnerability requires that the land above an aquifer be classified or zoned according to a predefined formula that, depending on the method, can involve a wide range of geological, hydrological, and hydrogeological factors. Typically, assessments produce a numerical index that provides a relative measure of contamination potential or risk. Assessment approaches are many and varied, but most have been developed to take full advantage of GIS (geographic information system) mapping techniques. Some examples include

- GOD (Groundwater occurrence, Overall lithology of aquifer or aquitard, and Depth to groundwater table) (Foster, 1987),
- AVI (Aquifer Vulnerability Index) (van Stemproot et al., 1993), and
- K-means cluster analysis (Javadi et al., 2017).

Over the years, DRASTIC (Depth to groundwater, Recharge rate, Aquifer type, Soil type, Topography, Impact vadose zone, Conductivity (hydraulic) aquifer) seems to have remained the most popular and widely used technique. It is a relatively simple indexing method developed jointly by the US Environmental Protection Agency (USEPA) and the National Water Well Association (NWWA) (Aller et al., 1985, 1987). The main advantages of DRASTIC are its simplicity, ease of use, and general reliability (Baalousha, 2006). It also requires a relatively limited dataset that is frequently available for most aquifers (Wang et al., 2012) and is highly compatible with GIS (e.g., Al-Adamat et al., 2003; Panagopoulos et al., 2006; Moghaddam et al., 2010). Examples of its application include

- Wen and others (2009), who used DRASTIC to classify the shallow aquifer of the Zhangye Basin in Northwest China into low, middle, and high vulnerability risk zones, thus completing an essential first step for resource management decision making;
- Pathak and others (2009), who developed an aquifer vulnerability map for the Kathmandu Valley of Nepal using a GIS-based DRASTIC model; and
- Kachi and others (2007), who applied DRASTIC to evaluate the vulnerability of the alluvial aquifer of Tebessa-Morsott, a vast closed depression in eastern Algeria.

DRASTIC essentially involves the calculation of a risk index by systematically weighting the contributions from seven environmental data sources:

D = Depth to the water table

R = Net recharge

A = Aquifer media

S = Soil media

T = Topography

I = Impact of the vadose zone media

C = Conductivity (hydraulic) of the aquifer

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The index is calculated according to Equation (1).

$$V = \sum_{i=1}^{7} W_i R_i \tag{1}$$

where:

V = Vulnerability index value

W_i = Weighting coefficient for parameter *i*

R_i = Rating value for parameter *i*

The weighting coefficients range from 1 to 5 depending on their relative importance. The depth to the water table and the nature of the vadose zone material are considered to be the most influential parameters and are assigned the highest weighting coefficient of 5. Net recharge has a weighting coefficient of 4, while the hydraulic conductivity of the aquifer and the nature of the aquifer media have weighting coefficients of 3. The lowest weighting coefficients (1 and 2) are assigned to the topography and soil media parameters, respectively. In short, the expression is as shown in Equation (2).

$$5D + 4R + 3A + 2S + T + 5I + 3C \tag{2}$$

The calculated index provides a relative measure of potential vulnerability of the aquifer to contamination, with higher index values indicating greater vulnerability than lower values (Liu et al., 2007; Wen et al., 2009). Despite the apparent universality of the DRASTIC approach, the method has limitations, the most serious of which is the subjectivity inherent in its factor rating scales and weighting coefficients (Babiker et al., 2005; Saidi et al., 2011). This problem can be amplified when one or more key parameters lack adequate field data such that, in some cases, the results of a DRASTIC analysis can be ambiguous and open to more than one interpretation (Lim et al., 2009). Concerns have also been expressed over DRASTIC’s limited ability to handle hydrogeological environments such as karst, where the aquifer vulnerability is dominated by characteristics of the groundwater system that are poorly represented in the basic DRASTIC model (Goldscheider, 2005; Polemio et al., 2009).

As a result, many researchers have tried to improve DRASTIC over the years (Panagopoulos et al., 2006; Nobre et al., 2007; Mohammadi et al., 2009; Saidi et al., 2011; Javadi et al., 2011; Neshat et al., 2014), modifying its algorithms, adding variables, and employing analytical techniques such as AHP (analytic hierarchy process) (Thirumalaivasan et al., 2003). OREADIC is one successful adaptation used by Qian and others (2011) to perform a vulnerability assessment of aquifers underlying the Yinchuan Plain in northwest China, a heavily populated region that relies on groundwater for its potable supply and greatly needs a reliable plan for resource management and protection (Figure 98).

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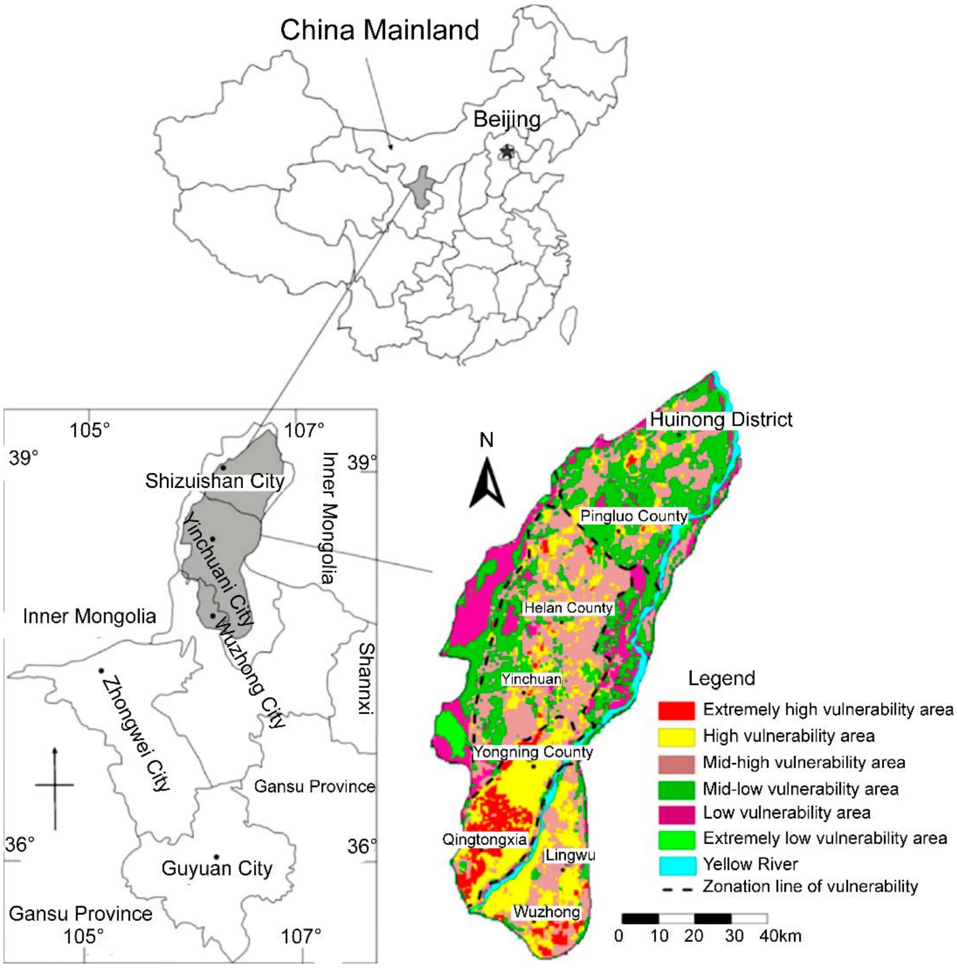

Figure 98 - Intrinsic aquifer vulnerability rankings for aquifers underlying the heavily populated Yinchuan Plain using OREADIC, an adaptation of DRASTIC (modified after Qian et al., 2011).

Ultimately, the analysis of aquifer vulnerability, by whatever method, can be extremely beneficial, often drawing attention to pollution risk that otherwise would be easily overlooked. That said, the analysis needs to be performed with considerable caution, the user remaining acutely aware of inherent limitations and shortcomings of the method. These include its inability to

- provide any indication or measure of the extent to which water quality will be degraded in the aquifer as a function of its vulnerability,
- provide any guidance on what measures should be taken to reduce the risk of contamination in systems determined to be highly vulnerable,
- give any sense of tolerance/uncertainty regarding the analytical results, and
- take account of the aquifer's natural ability to mitigate potential impacts on water quality by mixing with and thereby diluting any contamination that arrives.

As observed by Hiscock and Bense (2014), an aquifer's vulnerability can only be established with confidence through supporting, site-specific field investigations and access to reliable water quality data.

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Of greatest concern are the method's simplifying assumptions and the high degree of subjectivity involved in selecting appropriate parameters and choosing how they should be weighted and combined to best perform the analysis. The user needs to be comfortable that the method is producing valid results. The AVI method performs its assessment based exclusively on the types and thicknesses of the soils and rocks that lie between the aquifer and the ground surface. The soil and rock sediments are assigned a rudimentary K factor depending on their perceived hydraulic conductivity, with gravel and weathered hard rocks (considered the most permeable) given a value of 1 and unfractured igneous and metamorphic rocks (the least permeable) assigned a value of 9. The AVI is determined by calculating, for each layer present, the product of its K number and thickness, and summing these products over the entire sequence. For classification purposes, values of AVI < 30 are categorized as indicative of high vulnerability while those > 80 fall into the low vulnerability category. AVI values falling between those values are classified as representing medium vulnerability. The method is easy to apply, as a vulnerability index can be calculated with little effort wherever a well log is present.

Unfortunately, serious issues can arise when using AVI because the simplifying assumptions sometimes create scenarios that defy normal scientific logic. For example, strict applications of the methodology implies that the presence of 8m of highly permeable gravel (with a K factor of 1) would afford the same level of protection (AVI = 8x1 = 8) as 1 m of untethered clay or shale (K factor 8) (where AVI = 1x8, which also equals 8). This is not hydrogeologically credible. In a similar way, an overburden comprising 5 m of clay and 5 m of silt (AVI = (5x8) + (5x4) = 60), a sequence of layers which would seem to be quite protective, would have the same AVI value as 30 m of sand (AVI = 30x2 = 60), a scenario that would seem to afford minimal protection (source).

Figure 99 illustrates this example of the AVI approach. The method is often used in Ontario, Canada, to assess pollution risk for local aquifers.

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Soil/Rock Type	K-factor
gravel weathered limestone/dolomite (karst) permeable basalt	1
sand	2
peat (organics) silty sand weathered clay (<5m below surface) fractured igneous and metmorphic rock	3
silt limestone/dolomite	4
till (diamicton) sandstone	5
clay (unweathered marine) shale	8
unfractured igneous and metamorphic rock	9

Example of AVI Score Calculation

MATERIAL	K-factor	DEPTH RANGE (m)	THICKNESS (m)	PRODUCT
Clay	8	0-2	2	8x2=16
Silt	4	2-7	5	4x5=20
Clay	8	7-10	3	8x3=24
Sand and gravel aquifer			AVI SCORE:	60

AVI RANGE	CLASSIFICATION	
	CLASS	DESCRIPTION
0 to <30	1	High Vulnerability
30 to <80	2	Medium Vulnerability
>80	3	Low Vulnerability

Examples of Potential Pitfalls:

- The AVI score of 60 shown above for an overburden comprising 5m of clay and 5m of silt (which would normally be considered fairly protective). would also have been obtained with 30m of sand (i.e., 30x2 also =60) (30m of sand would likely provide minimal protection).
- An aquifer overlain by 3m of unweathered clay or shale (resulting in an AVI score of 8x3=24) would fall into the "high vulnerability" when 3m unweathered clay or shale would normally be expected to provide significant protection.
- According to the approach, 8m of highly permeable gravel (8x1= an AVI score of 8) would afford the same degree of protection as 1 m of unweathered clay or shale (1x8= an AVI award of 8), which seems highly unlikely.

Figure 99 - The AVI (aquifer vulnerability index) approach to vulnerability assessment showing the potential pitfalls associated with simplifying assumptions. The simplifying assumptions sometimes create scenarios that defy normal scientific logic as shown by the examples below the table (from Ontario Ministry of the Environment, 2006, and Ontario Ministry of Municipal Affairs and Housing, 2004).

In practice, potential pitfalls with the AVI approach are likely no more serious than those that can arise with DRASTIC and other index techniques. They are simply less obscure and easier to demonstrate. Used by a knowledgeable—yet wary—hydrogeologist, vulnerability assessments can be a useful aid to decision making related to resource protection. Problems of real concern most commonly arise when aquifer vulnerability maps get misused or over-interpreted in the hands of the inexperienced.

Similar criticisms and concerns can be leveled at groundwater resource protection techniques that aim to control/regulate land use activities in the recharge areas that contribute groundwater to a well. These areas are known as wellhead protection areas (WHPAs), source protection zones (SPZs), or zones of contribution (ZOCs) as shown in Figure 100. This technique also qualifies as a standard of practice approach.

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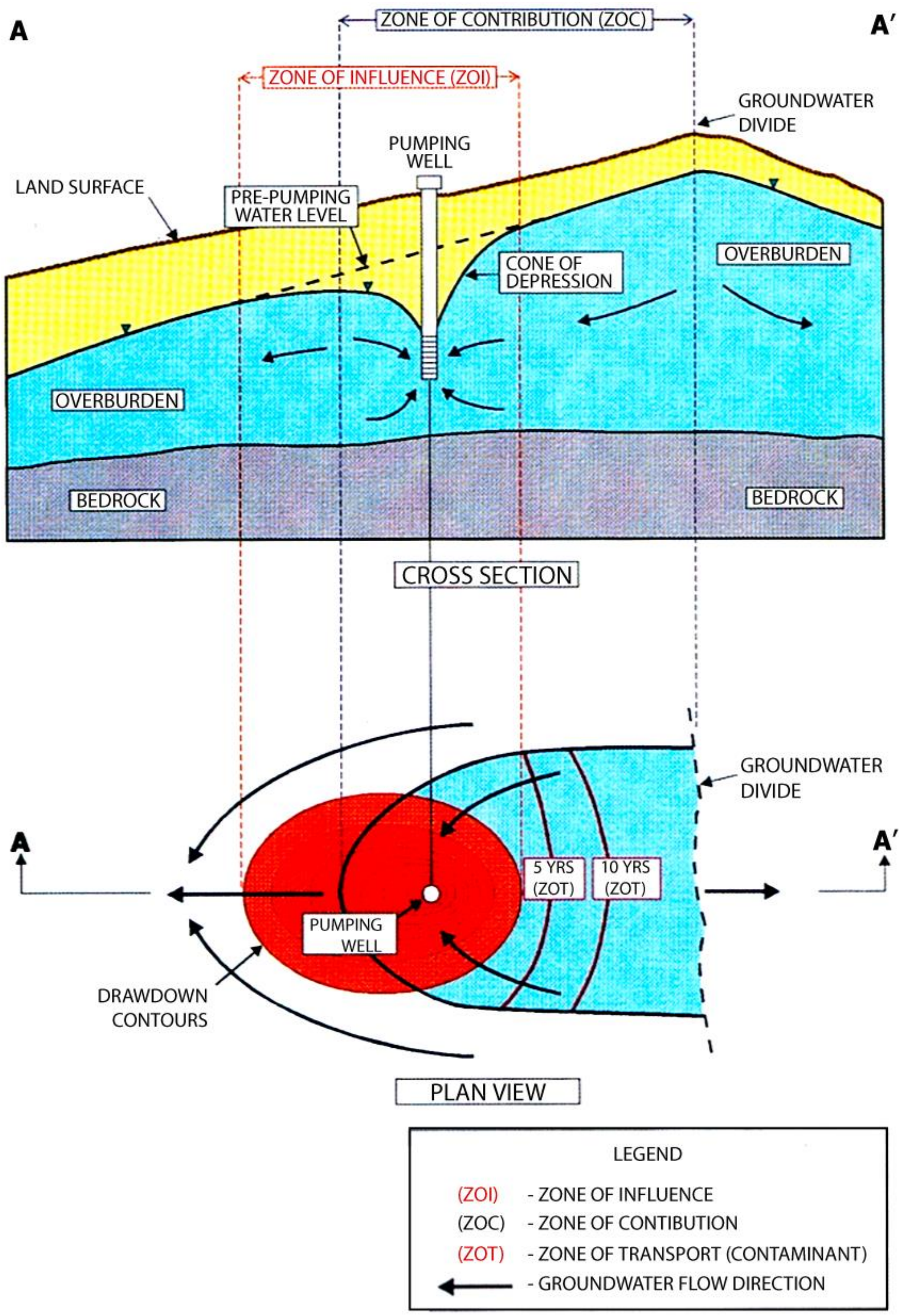

Figure 100 - Conceptual representation of wellhead protection areas and terminology for an aquifer with a sloping water table adjacent to a groundwater divide (modified after Livingstone et al., 1997).

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The use of WHPAs/SPZs/ZOCs for resource protection is popular worldwide (Figure 101), especially for the protection of municipal wells.

Figure 101 - The use of WHPAs/SPZs/ZOCs for resource protection has become popular worldwide, especially for the protection of municipal wells. Roadside signage supported by on-line publicity material is commonly used to engage the public as active contributors to the program goals (from (left) City of New Brighton, Minnesota, USA, 2010; (middle) City of Madison, Wisconsin, USA, 2015; (right) Ohio Environmental Protection Agency, 2023).

Typically, delineating wellhead protection areas allows certain high-risk activities to be excluded or, in many urban areas where they may pre-exist, strictly controlled. Examples include

- industrial sites;
- transport and storage of dangerous chemicals;
- building that would create extensive impermeable areas;
- highways and motorways, unless facilities are installed that would allow the safe collection of run off containing oils, grease, and road de-icing chemicals;
- intensive agriculture and breeding of pigs and cattle;
- pits and quarries;
- landfills for domestic and industrial waste;
- wastewater disposal sites; and
- military activities.

In many jurisdictions, levels of land use restriction would be enhanced near well(s), for example, where time of travel for the groundwater is estimated to be less than two years. This provides lead time for remedial work should an unforeseen spill occur. Regulations may also demand additional protection in the immediate vicinity of wells (e.g., within 100 m or, alternatively, where times of travel are likely to be ≈60 days or less), particularly to guard against contamination by pathogenic bacteria. Exemptions are typically given to chemicals used for water treatment purposes.

Delineating WHPAs/SPZs/ZOCs and estimating travel times can be approached by using various methods, all differing in their degree of complexity and accuracy. In most cases, methods that involve the use of numerical groundwater models provide the most accurate and reliable results. However, this presumes the model is based on a sound

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hydrogeological understanding of the aquifer and has been fully calibrated with accurate field data that includes both potentiometric heads and flows. An example of wellhead protection areas generated by a model is shown in Figure 102.

Figure 102 - Wellhead protection areas for wells servicing the Town of East Gwillimbury, York Region, Ontario, Canada. The wellhead protection is zoned and three zones are shown: WHPA-A with a 100 m radius, WHPA-B (2-year time of travel), and WHPA-C (5-year time of travel). Sites considered to represent a potential risk to water quality are also shown on the map (from Regional Municipality of York, Ontario, Canada, 2017).

The WHPA/SPZ/ZOC approach to groundwater resource protection is similar to index methods in that it provides no indication of the water quality that will be achieved in the aquifer and the extent, if any, to which water quality degradation may occur. It differs, however, in one major respect. The method protects individual wells but does not provide protection for the entire aquifer. Sources of groundwater contamination that lie external to WHPAs/SPZs/ZOCs may not find their way to the well(s) the WHPAs/SPZs/ZOCs are designed to protect, but they are destined to travel elsewhere, for example, to another well, spring, wetland, stream, lake, or the sea (Figure 103).

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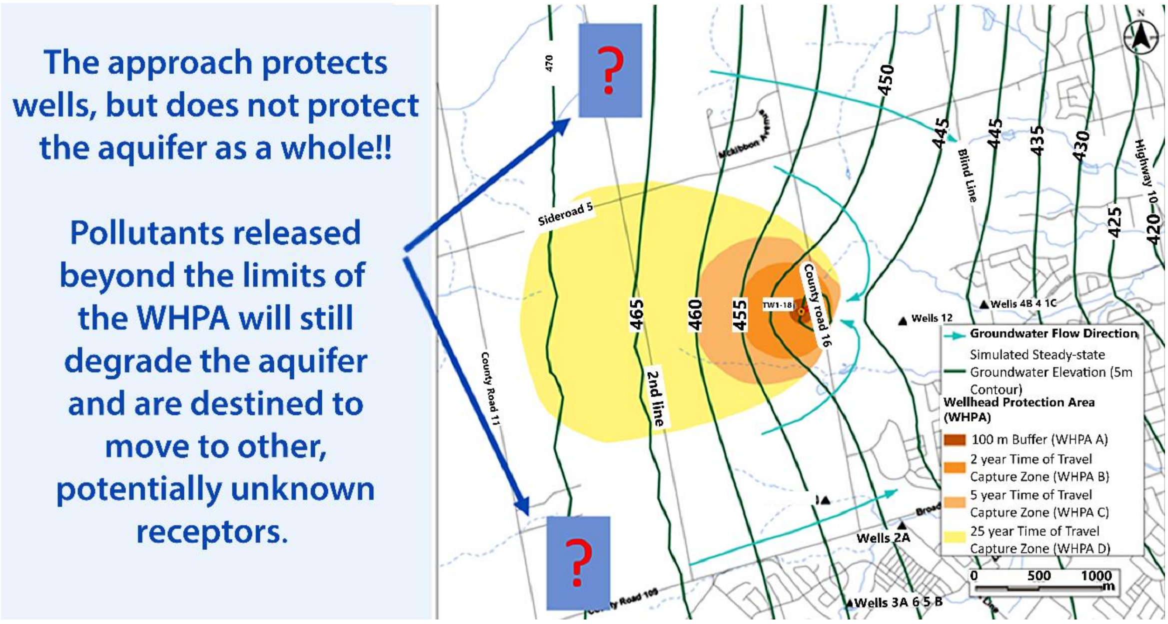

Figure 103 - The WHPA/SPZ/ZOC technique focuses on the protection of individual wells and can be effective for those wells if reliably delimited and the flow field remains stable over time. The practice of excluding potential sources of groundwater contamination from the protected area will inevitably increase the risk of groundwater contamination in the remaining "unprotected" parts of the aquifer (figure by Ken Howard).

I have three other concerns with the WHPA/SPZ/ZOC approach.

1. The quality of the result depends heavily on the approach adopted and the quality of the input data. When numerical models are used, it is essential that details of the overburden materials, aquifer, and any aquitards present are fully represented. Failure to include zones of high permeability or windows in aquitards can render the exercise meaningless.
2. WHPAs/SPZs/ZOCs are often used for planning purposes, especially in the context of urban development. The assumption is frequently made that WHPAs/SPZs/ZOCs are fixed in time and space when nothing could be further from the truth. They shrink and grow in response to changing pumping rates and seasonal variations in rates of aquifer recharge. They move depending on how neighboring wells are pumped, sometimes radically if wells competing for a share of the available aquifer recharge manage to achieve dominance.
3. In effect, WHPA/SPZ/ZOC methods of resource protection, while valuable, should carry the same warning labels attached to vulnerability index techniques, that is, they need to be used judiciously under the watch of an experienced, cautious, hydrogeologist. Once published, WHPA/SPZ/ZOC maps can be easily misinterpreted by those without adequate training.

6.3.2 Standards of Performance

Approaches to groundwater resource protection based on standards of performance offer considerable advantages over those based on standards of practice. Methods based on standards of practice involve strict adherence to process with no real appreciation of what the outcome may be in terms of water quality. In contrast, methods

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invoking standards of performance explicitly recognize that the goal of groundwater resource protection is to ensure groundwater quality meets the specified standards and is designed to comfortably meet those standards virtually unencumbered by rules of practice. However, provided it can be guaranteed that agreed water quality targets will be met, methods using the standards of performance approach enjoy the flexibility to adopt virtually any practice or technology that may be available. This can be extremely important when protecting groundwater in urban areas, which are host to multiple sources of potential groundwater pollution.

One of the best examples of a standards of performance approach to groundwater protection is the application of Ontario, Canada's Reasonable Use Guideline B-7 and Procedure B-7-1 (Ontario MOE, 1994a, 1994b). Guideline B-7 was first introduced as Policy 15-08 by the Ontario Ministry of the Environment in 1987 to deal with potential impacts of a subsurface contaminant release on the "reasonable use" of groundwater on adjacent properties. It was designed to deal primarily with regulated contaminant sources such as landfills, exfiltration lagoons, and large subsurface sewage systems, which would require approvals for their establishment, operation, or extension. However, the fundamental principles espoused in Guideline B-7 have considerable versatility and, with minor modifications and adaptations, represent a model for broad-scale groundwater protection anywhere in the world.

Guideline B-7 was one of the most progressive and practical groundwater policy documents ever formulated by the Ontario Ministry of the Environment, even from an international standpoint. It was, for example, one of the first guidelines published anywhere to acknowledge that zero impact is an impossible target and stringent, albeit practical, limits need to be established as to the degree to which groundwater quality can be changed as a result of human activity; groundwater moves extremely slowly compared with surface water and the contaminating lifespan of a chemical release can extend many decades beyond the period during which the release occurs; and significant attenuation can take place in the subsurface because of dilution, dispersion, chemical reaction, chemical transformation, decay, and biodegradation, thus it is appropriate to consider such processes in evaluating potential impacts.

In practice, Guideline B-7 establishes target water quality concentrations that must be met at the property boundary for all time. In the case of landfills, this would be the extreme edge of the site property; for a planned urban subdivision (should the approach be adapted for such uses), this would be the perimeter of the urban footprint.

As outlined in Procedure B-7-1, which accompanies Guideline B-7, the change in the quality of groundwaters at the property boundary must not exceed 50 percent of the difference between background and the quality criteria for any designated reasonable use, except in the case of drinking water. Where the designated reasonable use is drinking water, the quality must not be degraded by an amount that exceeds 50 percent of the difference between background and the Ontario Drinking Water Objectives for

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non–health related parameters. Neither must it be in excess of 25 percent of the difference between background and the Ontario Drinking Water Objectives for health-related parameters.

In effect, this approach imposes a permanent upper limit to the amount of contamination that the owner of the adjacent property should have to tolerate. To quote from Section 5.1 of the procedures document (B-7-1), "*it is the Ministry's judgment that such increases in contaminant levels will have no more than a negligible or trivial effect on the existing or potential reasonable use of the adjacent property*."

With the reasonable use target established, the proponent of the land use change becomes free to prepare a design that can guarantee this target can be met for all time. With straightforward contaminant sources such as infiltration lagoons, this would normally involve the application of two- and three-dimensional numerical models capable of simulating groundwater flow and contaminant transport. However, in the case of landfills, especially those in Ontario, this task has regularly fallen to POLLUTE, a computationally efficient model conceived and developed by Rowe and others (1994) that fully incorporates key aspects of landfill design including composite liners, geomembranes, covers, and leachate collection systems (Rowe & Fraser, 1997). POLLUTE has undergone several developments over the years and is currently distributed by GAEA Technologies Ltd. in version 8 (released September 2021 with many new features).

A more conventional example of the standards of performance approach is the use of MODFLOW (McDonald & Harbaugh, 1988), a popular finite-difference simulation model and MT3D (Zheng, 1990) a popular solute transport simulation software, to explore the potential impacts of a proposed new urban subdivision (the Seaton Lands) in the Duffins Creek watershed, east of Toronto, Canada (Howard & Maier, 2007). The site location is shown in Figure 104. The study focused primarily on the extent to which road salt (NaCl) would degrade groundwater in the uppermost aquifer and considered many development scenarios. The analysis was performed using a MODFLOW/MT3D model developed by Gerber (1999) that included nine layers with a grid discretization of 200 m by 200 m (110 columns and 150 rows) and was configured using borehole data from approximately 7,000 Ontario water well records. Calibration details are provided by Gerber and Howard (2002).

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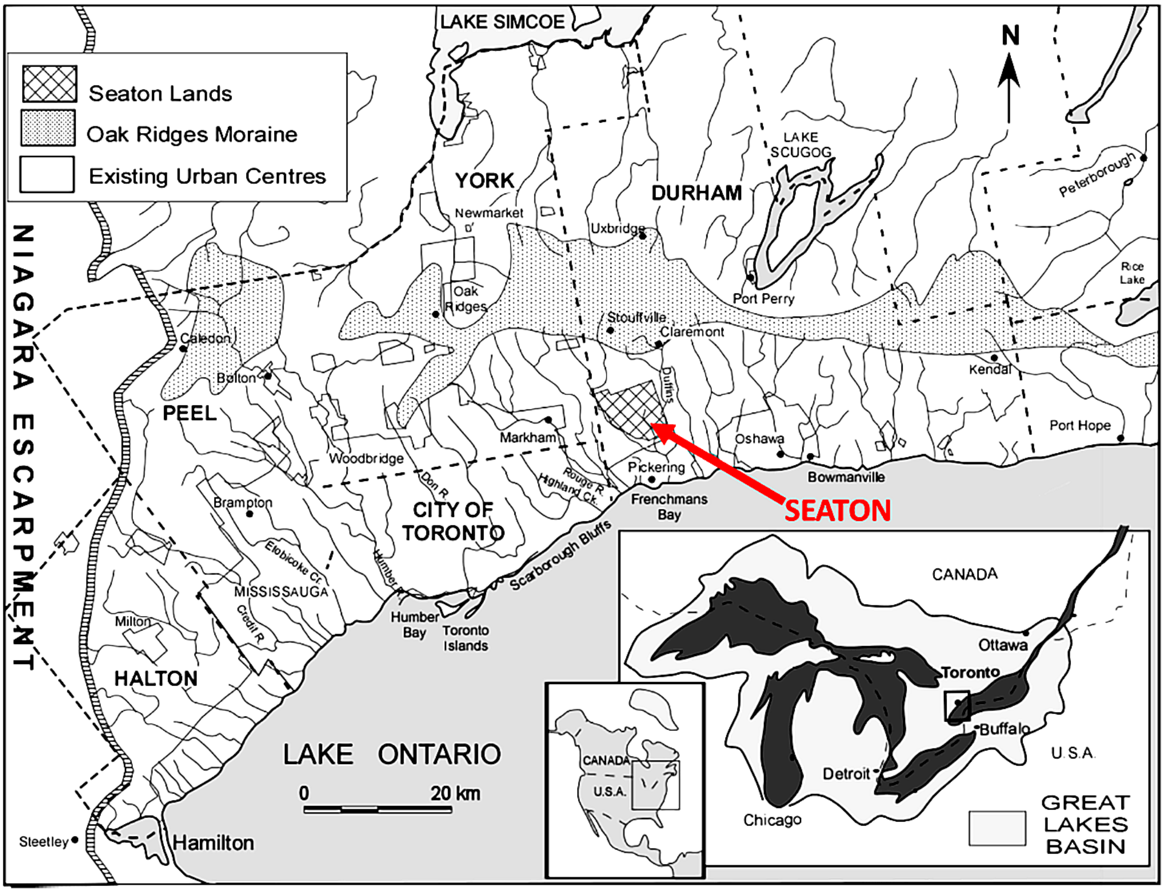

Figure 104 - Location of the Seaton Lands in the Duffins Creek watershed east of Toronto, Canada, 20 km is about 12.5 miles (Howard & Maier, 2007).

To examine the impacts of salt application, salt was applied to roads and highways using documented application rates (Howard & Maier, 2007).

1. Fifty five percent of the applied salt was assumed to enter the aquifer in urban areas serviced by storm water collection systems based on Howard and Haynes (1993).
2. One hundred percent of the salt applied was allowed to enter in rural areas.
3. For urban subdivisions, where salt application rates to roads are much lower than those for highways and arterial roads, salt loadings were estimated based on average road density.

Following data entry, the model was run to chemical steady state (a period of about 2,500 years) to predict the long-term impacts of the development on the multi-layer aquifer. Chemical steady state for the uppermost aquifer was achieved within about 200 years. Two scenarios were run: one that omitted the Seaton Lands development and another that included it. Several possible urban-density designs were considered. The results for these designs are shown in Figure 105 and Figure 106, respectively.

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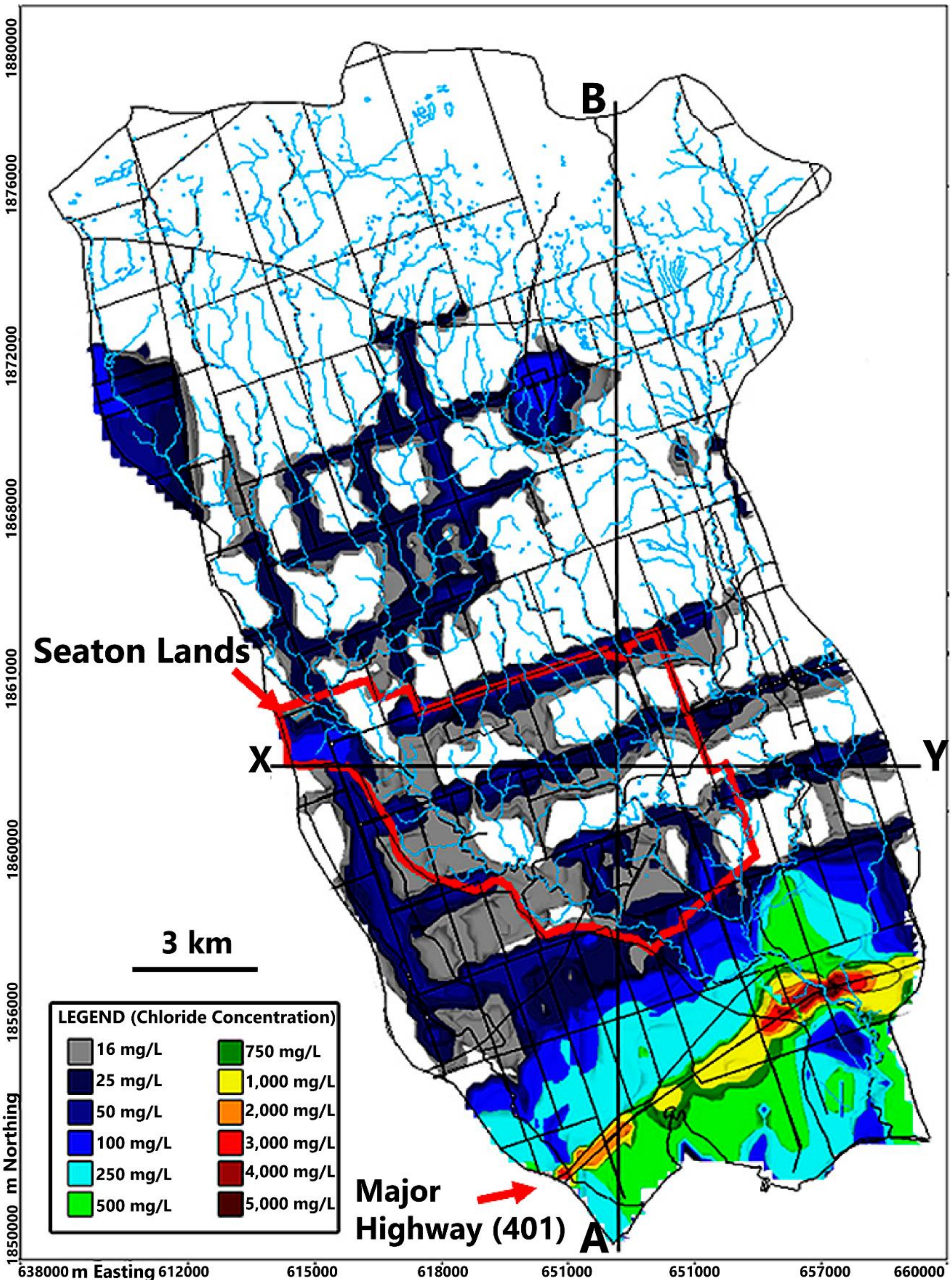

Figure 105 - Predicted long-term, steady-state chloride concentrations in the uppermost aquifer in the absence of urban development in the Seaton Lands study area. Most of the impacts are associated with arterial roads and the heavily urbanized area in the south that flanks Highway 401 (Howard & Maier, 2007).

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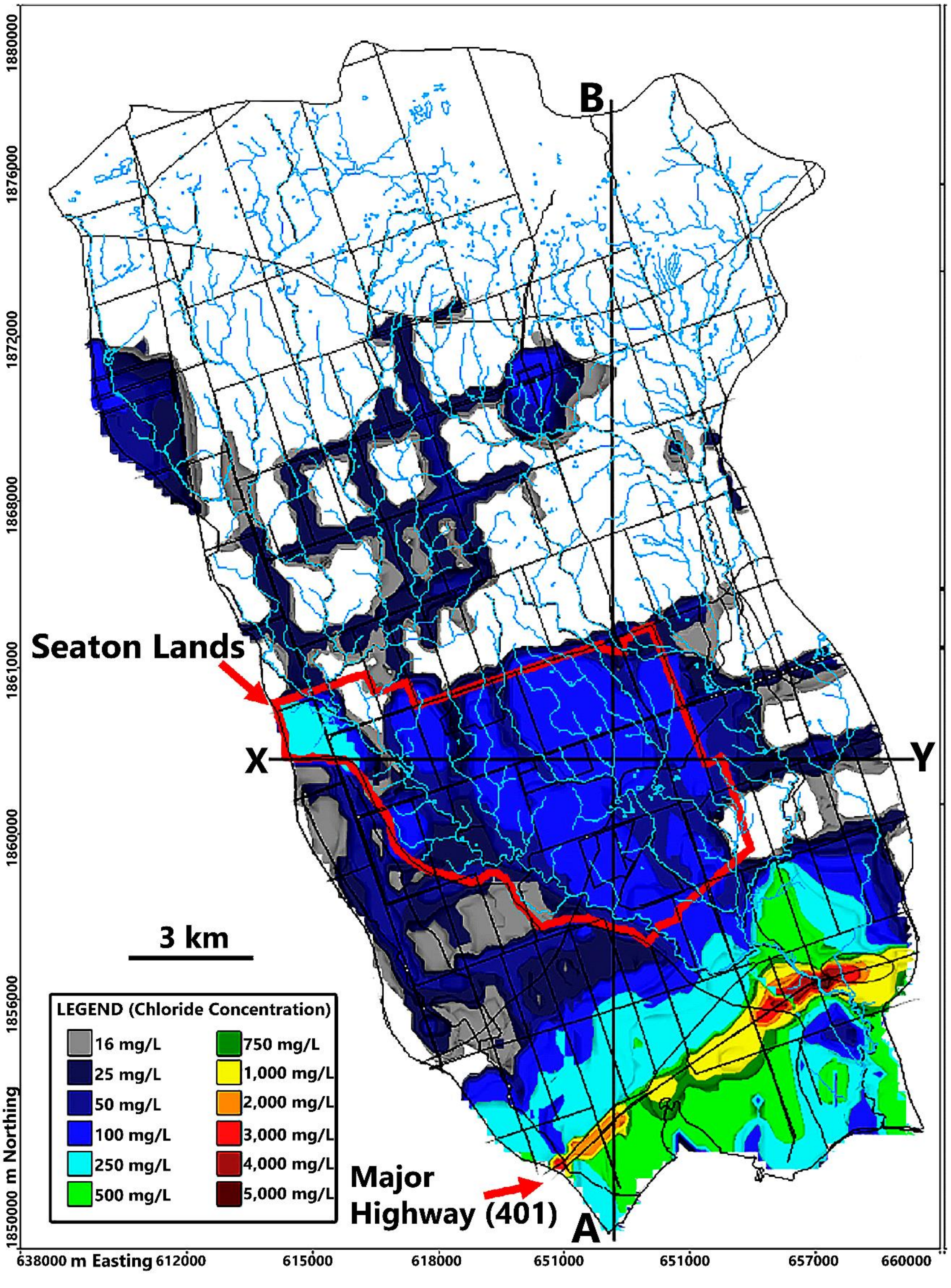

Figure 106 - Predicted long-term, steady-state chloride concentrations in the uppermost aquifer due to road salt application following development of the Seaton Lands study area. The development significantly increases chloride concentrations in the uppermost aquifer (Howard & Maier, 2007).

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A comparison of the two figures shows that development of the Seaton Lands will significantly impact groundwater in the uppermost aquifer. Throughout most of the new subdivision, chloride concentrations in shallow groundwater can be expected to reach ≈100 mg/L which—while high—remains below the aesthetic water quality standard of 250 mg/L set for drinking water. Significantly, the model suggests the 250 mg/L standard could be challenged in the northwest corner of the site, thereby demonstrating one of the major benefits of adopting a standards of performance approach to groundwater resource protection. Supplied with quantitative information on the extent to which water quality may degrade, adjustments can be made to the proposed subdivision plans that could reduce water quality impacts. In the northwest corner of the site, for example, a reduction in road density could alleviate potential problems.

In practice, the scenario described here is hypothetical since the application of road salt is not strictly regulated in the province of Ontario. Locally, setbacks/buffer zones (examples of standards of practice) may be established to protect surface water features such as wetlands or sensitive streams, but most jurisdictions see the use of road de-icing chemicals as essential for human safety on the roads, preferring to minimize potential damage to the environment by promoting best efforts or encouraging the adoption of best management practices.

Certainly, implementation of setbacks or buffers (standards of practice) would do little, if anything, to protect groundwater from the impacts of road salt, especially at the urban subdivision scale. However, NaCl road salt would prove an excellent candidate for regulation using Ontario's reasonable use standards of performance approach, should rules ever be amended to permit its inclusion. Application of the reasonable use guidelines for NaCl road salt use would, for a new subdivision, create a target maximum chloride concentration of ≈130 mg/L at the boundary of the subdivision (this concentration representing the mid-point between natural background (10 mg/L) and the aesthetic drinking water quality objective of 250 mg/L). If this regulatory approach were adopted for the Seaton Lands development, Figure 106 suggests that most of the site would comply with the regulation.

The use of numerical models is not obligatory when it comes to the protection of groundwater resources using a standards of performance approach. On the contrary, many assessments can be made with a few relatively simple calculations involving knowledge of the mass of chemical released vertically to each square meter of aquifer every year (mass per unit area or, in terms of dimensions, ML^{-2}) divided by the annual rate of aquifer recharge (represented as a depth of water, L). This will provide a mean annual concentration for that chemical (ML^{-3}) which can be compared to the water quality standard. If the mass of chemical released is such that the concentration remains, consistently, safely below the drinking water standard, that chemical will never pose a threat to the aquifer's drinking water quality.

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For example, consider the construction of a new urban subdivision 5 km by 1 km i.e., 5×10^6 m^2. If annual recharge through the site to the underlying aquifer is 200 mm/a, 10^6 m^3 (10^9 L) of water passes through the site each year. The residents apply fertilizer to their lawns as nitrate-nitrogen; a proportion of this fertilizer finds its way to the aquifer. Residents also have septic systems, which add more nitrate-nitrogen to the aquifer together with significant chloride. The aquifer receives additional chloride due to the use of de-icing chemicals applied to driveways and roads. Chemical loading on the aquifer will depend on resident density and road density, and a few simple calculations (essentially a chemical audit) will allow impacts on water quality to be estimated.

As an example, one density scenario for the new subdivision may result in the release of 50 tons of chloride and 6 tons of nitrate-nitrogen to the aquifer each year, meaning that the average concentrations of chloride and nitrate-nitrogen in recharge entering the aquifer will be 50 and 6 mg/L, respectively. These concentrations would be comfortably within drinking water quality standards (250 mg/L and 10 mg/L, respectively) and may be deemed acceptable. Similar calculations (performance assessments) could be made using higher urban densities (residents and roads) to determine what (in terms of water quality degradation) the aquifer could reasonably sustain.

Performing chemical audits can become a challenge when other chemicals are considered (e.g., multi-chemical leachate "soups" from landfills and leaks/spills emanating from gasoline stations), since estimating aquifer loading rates can be difficult without good statistical data for the size and frequency of unplanned releases. However, the potential value of such audits can be immense, especially in urban areas where multiple sources of aquifer contamination are present and long-term degradation of shallow water quality degradation is always a threat.

A similar but alternative approach is to use the concept of impact potential (IP), defined as the volume of water that would be contaminated to the drinking water quality standard by the mass of chemical released to the aquifer (Howard & Livingstone, 2000). The IP (L) is calculated by dividing the mass of chemical released (in mg) by the drinking water quality standard for that chemical (mg/L). Where the calculated impact potential (in L) is less than the volume of water entering as recharge over the same contaminating lifespan, water quality standards will not be exceeded; that is, sufficient recharge is available to dilute the mass of chemical released to a concentration that will meet the standard. In the example described above, IPs for chloride and nitrate-N for a one-year period would be 2×10^8 and 6×10^8 L, respectively—that is, in each case, significantly below the $1\text{x}10^9$ L of water that enter the aquifer each year as recharge.

If readers still question the potential value of using chemical audits as an approach to groundwater resource protection, they may be persuaded by the results of an audit conducted for part of the Greater Toronto Area (Canada) by Howard and Livingstone (2000). The study area is shown in Figure 107.

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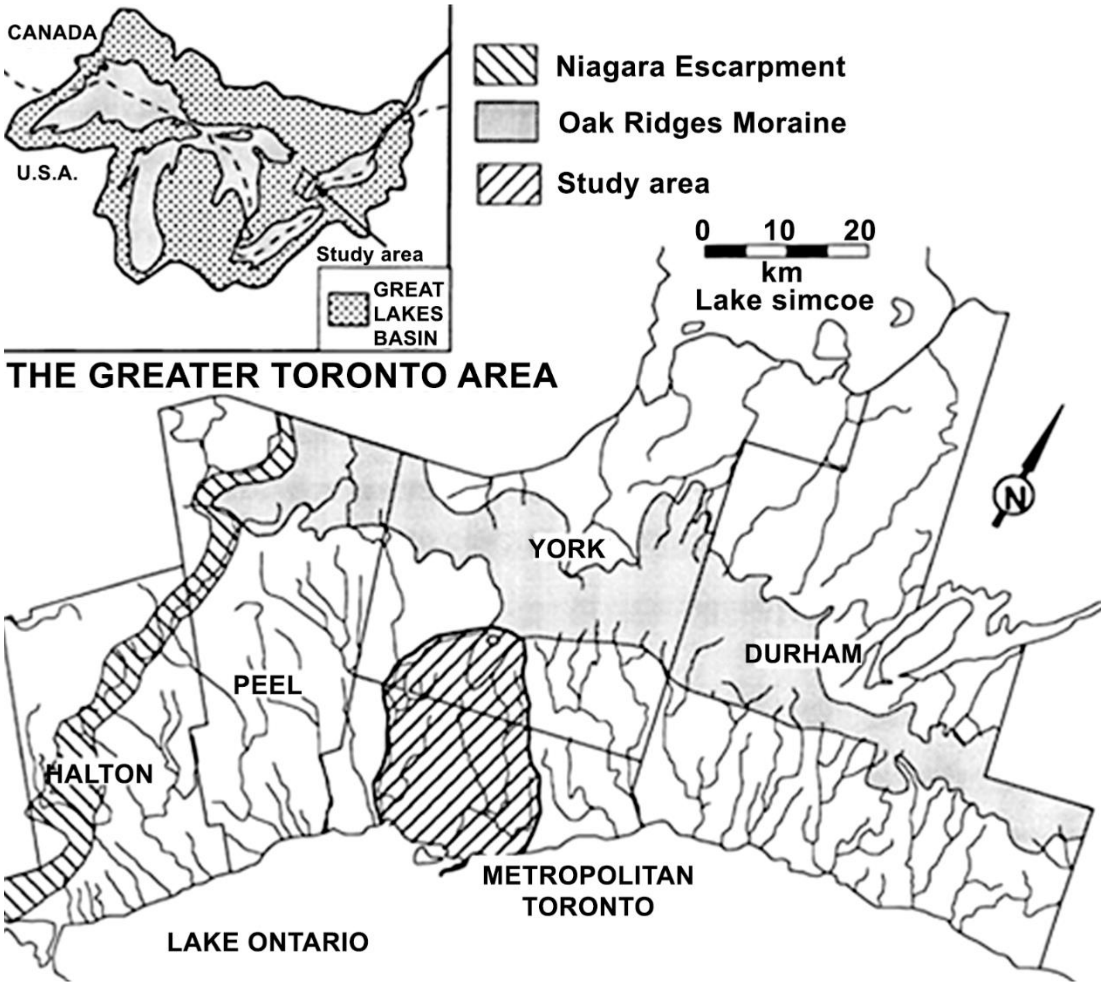

Figure 107 - Greater Toronto Area (Canada) showing the 700 km^2 area selected for the chemical audit (Howard and Livingstone, 2000).

The purpose of the audit was to estimate the mass of chemicals released to the subsurface from numerous potential sources of contamination between ≈1950 and ≈1990, a period of around 30 to 50 years when the city underwent massive, post-war urban growth. Point sources audited included 82 active and abandoned landfill sites; 2,100 USTs containing gasoline; multiple snow dumps; and ≈5,000 septic systems. Major distributed sources included de-icing chemicals applied to 4,200 lane-kilometers of road and highway and chemical fertilizers/pesticides applied to gardens and parkland. To err on the side of caution, the likely role of biodegradation as an ameliorating process was ignored.

Potential leakage from the sewer system was not considered, since pipes were unpressurized and in relatively good condition, especially compared to those underlying many European cities. Groundwater is not used by the city for supply, and its water quality is not routinely monitored. However, groundwater feeds local rivers and streams and eventually enters Lake Ontario either directly or indirectly. A primary objective of the study was to establish the potential for contamination of Lake Ontario via groundwater pathways.

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Estimated masses were converted to IPs; as shown in Table 31. To put the IPs in context, the volume of groundwater immediately underlying the study area is estimated at approximately 10^{13} L, and the volume of groundwater entering the study area each year as recharge and leaving as urban stream baseflow is about 10^{11} L (i.e., approximately 3 to 5×10^{12} L over the past 30 to 50 years). Since IPs for chemicals in most of the contaminants listed in Table 31 are estimated to exceed this range, serious questions must be raised over groundwater quality. Recognizing the residence time for groundwater is around 100 years (1 percent aquifer replenishment rate), much of the contaminated groundwater is still on its way to Lake Ontario. While the chemical audit has achieved little other than to forewarn of impending impacts to Lake Ontario, its use here illustrates the potential value of audits in future urban design.

Table 31 - Impact potentials: Volume of water contaminated to the drinking water quality standard by the mass of each chemical released (L) (Howard and Livingstone, 2000).

Parameters	Landfills	Septic systems	Underground storage tanks	Snow dumps	Former coal tar plants	Road de-icing chemicals	Agriculture
Inorganics							
Chloride	2.5×10^{11}	1.4×10^{10}		8.7×10^{8}		3.7×10^{12}	1.3×10^{10}
Sodium	1.0×10^{12}			7.3×10^{8}		3.0×10^{12}	
Total nitrogen	2.7×10^{12}	3.1×10^{11}					2.7×10^{12}
Copper	1.6×10^{13}			1.3×10^{12}			
Lead	4.6×10^{12}			1.1×10^{12}			
Chromium	1.1×10^{12}						
Cyanide	4.6×10^{12}						
Mercury	1.2×10^{11}						
Arsenic	2.8×10^{11}						
Organics							
Benzene	4.0×10^{11}		3.0×10^{13}		3.2×10^{8}		
Ethylbenzene	2.6×10^{12}		4.1×10^{13}		1.7×10^{8}		
Tetrachloroethene	6.8×10^{10}						
Trichloroethene	2.9×10^{11}						
Dichloromethane	6.5×10^{10}	5.2×10^{9}					
Phenol	2.8×10^{15}						
1,4-Dichlorobenzene	4.1×10^{11}	2.0×10^{11}					
Toluene	1.9×10^{13}	9.8×10^{12}	6.2×10^{14}		1.3×10^{9}		
Xylene (total)	3.1×10^{13}		2.7×10^{14}		3.0×10^{8}		

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6.4 Resource Protection/Management Using Models Specifically Designed for the Urban Environment

Most groundwater-dependant cities throughout the world obtain their water from external well fields. In this scenario, popular numerical models are as follows:

- Finite Element Modeling of Flow (FEFLOW), a versatile finite element modeling code developed by WASY GmbH, Berlin; and
- Modular Three-Dimensional Finite-Difference Groundwater Flow Model (MODFLOW) Flex, one of several commercially developed forms of the USGS finite-difference modeling code MODFLOW (McDonald & Harbaugh, 1988, 2003).

These models are useful in establishing appropriate resource management and protection plans. They can simulate transient groundwater flow in complex, multi-layer aquifer systems and can fully represent the behavior of entrained contaminants.

In some cases, aquifers underlie urban areas, usually because of urban growth that brings well fields that were once external or peri-urban to within city limits. Aquifers located within urban boundaries pose unusual challenges for numerical modeling, as they often feature characteristics that can prove problematic. These potentially problematic characteristics may include

- pronounced vadose zone activities/processes, for example, urban karst effects and numerous, near-surface pollutant releases that together can exert a strong influence on groundwater flow and contaminant transport in underlying aquifer(s);
- multiple sources and mechanisms of aquifer recharge, many of which lack quantifiable description (or even physical understanding);
- misalignment in response times for surface water and groundwater, intensified by the "flashy" behavior of surface runoff and urban streams;
- aquifer properties that can dramatically change with time due to construction activities involving use of fill, installation of tunnels, and erection of buildings with deep foundations;
- complex boundary conditions that can change rapidly with time;
- densely spaced points of abstraction (due mainly to multiple private users but exacerbated by dewatering for construction), which often change in time (and sometimes in space) in an unpredictable manner, thereby strongly influencing the flow of groundwater and transport of contaminants;
- highly heterogeneous land use changing rapidly with time resulting in
- a marked heterogeneity in the spatial and temporal distribution of recharge;
- significant (and often unknown) changes in pollutant loadings, which in turn result in heterogeneous distributions of pollutants in 3-D space throughout both the vadose zone and the aquifer(s);
- the presence of legacy pollutants released during previous generations;

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- the unabated introduction of a very wide range of chemicals, some novel with unknown or unpredictable behavior; and
- pollutants with imperfectly understood transport properties, including NAPLs (nonaqueous phase liquids) and particulates (especially bioparticles).

While many of these characteristic features can and, in fact, have been represented with varying degrees of success in popularly used models such as MODFLOW and FEFLOW, interest is growing in the design of specialized models or model components that deal specifically with the types of hydrogeological conditions commonly encountered in urban environments. Three examples are briefly introduced below: AISUWRS, UGROW, and urban members of the MIKE SHE family of software products.

6.4.1 AISUWRS (Assessing and Improving Sustainability of Urban Water Resources Systems)

One of the first and most innovative approaches to urban water systems modeling is known as AISUWRS (Assessing and Improving Sustainability of Urban Water Resources Systems). This modeling tool comprises a series of closely linked urban component models that were developed during an international research project funded by the European Commission; the Department of Education, Science and Training of Australia; and the United Kingdom Natural Environment Research Council. The participating organizations included

- University of Karlsruhe, Germany (Coordinator);
- British Geological Survey, United Kingdom;
- Commonwealth Scientific and Industrial Research Organization (CSIRO), Australia;
- FUTUREtec Gmbh, Germany;
- GKW Consult, Germany;
- Institute for Mining and Geology, Slovenia; and
- University of Surrey (Robens Centre for Public and Environmental Health), United Kingdom.

Full details of the model are published in a comprehensive *Urban Water Resources Toolbox* (Wolf et al., 2006).

The AISUWRS project was driven by a perceived need to fully integrate groundwater into urban water management. At the heart of the project was the recognition that, while land and water use in urban areas is highly complex, cities are expanding, and municipal water utilities can no longer afford to neglect water in an underlying productive aquifer just because it is difficult to assess.

As originally conceived (Eiswirth, 2002), the AISUWRS model linked an existing Urban Volume and Quality model (UVQ model) developed through Australia's CSIRO urban water program (UWP) with a groundwater flow model (e.g., FEFLOW or MODFLOW) via a series of (ARCINFO®) GIS layers (Figure 108). At the heart of AISUWRS

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and key to its success is the highly innovative UVQ model, which estimates water flows and contaminant loads within the urban water system (e.g., recharge, supply, wastewater, stormwater) on a daily scale. It is aided by various additional model components including the Network Exfiltration and Infiltration Model NEIMO and Ulflow/SLeak/Posi (Figure 109), which are used to calculate losses (flow and contaminants) from sewers and pressurized water supply pipes, and natural vertical flow in areas free of construction.

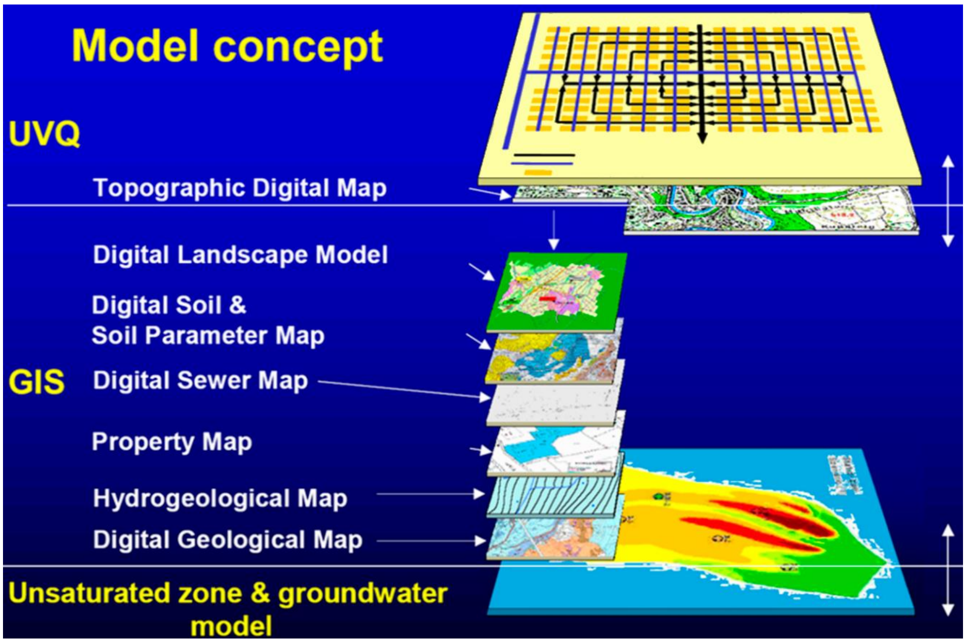

Figure 108 - AISUWRS model concept (as originally conceived by Eiswirth, 2002).

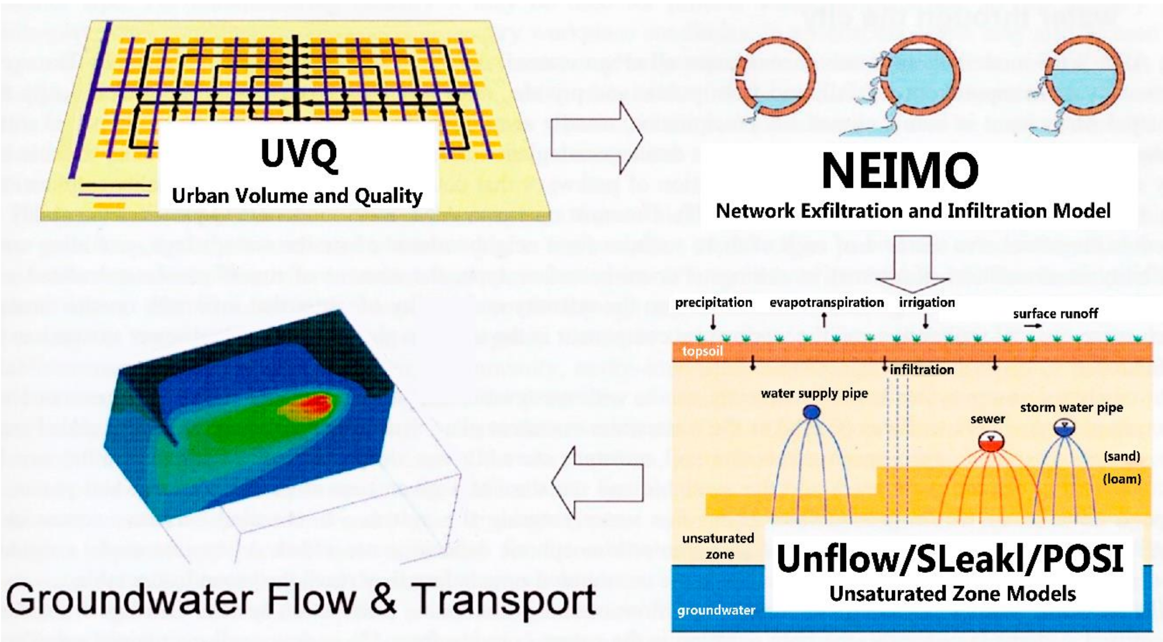

Figure 109 - Major model compartments demonstrating the integrated approach of AISUWRS (after Wolf et al., 2006).

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A UVQ flow diagram showing all the urban water cycle components represented by the model is provided in Figure 110, and an example of a completed urban water balance is provided for the City of Doncaster, United Kingdom, in Figure 111.

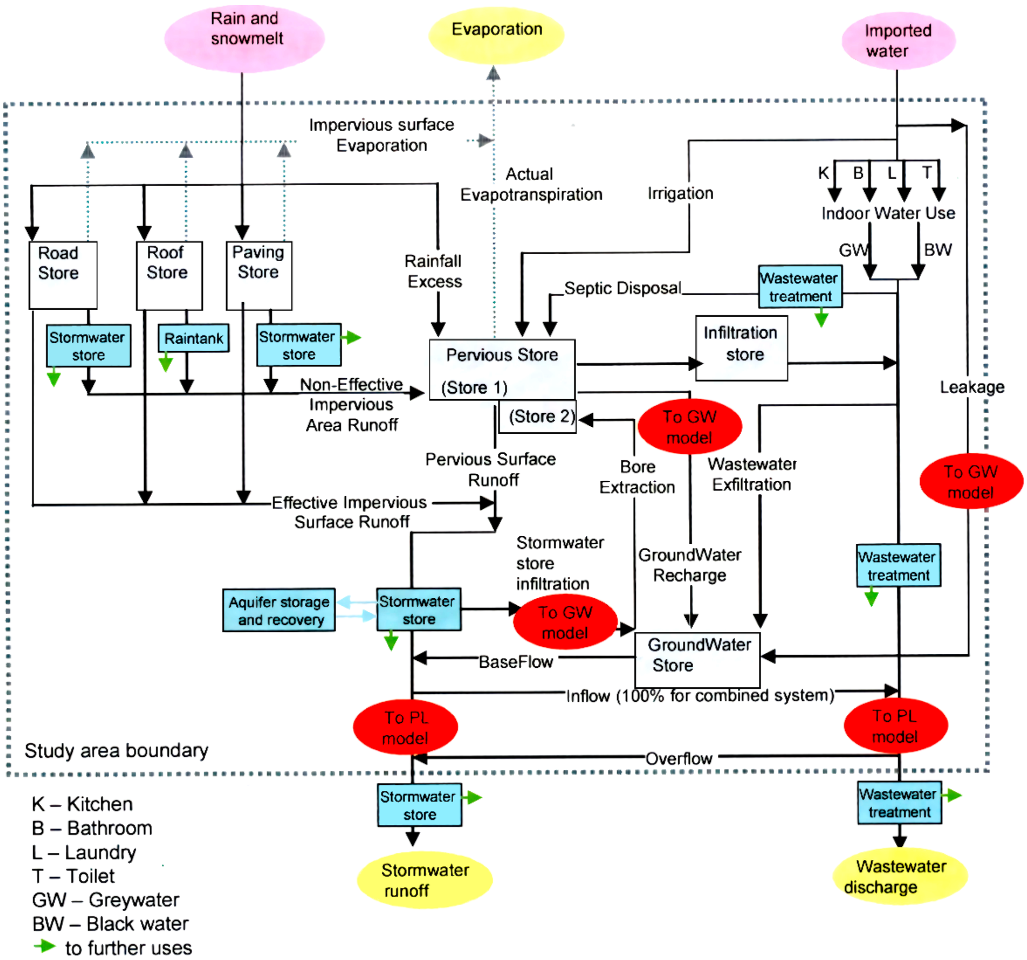

Figure 110 - UVQ flow diagram representing all the key components of the urban water cycle (after Wolf et al., 2006).

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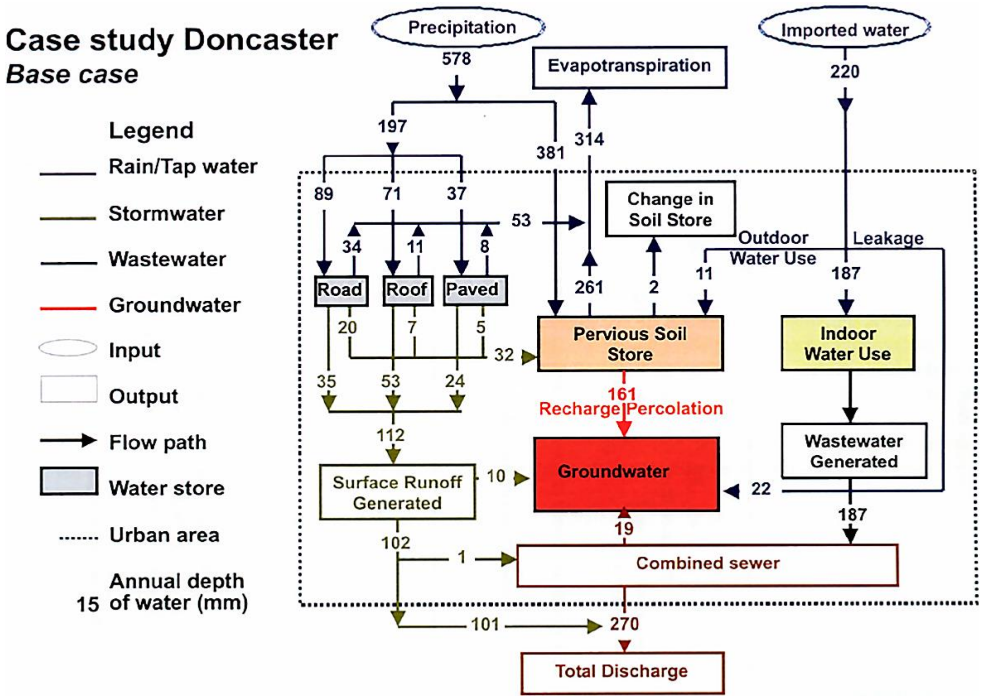

Figure 111 - Urban water balance determined using the UVQ for the City of Doncaster, England (after Wolf et al., 2006).

The fully linked system including the decision support system (DSS) and the Microsoft Access database is shown in Figure 112.

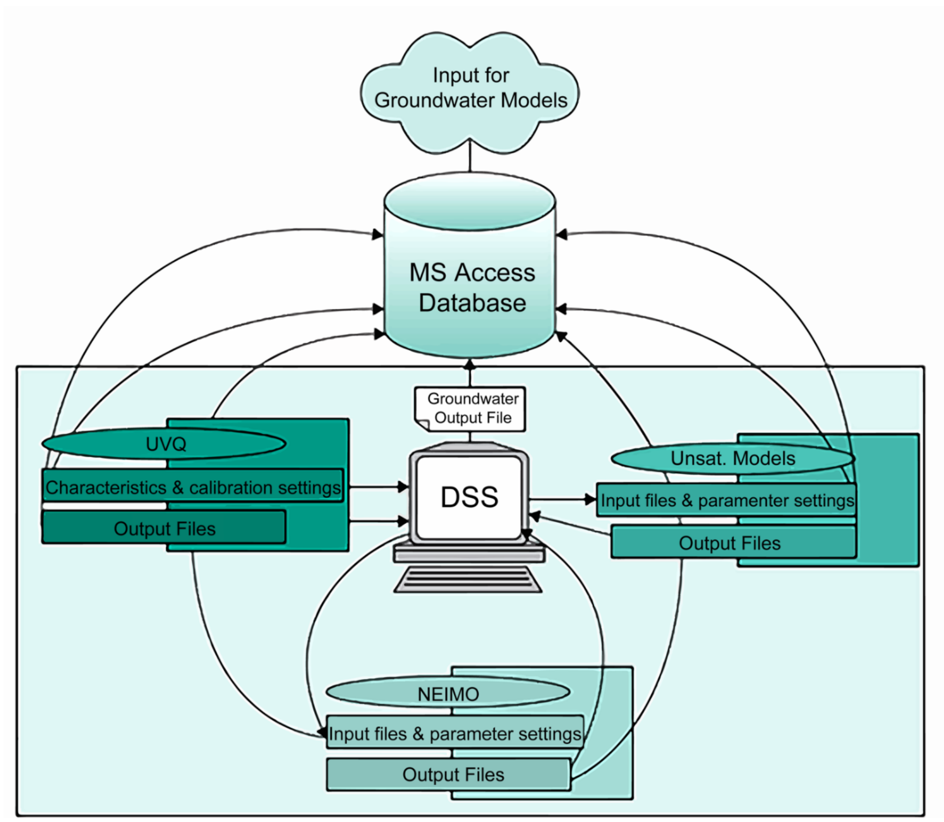

Figure 112 - Linkage between the key AISUWRS model components, the decision support system (DSS), and the Microsoft Access database (after Pokrajac & Howard, 2010 and Wolf et al., 2006).

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6.4.2 The UGROW Model

One disadvantage of the AISUWRS urban water modeling tool is its coupling approach that ultimately places reliance on an independently developed groundwater flow model, such as FEFLOW or MODFLOW, to complete the package and deliver the results. Achieving a seamless "marriage" between model components can be a common problem with coupled models, especially when individual components are designed to standalone by those with little appreciation of connectivity issues that may lie ahead. This is often why such models are underutilized.

Pokrajac and Howard (2010) managed to overcome the linkage/connectivity problem through the development of UGROW (Urban GROundWater), as shown in Figure 113, Figure 114, and Figure 115—a complete and fully integrated urban water systems model that included a dedicated finite element groundwater flow model as part of the modeling package. UGROW was developed with the support of the sixth phase of UNESCO's International Hydrological Programme (IHP-VI, 2002–2007).

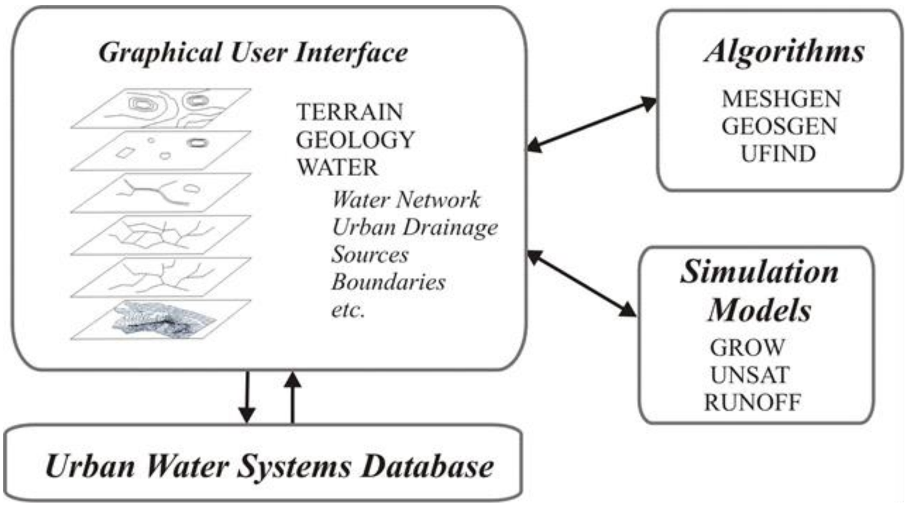

Figure 113 – Basic structure of UGROW (Pokrajac and Howard, 2010).

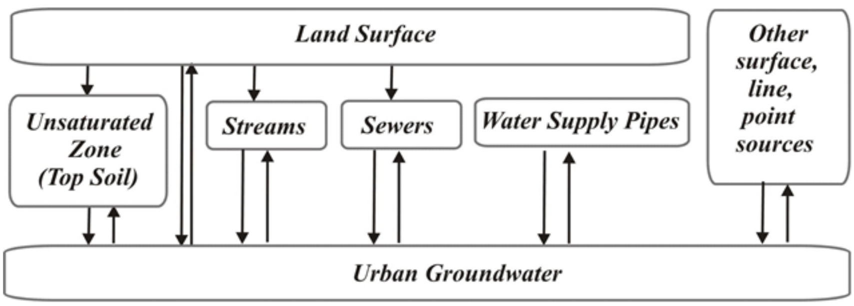

Figure 114 - Interaction between control volumes in the urban water balance (Pokrajac and Howard, 2010).

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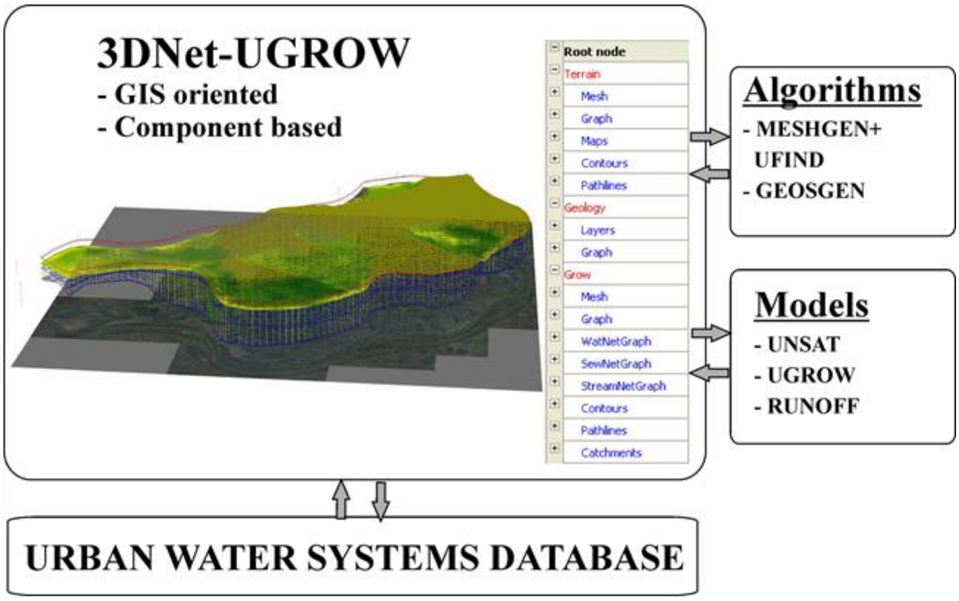

Figure 115 - An example of 3DNet-UGROW on-screen linkages (Pokrajac and Howard, 2010).

Early versions of UGROW were field-tested using three urban groundwater case study sites:

- Rastatt, Germany, where a major problem is groundwater infiltration into sewers, which results in an overloading of the water treatment plant;
- Pančevački rit, Serbia, where the urban water balance is poorly known due to many contributing water systems; and
- Bijeljina in Bosnia, where groundwater has been seriously polluted by septic tanks.

In each case, UGROW performed successfully and the feedback from each study was used to refine model components. The Rastatt study allowed UGROW to be compared with AISUWRS, an exercise that produced positive results. It was found that users who were not involved in the code development could operate UGROW successfully, albeit with appropriate support. All the studies showed that care must be taken with the model parameterization and interpretation and that appropriate sensitivity analyses should be performed before proceeding with the modeling task.

As published in 2010, the most serious limitation of UGROW was its inability to incorporate more than one aquifer system. While this could be an issue where cities overlie multi-layer aquifer systems, much can be achieved by focusing on the uppermost aquifer, since, generally, protecting the uppermost aquifer automatically ensures that deeper aquifers receive comparable protection.

6.4.3 Urban Members of the MIKE SHE Software Family

The development of the AISUWRS and UGROW models raised considerable awareness for the complexity of urban water systems, how various system components interact, and the difficulties of acquiring appropriate data for quantifying flows and

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contaminant fluxes. It also demonstrated the critical need for managing and protecting urban water holistically and with full consideration (quantity and quality) of all urban water balance components. Work on urban water systems continues relentlessly in universities and urban research centers throughout the world. However, although details of the AISUWRS and UGROW models have been widely published, the models were never commercialized, and their application to urban problems globally has been limited.

Fortunately, several software companies have been developing numerical models for commercial distribution for decades; many of these models can be applied to problems frequently encountered in urban areas. A full review of models suitable for urban aquifers and urban water systems is beyond the scope of this section, however—any review would soon be out of date, as modeling software innovation tends to evolve at a rapid pace. However, one suite of modeling tools with a long urban track record and a popular following is the MIKE SHE family of software.

MIKE SHE (Figure 116) is an integrated hydrological modeling system used for the analysis, planning, and management of a wide range of water resources and environmental problems related to surface water and groundwater. It is applied particularly to gauge surface water impact from groundwater withdrawal, conjunctive use of groundwater and surface water, wetland management and restoration, river basin management and planning, and impact studies for changes in land use and climate.

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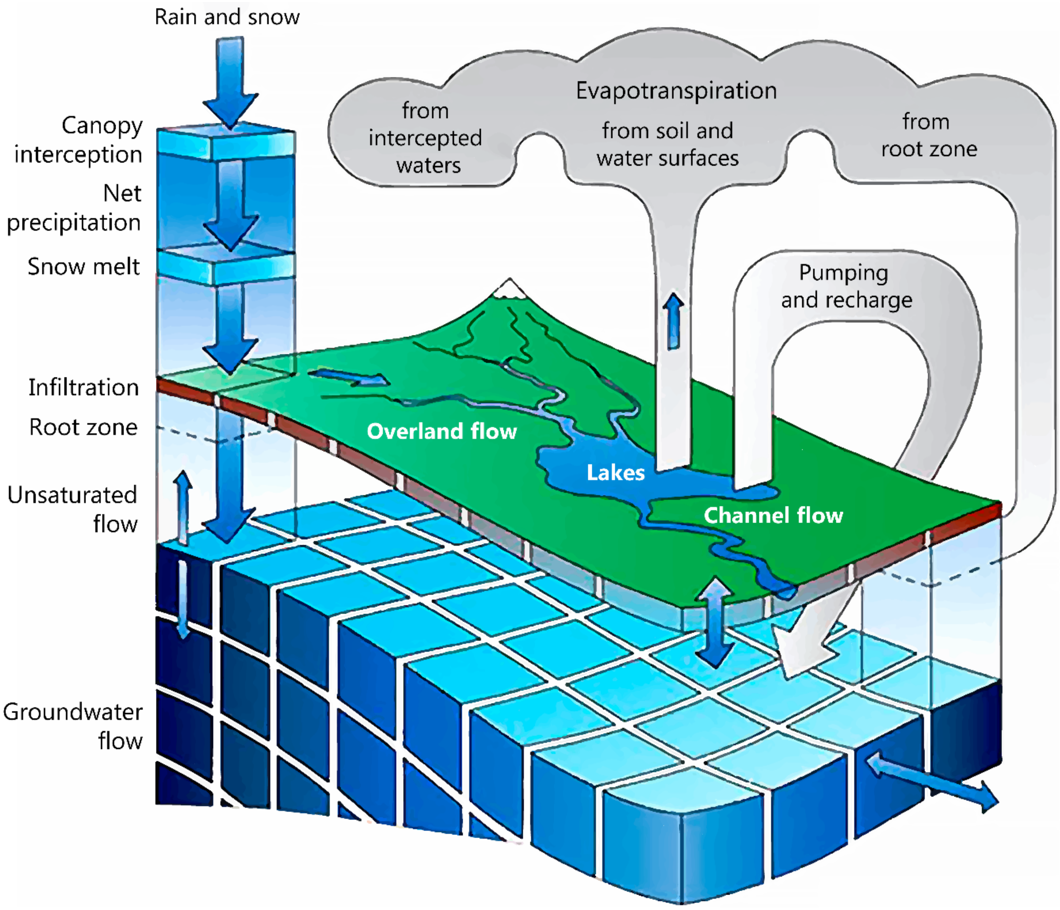

Figure 116 - Hydrological process simulated in MIKE SHE (from Graham & Butts, 2005).

The program's origins lie in a consortium of three European organizations: the Institute of Hydrology (United Kingdom), SOGREAH[4] (France), and DHI (Denmark) (Abbott et al., 1986) that were responsible for development of SHE.[5] Since those early days, DHI has continuously invested resources into research and development of MIKE SHE and has created an expanding family of models, many of which relate directly to urban water issues (Figure 117).

[4] Société Grenobloise d'Études et d'Applications Hydrauliques
[5] Système Hydrologique Européen

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INTEGRATED PLATFORMS

- MIKE+
  - Water Distribution
  - Collection Systems
  - Rivers
  - Flooding
- MIKE Cloud
  - Mesh Builder

CITIES

- MIKE+
- MIKE URBAN+
- MIKE URBAN
- WEST
- WaterNet Advisor
- DIMS.CORE

GROUNDWATER AND POROUS MEDIA

- FEFLOW

COAST AND SEA

- MIKE 3 Wave FM
- MIKE21/3
- MIKE 21 Boussinesq Waves
- MIKE 21 Mooring Analysis
- MIKE 21 /3 Oil Spill
- MIKE 21 /3 Particle Tracking
- MIKE 21/3 Sand Transport
- MIKE 21/3 Mud Transport
- MIKE 21 Shoreline Morphology
- MIKE 21 Spectral Waves
- MIKE ECO Lab
- ABM Lab
- LITPACK

WATER RESOURCES

- MIKE+
- MIKE 21C
- MIKE HYDRO Basin
- MIKE HYDRO River
- MIKE SHE

GENERAL

- MIKE FLOOD
- MIKE ECO Lab
- MIKE OPERATIONS

ADDITIONAL TOOLS

- MIKE Animator Plus
- MIKE C-MAP

DATA & OPERATIONAL DECISION MAKING

- MIKE OPERATIONS

MIKE FOR DEVELOPERS

- MIKE API

Figure 117 - DHI's expanding family of MIKE SHE software (from MIKE Powered by DHI).

The MIKE SHE family of software products adopts a coupling approach, using the powerful and highly versatile FEFLOW model to handle most of the flow in the subsurface. With respect to urban areas, MIKE URBAN / MIKE URBAN+ / MIKE+ (Figure 118) come into play for overland flow and flow through engineered water systems. For more details, the reader is referred to the DHI website↗.

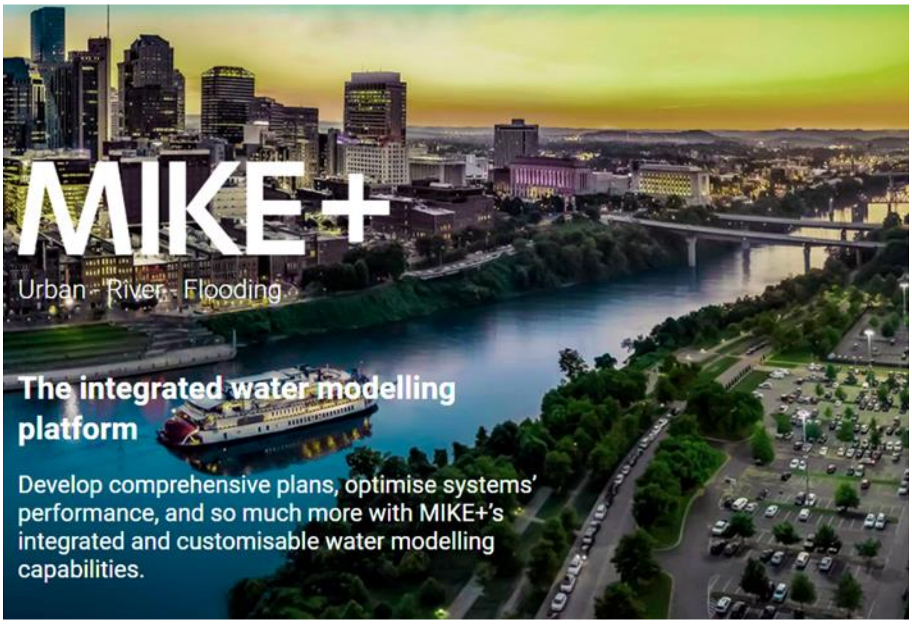

Figure 118 - MIKE URBAN / MIKE URBAN+ / MIKE+ are popular water modeling tools for the city environment (from MIKE Powered by DHI, 2023).

6.5 Exercises Related to Section 6

Exercises related to Section 6 are available at this link↴.

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7 Urban Groundwater Governance

The challenge of providing the world's rapidly growing urban population with adequate and sustainable supplies of water is formidable. However, the knowledge base is strong, urban groundwater is well understood, and there is good reason for optimism. Unfortunately, sound science coupled with innovative technologies form only part of the global solution when it comes to urban water supply issues. Ultimately, it is essential that urban water resources be responsibly managed. Yet, while the science and engineering needed to manage groundwater is advanced, the key to implementing sound water resources management lies in a reliable structure of water governance. Governance issues (Figure 119) are discussed briefly here.

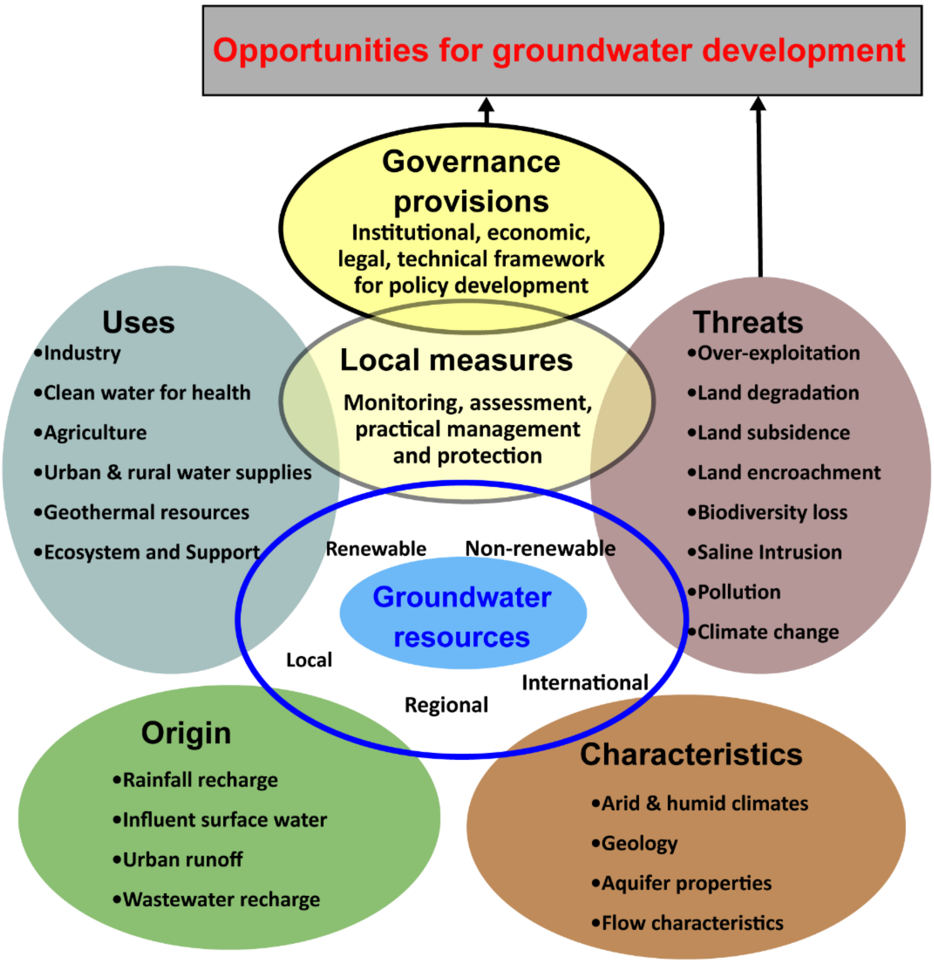

Figure 119 - Diagrammatic representation of the linkages between the origins, characteristics, and uses of groundwater resources and the threats to and opportunities for change presented by the good governance of groundwater (graphic from Howard, 2023).

The meaning of groundwater governance has been defined in many ways and is a popular topic of debate (for example, Varady et al., 2013). It also tends to be interpreted and utilized somewhat differently by hydrogeologists and water engineers than by social scientists. However, for the purposes of discussion, the definition of groundwater

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governance adopted in this section is derived from Foster and van der Gun (2016) and is a slightly modified version of the definition proposed by Foster and Garduño (2013, p. 317). It reads as follows:

> *"Groundwater governance comprises the promotion of responsible collective action to ensure socially sustainable utilization, control, and protection of groundwater resources for the benefit of humankind and dependent ecosystems."*

The task of managing groundwater in urban areas is beset with many difficulties. First and foremost is the hydrogeological complexity of the urban subsurface, a region altered by decades of construction work that often hosts multiple sources of locally derived contaminants—many of which are a legacy of unsafe historical practices such as unregulated disposal of waste.

The problem is compounded by an escalating demand for water by a rapidly growing population, coupled with the burden of managing, safely and responsibly, increasing volumes of wastewater without creating further deterioration of groundwater quality. Under such scenarios, there can be little surprise that strategies for urban groundwater management are rarely planned proactively, purposefully, based on sound science and reliable field data. On the contrary, they tend to be developed on the fly, with decisions often made hurriedly under political pressure, with severe budgetary constraints, and in response to unanticipated demand and/or imminent water quality problems. This needs to change, and the starting point is the establishment of a system of groundwater governance that fully respects the role groundwater plays in the urban water cycle and is ready and willing to integrate groundwater with its unique attributes into all aspects of urban land use planning, water supply, and waste management, whatever its present status or function.

7.1 Evolving Perspectives on Urban Groundwater Management and Governance

Concern for the threat to global water supplies from rapidly growing cities was first explicitly highlighted in the Dublin Statement on Water and Sustainable Development (or Dublin Principles) endorsed at the International Conference on Water and the Environment (ICWE) on 31 January 1992. Since that time, urban growth—and the formidable challenge of supplying rapidly growing cities with adequate supplies of water for drinking and sanitation—has remained a major focus of the global water sector.

Urban water supply issues have been profiled at every World Water Forum since 1997 (discussed in Section 1.2) and featured strongly during the sixth phase (2002–2007) of UNESCO's International Hydrological Programme (IHP-VI Focal Area 3.5, "Urban areas and rural settlements"). The protection and management of groundwater in urban areas was also a major focus of the World Bank's Groundwater Management Advisory Team

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(GW-MATE), resulting in numerous briefing notes, case profiles, and strategic overviews (discussed in Section 5.2).

One particularly important symposium took place at UNESCO in Paris in September 2007 (Figure 120). Key problems were identified as

- fragmented institutions (in part, geographically, but also across different facets of the urban water cycle and urban water components),
- weak regulatory and institutional frameworks with excessive centralization and an unclear division of responsibilities between the central and local governments,
- a lack of trained and qualified staff at all levels,
- inefficient and outdated management practices,
- misguided decision making due to short-term political or commercial interests that, in turn, lead to inadequate capacity to address urban water challenges, and
- limited stakeholder participation that led to local tensions in the community and increasing numbers of urban water conflicts.

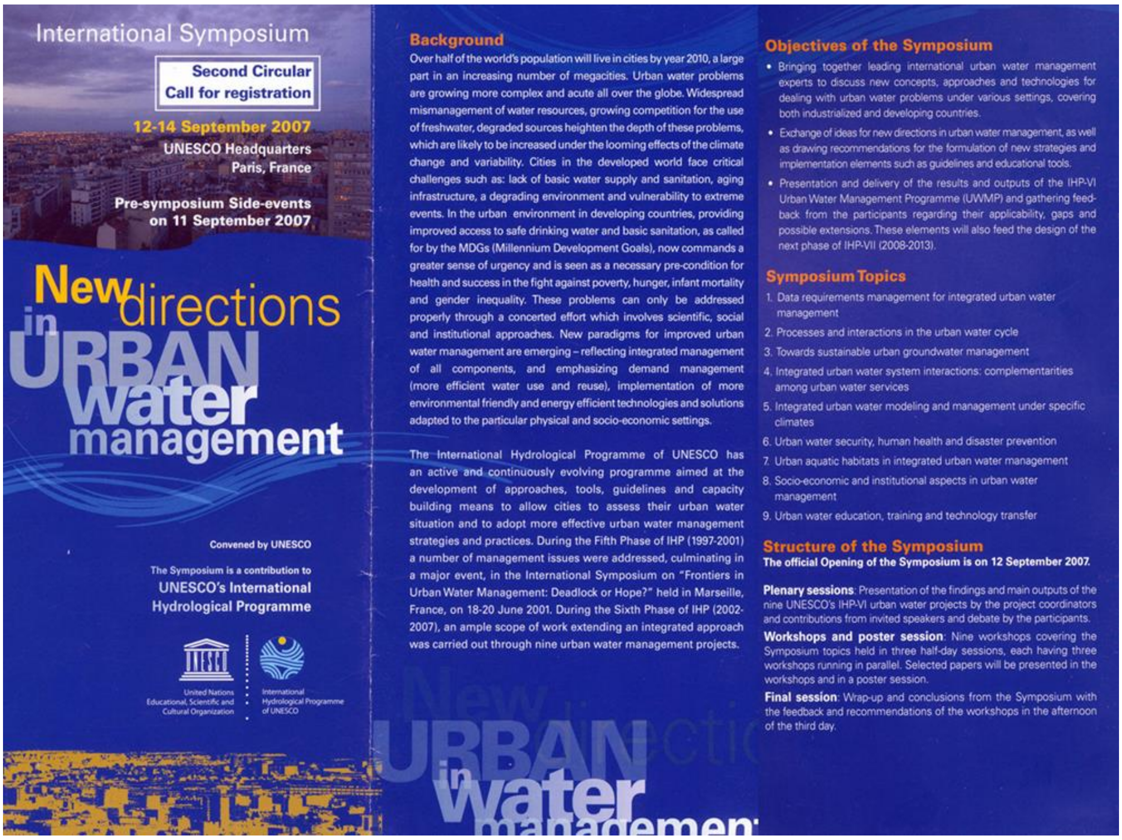

Figure 120 - The International Symposium on New Directions in Urban Water Management (September 12–14, 2007) drew participants from around the world. It set the stage for *The Paris-2007 Statement on New Directions in Urban Water Management*, a strong declaration that re-emphasized the enormous burden placed on water resources by unprecedented rates of population growth and urbanization. It also drew attention to a "widespread crisis of urban water governance, particularly in developing countries" (UNESCO-IHP, n.d.).

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The Paris-2007 Statement stressed the need for new approaches to the management of water in urban areas that would include

practicing the protection and sustainable management of groundwater as an indispensable source of water supply for a large part of the world's population;

- moving away from water supply management towards a focus on the management of water demand;
- acquiring, storing, managing, and sharing of data and essential information on urban water and aquatic habitats to support effective and responsible management of the entire urban water cycle;
- using the entire water cycle (including groundwater), with all its components and their interactions, as a unifying framework for effective management;
- managing the interactions between components of the urban water cycle, giving particular attention to the mutual interaction between engineered infrastructure and the natural environment;
- making full use of new advances in urban water cycle modeling tools at appropriate spatial and temporal scales;
- recognizing that, while water is vital for human life, it also fulfils important aquatic functions in groundwater-dependent ecosystems and ecohydrology;
- acknowledging and accounting for the complex socio-economic issues associated with urban water management (e.g., concepts such as social inclusion, affordability, user participation, preferences, and acceptability);
- making a clear distinction between water as a service and water as a resource, especially with respect to understanding conflicts and their resolution (the lack of such a distinction is the root cause of many urban water tensions/conflicts and is a key consideration for water rights and allocation issues); and
- adopting a participatory approach to the management of urban water (including decision making) by actively engaging with a wide circle of stakeholders at all levels of society (e.g., users (customers), professionals (planners, builders), and interest groups).

In concluding, the Paris-2007 Statement identified a series of key concepts on which sustainable urban water management should be based. These concepts included

1. enhancing resilience of urban water systems to global change pressures,
2. making interventions over the entire urban water cycle,
3. invoking demand management including a reconsideration of the way water is used (and reused),
4. making more prudent use of existing infrastructure,
5. making more frequent use of local and natural systems,
6. improving governance and financial management structures, and
7. promoting more active stakeholder participation.

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What is significant about the Paris-2007 Statement, as profoundly important as it may have seemed, was the scant mention of groundwater, at least in terms of its properties, compared with surface water: Groundwater is out of sight, has high storage volumes, is slow moving, and is highly resilient to climate change. A further consideration is how groundwater's properties should influence how urban water (a combination of groundwater and surface water) should be managed. This failure to fully acknowledge the role of groundwater may have been significant, but hardly unexpected, since groundwater resources seldom received major global investments at this time (Figure 121) and was rarely, if ever, featured in World Water forums and United Nations World Water Development Reports. The actual source of the water was never deemed to be important.

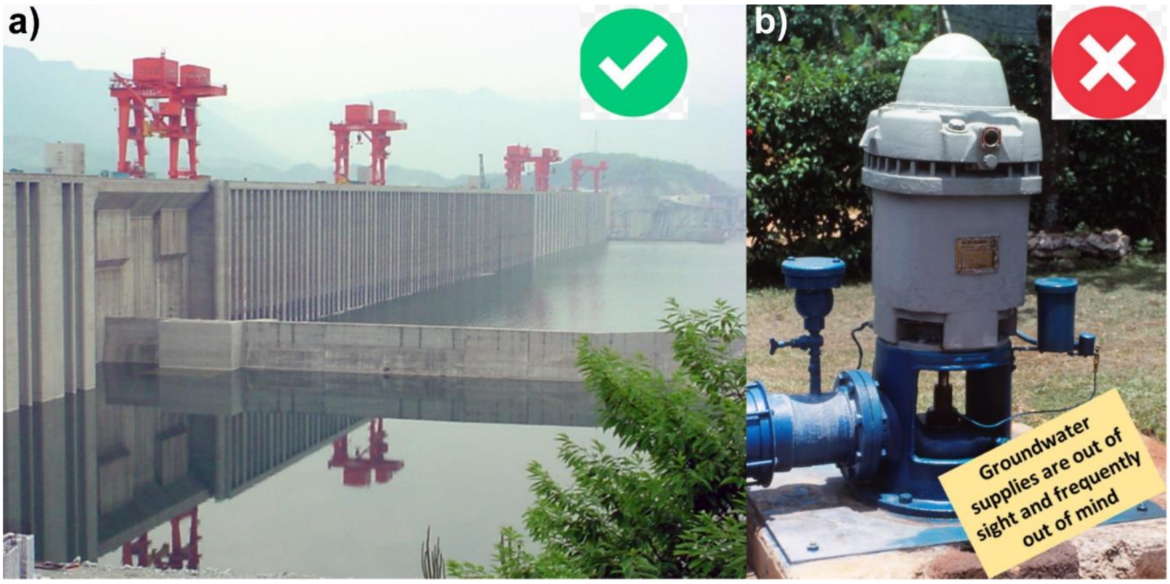

Figure 121 - Global investment in water has always favored spectacular, expensive, but easily monitored surface water schemes such as dams and pipelines over low public-profile wellfield developments involving "invisible" groundwater. Figure a) shows the Three Gorges Dam (China), and b) shows a municipal well near Kingston, Jamaica (Photography by Ken Howard).

Some would argue the situation has changed little since then, with surface water still dominating business at the World Water Council. Whether this is a fair assessment or not, the implications of ignoring the unique and special attributes of groundwater when it comes to urban water supply are important and discussed in more detail below.

7.2 Urban Groundwater and the Failure of IWRM/IWM/IUWM

Integrated Water Resources Management (IWRM), as shown in Figure 122—together with its similarly inspired sister, water management principles (IWM - Integrated Water Management and IUWM - Integrated Urban Water Management)—was first advocated by the United Nations in the 1950s. IWRM principles featured strongly at the United Nations Water Conference held in Mar del Plata, Argentina, in March 1977, but the concept did not achieve any serious traction until the 1990s when the principles of IWRM

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began to receive strong endorsement from various water agencies and were regularly promoted at international meetings.

Figure 122 - General framework for Integrated Water Resource Management (JFT Consulting, 2021).

Integrated Water Resource Management (IWRM) is defined as "*a process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems*" (Global Water Partnership 2000, p. 22). In practice, IWRM (and comparable approaches) involves the application and integration of knowledge from a range of disciplines, together with insights from stakeholders, for the purpose of finding and supplying fair, efficient, yet sustainable solutions to the world's pressing water problems. It has been described as a comprehensive tool for managing, developing, and delivering water to consumers in a way that is socially, economically, and environmentally responsible (van Hofwegen & Jaspers, 1999).

IWRM's open, multi-stakeholder approach to water management has enjoyed global acclaim. It was adopted as a key management principle in the European Union's Water Framework Directive of 2000 and has since provided guidance for subsequent EU water development programs such as the EU Water Initiative, launched in 2002 at the World Summit on Sustainable Development in Johannesburg. However, not everyone endorses its value and functionality so enthusiastically. Despite its popularity and strong promotion by the water agencies, IWRM seems to have enjoyed only limited success. It has not proved to be a panacea for the world's water problems and may have done little more than raise awareness for the complexity of urban water problems, including the need for improved water governance.

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IWRM's primary attribute, namely, its demand that global water issues be approached holistically, is commendable. However, the number of factors that need to be considered and integrated in this holistic approach tends to be so large that practical implementation is impossible unless the list is strongly redacted. Herein, serious problems can arise. Inevitably, some important elements get ignored while others are combined. According to the 2006 Stockholm Water Prize Laureate, Asit K. Biswas (2008), who is shown in Figure 123, the application of IWRM to real world water projects has left much to be desired. On a scale of 1 to 100 (one being no integrated water resources management and 100 being full integration), Biswas argues there is no IWRM project in the world that would earn a score of 30 or more from an objective analyst.

Figure 123 - Prof. Asit K. Biswas, 2006 Stockholm Water Prize Laureate and founder of the International Water Resources Association (IWRA) (photograph by the China Institute of Water Resources and Hydropower Research (IWHR), 2018).

In effect, the ability of IWRM to succeed in a practical sense relies heavily on the factors selected for integration, a process that from the outset is highly subjective. Since the term water includes both groundwater and surface water sources, then by implication, it is fair to assume that groundwater and the role of aquifers would be automatically included as a factor within IWRM's process of integration. Unfortunately, in practice, nothing could be further from the truth, particularly in rapidly growing urban areas. In many cases, groundwater is either ignored or simply lumped together with surface water, despite the fact groundwater and surface water operate on distinctly different scales of time and space. In many applications of IWRM, very little recognition or weight is given to the vital function that groundwater plays in the global water cycle and the immense benefits that could be derived from the improved management of groundwater. A failure to recognize

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the unique and special attributes of groundwater represents one of the lost opportunities of IWRM.

Integration has its merits, but it does not have to be carried out under the guise of IWRM. Foster and others (2011) suggested that a fully integrated approach to municipal water services and urban development is required, and this should include consideration of groundwater sustainability—especially where local aquifers are providing an important component of municipal water supply. The overriding challenge is to provide a framework for water governance in urban areas that will facilitate the development of water supply solutions as part of an urban water resource management plan or action plan (Foster et al., 2010a) that is fully and proactively embedded within the urban planning process. This plan needs to consider both quality and quantity aspects of the resource and be developed in close collaboration with all stakeholders using good data, sound science, and reliable demographic projections. Such an action plan is described below. Current problems with urban groundwater management will be resolved only if governments are willing to work in collaboration with groundwater users rather than attempting to regulate and control them.

7.3 The GEF Project - Groundwater Governance - A Global Framework for Action (2011–2014)

The GEF Project - Groundwater Governance - A Global Framework for Action (GCP/GLO/277/GFF) is outlined in Figure 124. It was a multi-million-dollar project supported by the Global Environment Facility (GEF) and implemented by the Food and Agriculture Organization of the United Nations (FAO) jointly with UNESCO's International Hydrological Programme (UNESCO-IHP), the International Association of Hydrologists (IAH), and the World Bank. The project ran from 2011 to 2014 but was many years in the making. It was conceived during the early 2000s in recognition of an emerging water crisis that involved the following factors:

1. rapid population growth was placing an enormous burden on water resources, with over 1 billion people lacking access to safe drinking water;
2. severe competition for water resources developed among agriculture, livestock, energy, mining, and industrial sectors, which represented a serious and increasing threat to economic development, food security, and poverty reduction; and
3. vital ecosystems were being deprived of water and this was leading to land degradation, loss of biodiversity, and the threat of impoverished livelihoods.

The project was developed in recognition of the following facts:

- Little weight is given to the vital function of groundwater in the global water cycle, nor to the immense benefits provided by proper management of groundwater.

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- Groundwater and surface water behave on very different spatial and time scales and require distinctive approaches to resource management that are often not fully understood, appreciated, and accommodated within the principles of IWRM.
- Groundwater is a vast but seriously undervalued resource that, compared with surface water, attracts little attention and only minor financial investment.
- Governance of groundwater uses and protection of the aquifers that provide the resource call for different levels and styles of management and regulation than is conventionally used with “visible” surface water.

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Groundwater Governance - A Global Framework for Action

Groundwater Governance - A Global Framework for Action (2011-2014) is a joint project supported by the Global Environment Facility (GEF) and implemented by the Food and Agriculture Organisation of the United Nations (FAO), jointly with UNESCO's International Hydrological Programme (UNESCO-IHP), the International Association of Hydrologists (IAH) and the World Bank.

The project is designed to raise awareness of the importance of groundwater resources for many regions of the world, and identify and promote best practices in groundwater governance as a way to achieve the sustainable management of groundwater resources.

The first phase of the project consists of a review of the global situation of groundwater governance and aims to develop of a Global Groundwater Diagnostic that integrates regional and country experiences with prospects for the future. This first phase builds on a series of case studies, thematic papers and five regional consultations.

Twelve thematic papers have thus been prepared to synthesize the current knowledge and experience concerning key economic, policy, institutional, environmental and technical aspects of groundwater management, and address emerging issues and innovative approaches. The 12 thematic papers are listed below and are available on the project website along with a Synthesis Report on Groundwater Governance that compiles the results of the case studies and the thematic papers.

The second phase of the project will develop the main project outcome, a Global Framework for Action consisting of a set of policy and institutional guidelines, recommendations and best practices designed to improve groundwater management at country/local level, and groundwater governance at local, national and transboundary levels.

Thematic Papers

- No.1 - Trends in groundwater pollution; trends in loss of groundwater quality and related aquifers services
- No.2 - Conjunctive use and management of groundwater and surface water
- No.3 - Urban-rural tensions; opportunities for co-management
- No.4 - Management of recharge / discharge processes and aquifer equilibrium states
- No.5 - Groundwater policy and governance
- No.6 - Legal framework for sustainable groundwater governance
- No.7 - Trends in local groundwater management institutions / user partnerships
- No.8 - Social adoption of groundwater pumping technology and the development of groundwater cultures: governance at the point of abstraction
- No.9 - Macro-economic trends that influence demand for groundwater and related aquifer services
- No. 10 - Governance of the subsurface and groundwater frontier
- No.11 - Political economy of groundwater governance
- No.12 - Groundwater and climate change adaptation

Figure 124 - Groundwater Governance - A Global Framework for Action (2011–2014) was a project designed to raise awareness of the importance of groundwater resources for many regions of the world and to identify and promote best practices in groundwater governance to achieve the sustainable management of groundwater resources (Howard, 2012).

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The overriding objective of this global project was to develop a Framework of Action (i.e., a menu of region-specific policy options) of global relevance, whose application would enable sound governance of groundwater resources and facilitate the promotion of local actions for their improved management and protection. Within this objective, the project aspired to build a broad coalition that would foster change, support policy reforms, and promote sustainable groundwater management to promote alternative approaches to existing groundwater use and contribute in a major way to resolution of global water challenges.

The project was a massive undertaking and generated global interest and participation. It included five Regional Consultations (RCs) led by UNESCO - IHP, involved 479 participants representing 90 countries, and included three case studies conducted by the World Bank (India, South Africa, and Kenya). Physical outputs included 12 peer-reviewed thematic papers (listed in Figure 124), a Global Diagnostic (GD) (Food and Agriculture Organization of the United Nations (FAO), 2015a), a Vision for Groundwater Governance, and a Framework for Action (FAO, 2015b). Various spin-off papers have also been published in the peer-reviewed literature, for example, Howard (2015) and Foster and van der Gun (2016).

In practice, the true success of the project will not be known for decades. It is one thing to have a Framework for Action; to follow through requires political will and strong financial support. As noted by Foster and van der Gun (2016), groundwater governance is still in its infancy in many countries of the world, and it will be a long-term process. Expectations for what can be achieved should be realistic, and incremental advances are preferable as it will take decades before the success of the project is achieved. The authors go on to say that worldwide strengthening of groundwater governance represents a real challenge for hydrogeologists to widen their vision and activity beyond their narrow scientific discipline.

In terms of this book on urban groundwater, it should be remembered that the GEF Project was not limited to urban areas but dealt with governance as it related to groundwater throughout the world—that is, it also had to contend with groundwater for agricultural use, fossil groundwater, mining and energy, transboundary aquifers, groundwater dependent ecosystems, and climate change impacts. In fact, these issues were essentially uncharted with respect to effective groundwater governance.

Urban groundwater is no less complex than any other global groundwater issue. However, when it comes to the governance of urban groundwater, much can be drawn from the experience of the World Bank's Groundwater Management Advisory Team (GW-MATE) that, for over a decade, conducted considerable work on the protection and management of groundwater in urban areas and the role good governance can play. Examples are the work by Foster and others (2010a, 2010b, 2010c), together with their contributions to the peer-reviewed journal literature (e.g., Kemper 2004; Foster et al., 2011). GW-MATE has provided valuable blueprints for the development of resource management

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action plans together with the governance provisions required for their effective implementation. Much of the following section is taken from GW-MATE's work.

7.4 A Framework for Water Governance in Urban Areas

Groundwater is a classical common pool resource, a term used to describe a natural resource that by virtue of its size and/or characteristics makes it virtually impossible to exclude potential beneficiaries from its use. According to Hardin (1968), common pool resources are vulnerable to the "tragedy of the commons" whereby current and potential stakeholders behave solely in terms of their personal, short-term interests rather than consider the long-term requirements of the community. While Hardin suggested that the tragedy of the commons could be avoided by more government regulation or by privatizing the commons property, Elinor Ostrom (Figure 125) argued that simply passing control of local areas to national and international regulators could create even greater problems (Ostrom,1990, 2005).

Figure 125 - Elinor Ostrom, a political scientist at Indiana University, shared the Nobel Prize in Economics in 2009 for her lifetime work investigating how communities succeed or fail at managing finite common pool resources such as forests, grazing land, and water (from https://www.onthecommons.org/magazine/ostrom%e2%80%99s-nobel-prize-milestone-commons-movement/↗).

Ostrom believed the tragedy of the commons was not as pervasive or as difficult to solve as Hardin had suggested, since the instinct amongst locals is to resolve the commons problem themselves; when the commons is taken over by national regulators, this approach is no longer viable. Ostrom's work (Ostrom, 1990, 2005, 2009a, 2009b) created a series of

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design principles—the Ostrom Principles—that would provide for stable, local management of the common pool.

In terms of groundwater governance, the Ostrom Principles provide a practical guide to cooperative groundwater management and are especially appropriate for urban settings given the large number of potentially competing stakeholders. Foster and others (2010a) suggest that based on the Ostrom principles, the management approach would involve

- clearly defined boundaries for evaluating and allocating the resource, and congruence with prevailing local conditions and constraints;
- formal recognition by government of the water management rights of the community;
- collective arrangements for decision making;
- layers of nested stakeholder groups to cope with larger resource systems;
- effective monitoring with stakeholder involvement;
- graduated sanctions on users who do not comply with communal rules; and
- low-cost, efficient conflict-resolution mechanisms.

The nature of groundwater also means that links with the governance of the environment and other land and water resources are highly relevant. As an example, groundwater resources are highly dependent upon land use in aquifer recharge areas; any changes in this land use can significantly affect both the quantity and quality of recharge. Thus, groundwater governance cannot be addressed without due consideration for the processes that determine or control land use. In urban areas, land use classification and control are generally the domain of municipal or local government. The absence of mechanisms that allow water resource agencies to influence the process is an governance weakness.

In a similar way, the groundwater supply for many cities is imported from peri-urban well fields such that rural land use practices and the intensification of agricultural production (often determined by national agriculture and food policy agencies) exert a very strong influence on groundwater recharge rates and the quality of the water. The inability of water resource agencies to influence rural land use practices is another governance weakness. In effect, and as discussed by Foster and van der Gun (2016), identifying and developing appropriate functional linkages, often across various levels of government (Figure 126), is a key prerequisite for strengthening governance in a global sense but is particularly valid in urban areas where multiple levels of government frequently exist.

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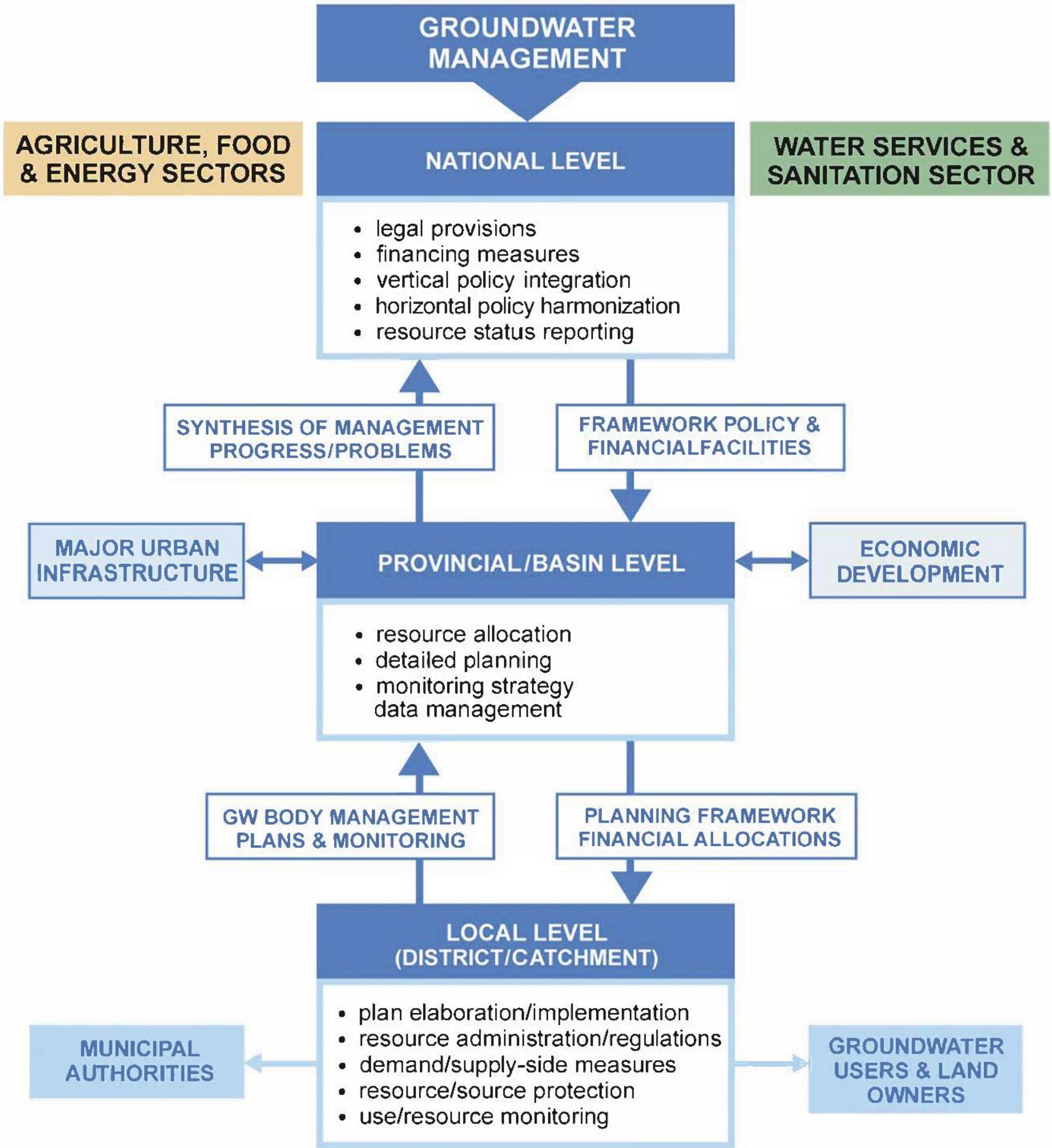

Figure 126 - Broad scheme of governance provisions required for vertical integration and horizontal coordination of groundwater management (modified after Foster & van der Gun, 2016).

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A GW-MATE blueprint for the development and implementation of an action plan for sustainable groundwater management is shown in Figure 127 with governance provisions highlighted (Foster et al., 2010a).

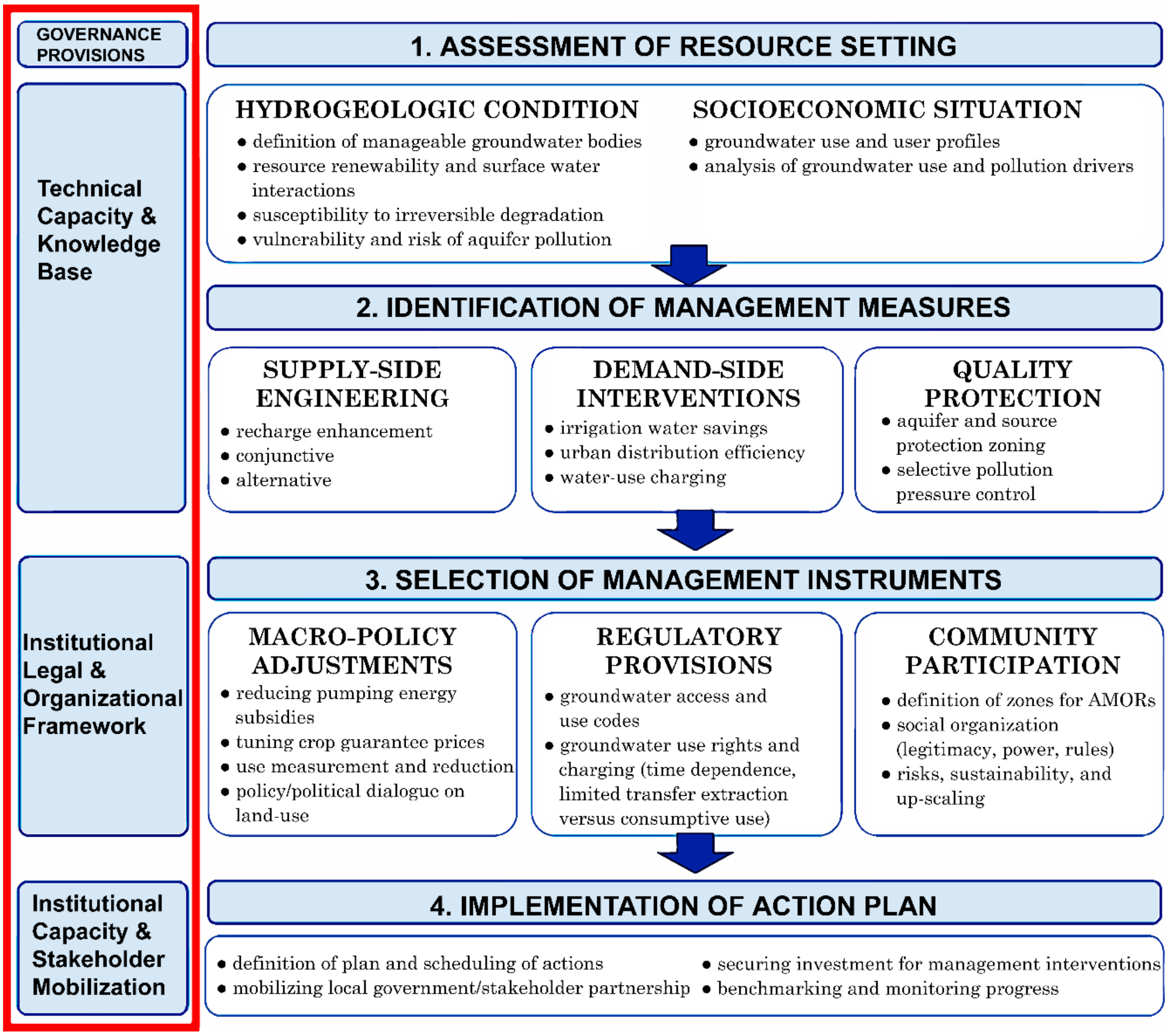

Figure 127 - A framework for the development and implementation of a groundwater management action plan, with corresponding governance provisions (modified after Foster et al., 2010a).

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A checklist of 20 key benchmarking criteria for the evaluation of groundwater governance provision and capacity is provided in Table 32.

Table 32 - Checklist of 20 key benchmarking criteria for the evaluation of groundwater governance provision and capacity (modified after Foster et al., 2010a).

Type of Governance Provision / Capacity	Check list			
	#	**Type***	**Criterion**	**Context**
Technical	1	C	Existence of Basic Geological and Hydrogeological Maps	For identification of groundwater resources with classification of typology
	2	C	Groundwater Body/Aquifer Delineation	
	3	A	Groundwater Piezometric Monitoring Network	To establish resource status
	4	B	Groundwater Pollution Hazard Assessment	For identifying quality degradation risks
	5	A	Availability of Aquifer Numerical Management Models	At least for strategically critical aquifers
	6	B	Groundwater Quality Monitoring Network	To detect groundwater pollution
Legal and Institutional	7	A	Water Well Drilling Permits and Groundwater Use Rights	For large users, with needs of small users noted
	8	A	Instrument to Reduce Groundwater Abstraction	Water well closure or constraint in critical areas e.g., overexploited or polluted areas
	9	A	Instrument to Prevent Water Well Operation	
	10	A	Sanction for illegal Water Well Operation	Penalizing excessive pumping above permit
	11	A	Groundwater Abstraction and Use Charging	Resource tariff on larger users
	12	B	Land-Use Control on Potentially-Polluting Activities	Prohibition or restriction since a potential groundwater hazard
	13	B	Levies on Generation/Discharge of Potential Pollutants	Providing incentive for pollution prevention
	14	C	Government Agency as 'Groundwater Resource Guardian'	Empowered to act on cross-sectoral basis
	15	C	Community Aquifer Management Organisations	Mobilizing and formalizing community participation
Cross-Sector Policy Coordination	16	C	Coordination with Agricultural Development	Ensuring real water saving and pollution control
	17	C	Groundwater-Based Urban/Industrial Planning	To conserve and protect groundwater resources
	18	B	Compensation for Groundwater Protection	Related to constraints on land-use activities
Operational	19	C	Public Participation in Groundwater Management	Effective in control of exploitation and pollution
	20	C	Existence of Groundwater Management Action Plan (Figure 127)	With measures and instruments agreed

* A: Groundwater extraction related.
B: Groundwater quality related.
C: Groundwater extraction and quality related.

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While there can be no "one-size-fits all" when it comes to models of sound groundwater governance, Figure 127 provides a viable framework for governance that can be adapted to a wide range of situations including urban areas. Significantly, the starting point is a sound knowledge base and a fundamental quantitative assessment of the resource setting. Assessment of the resource setting needs to include two items:

1. the local hydrogeological conditions (aquifers/aquitards present, their hydraulic properties, aquifer recharge rates, economic reserves, water quality and vulnerability to pollution), and
2. the socio-economic situation (demand, users, and groundwater-use drivers such as well construction costs and energy subsidies).

In turn, a reliable understanding of the resource setting will allow appropriate management measures to be established. These include

- management measures on the supply side (e.g., recharge enhancement and opportunities for conjunctive use),
- management measures on the demand side (e.g., water pricing and distribution controls), and
- management measures to guarantee groundwater quality protection (e.g., aquifer vulnerability mapping and restrictions on potential pollutant sources in defined wellhead protection areas).

A good example of the application of demand-side and supply-side control measures is that of Lima, Peru in the late twentieth century as illustrated by Figure 128. Ultimately, the key to good urban groundwater governance and the successful execution of urban groundwater action plans depends heavily on the institutional framework and how efficiently it functions (Figure 127 and Table 32). Typically, the institutional framework includes departments and agencies at various federal and state levels that would work interactively with regional and city governments. Janakarajan and others (2006) observe there is often a need to redefine responsibilities to better coordinate the various levels of decision making, avoid the overlapping of tasks, and limit the capacity for intervention by discretionary powers.

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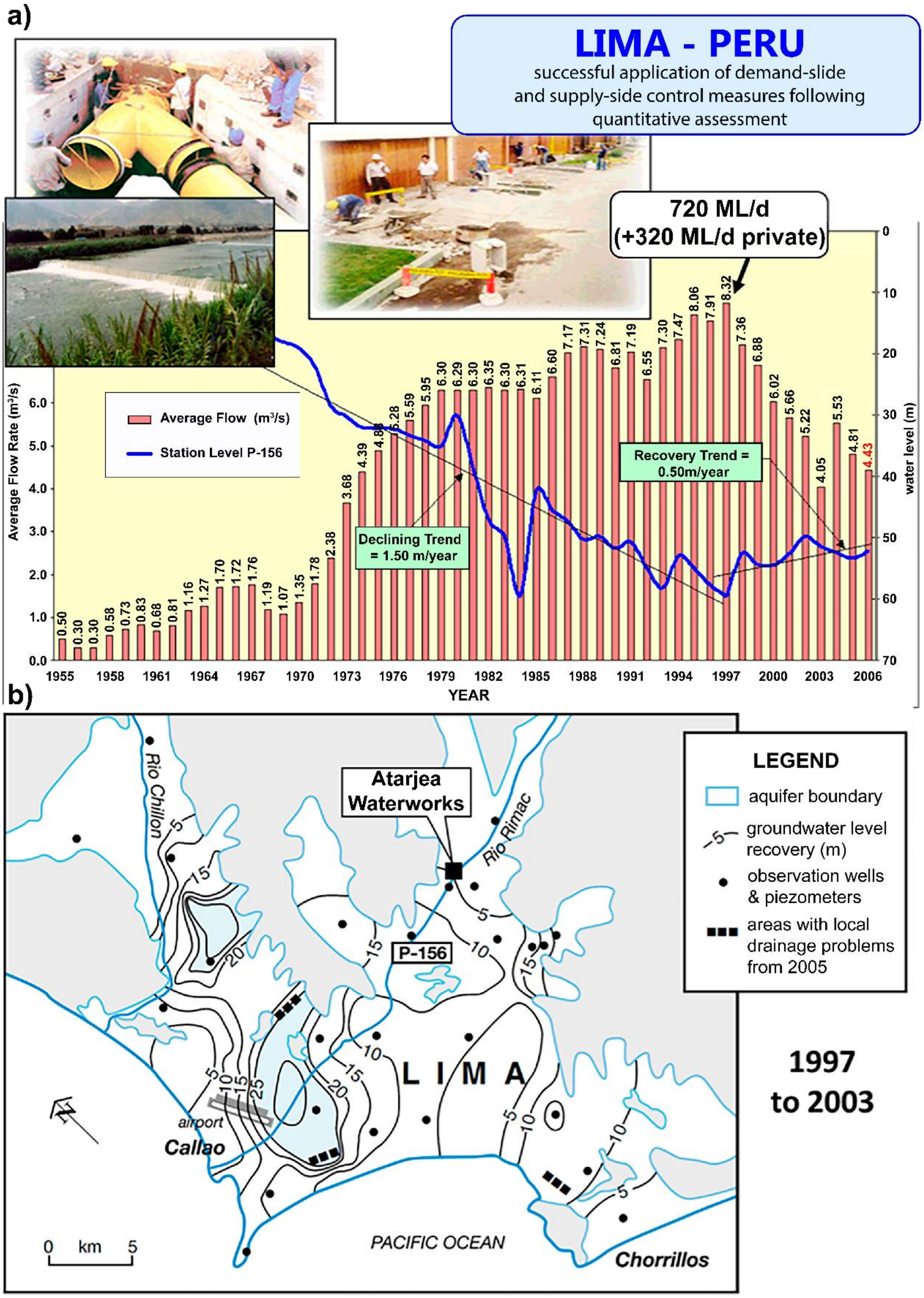

Figure 128 - In Lima, Peru, successful application of demand-side and supply-side control measures (supported by conjunctive use of ground and surface water) led to a) stabilization and steady recovery of groundwater levels, b) as mapped in the Lima area with 5 km/≈3 miles (modified after Foster et al., 2010d).

Many institutional framework models exist; however, based on their extensive operational experience, GW-MATE prefers a decentralized approach to groundwater management that includes very strong participation of stakeholders. It should be noted that while GW-MATE favors decentralization, it believes that federal governments should ensure state/provincial/municipal level agencies receive adequate funds to hire and retain the well-trained professionals required to perform the necessary work.

In fact, numerous studies have shown that stakeholder involvement is an essential instrument of good water governance, especially in urban settings, for three reasons:

1. Top-down management decisions taken unilaterally by government agencies without broad social agreement tend to be impossible to implement. Stakeholders need to feel a sense of ownership for decisions that are made.
2. Essential groundwater management activities such as data collection, policing, and collection of water payments can be carried out more efficiently and economically when performed in a cooperative manner.
3. Involvement of stakeholder groups can strongly facilitate the integration and coordination of decisions that relate to all aspects of land use, the groundwater resource, and the management of waste.

Recognizing that groundwater imported from rural and peri-urban wellfields is frequently a source of serious urban-rural tension (Howard, 2012), promoting close cooperation between all stakeholders must be a high priority. This includes urban, peri-urban, and rural users.

Finally, in support of its blueprint for the development and implementation of an action plan for groundwater management (Figure 127), GW-MATE recommends a series of policy actions that, if followed, can help preserve the physical sustainability of the groundwater resource (Foster et al., 2010b). The policy actions include

- definition of areas with critical levels of resource exploitation as a basis for restricting further development of the urban groundwater resource;
- establishing clear criteria by area for issuing of permits for pumping (in terms of safe well separation and maximum rates of extraction);
- exerting close control over both municipal and private groundwater pumping based on defined areas—including, where necessary, the relocation of municipal wells, increased resource-use tariffs, and even the closure of private (personal) wells;
- maximizing use of aquifer recharge enhancement techniques (MAR - Managed Aquifer Recharge), taking care to avoid the introduction of potential groundwater pollutants; and,
- monitoring and periodic evaluation of groundwater resource status (quality and quantity), including the application of numerical aquifer simulation models.

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Furthermore, they recommend greater integration of urban water supply, land use, and provision of mains sewerage to prevent persistent and potentially costly problems. The types of measures required include

- giving priority for mains sewerage installation to recently urbanized areas that overlie aquifers with good quality groundwater, thus affording protection from water quality degradation;
- utilizing urban allocations of parkland or low-density housing areas to establish municipal source protection and/or exclusion zones around any municipal wells that are favorably located;
- assessing the sanitary protection standards of municipal wells, the risks of wellhead contamination, and how they can be reduced;
- undertaking groundwater pollution vulnerability mapping and hazard assessment, recognizing some municipal wells may need to be abandoned where the contamination risk by toxic chemicals is high; and
- preventing the creation of pollutant discharges in upstream areas of municipal wells that could infiltrate and compromise well water quality.

In lower-income countries, a common and increasing problem is that most private urban water wells are either illegal or unregulated, a situation that is counterproductive for both the private user and the local administration. In many cases, the most beneficial course of action would be to legalize such wells (e.g., Figure 129).

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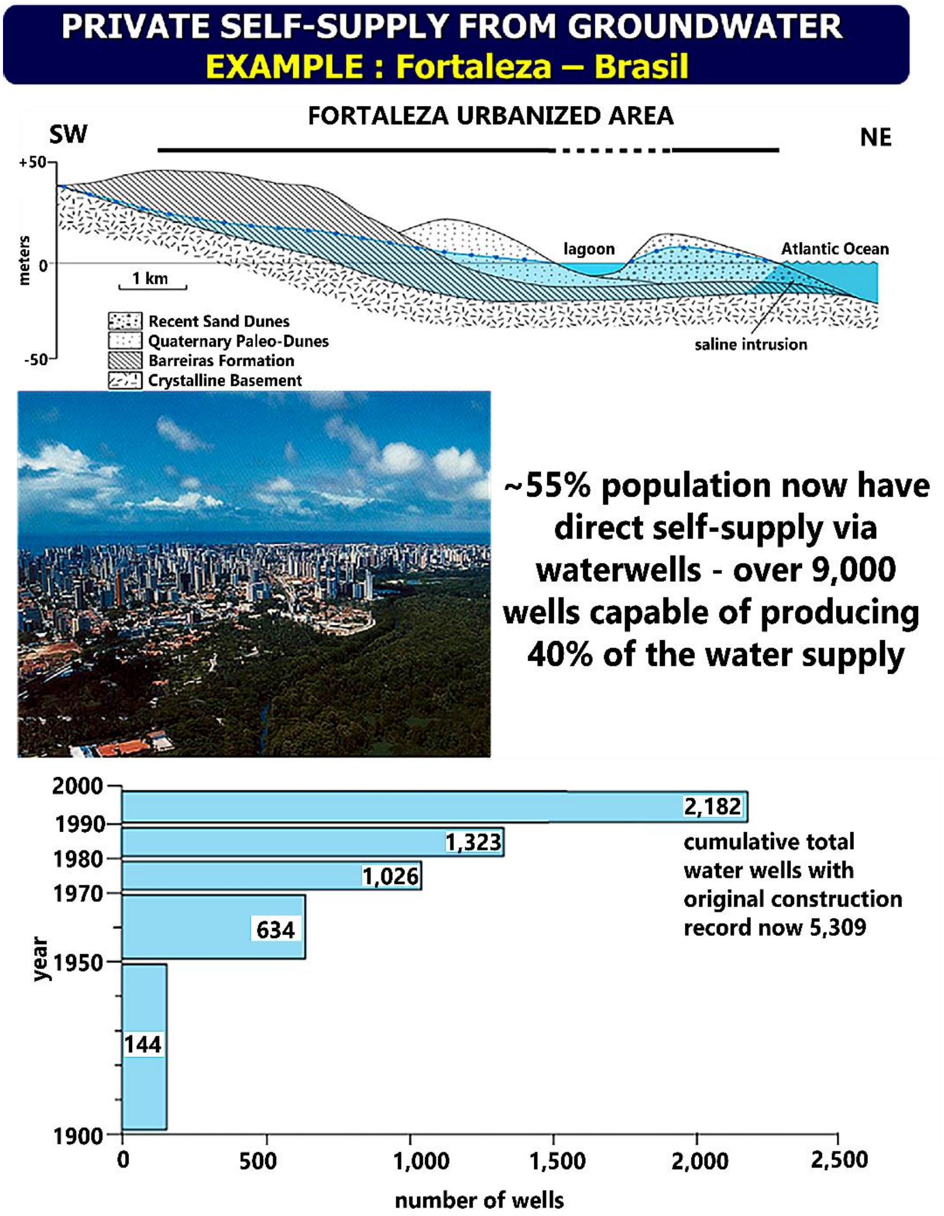

Figure 129 - Attempts to regularize the private use of urban groundwater have been made in places (e.g., Fortaleza in Brazil) where the municipal utility has argued for the levying of a volumetric water charge to reflect mains sewer use by private groundwater abstractors (Modified after Foster et al., 2006).

Legalizing private urban water wells potentially allows

- users to receive sound advice (re: pollution risks/alerts, necessary precautions, etc.) and enjoy protection against potential impacts associated with excessive pumping and/or well interference;
- sanitary completion standards for improved and potential interaction, with in situ sanitation units (e.g., latrines, cesspools, septic tanks) minimized or prevented;
- local administrators to secure data on private well use and establish better relationships with private users;

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- opportunities to collect monitoring data for water levels and groundwater quality; and
- opportunities to collect tariffs for the use and disposal of water.

Importantly, water supply security can be significantly enhanced by adopting an adaptive approach to water resource management (Gleeson et al., 2012) that allows decision makers to adjust and fine tune management plans as more is learned about the aquifer and its behavior. This is called the adaptive management plan (AMP) approach. The AMP should never be used to avoid the need for a full and thorough scientific investigation of the aquifer system but, instead, should be seen as an opportunity to refine a fundamentally robust management plan as more data on groundwater levels and quality trends become available.

The best aquifer management plans are underpinned by a transient numerical aquifer model as a decision support tool that is calibrated and later verified with groundwater abstraction and drawdown data. This model would be used to explore management options, including prospects for conjunctive use (Paling, 1984; Foster et al., 2010d), as shown in Figure 130 and Figure 131. Such models could also be used to support AMPs.

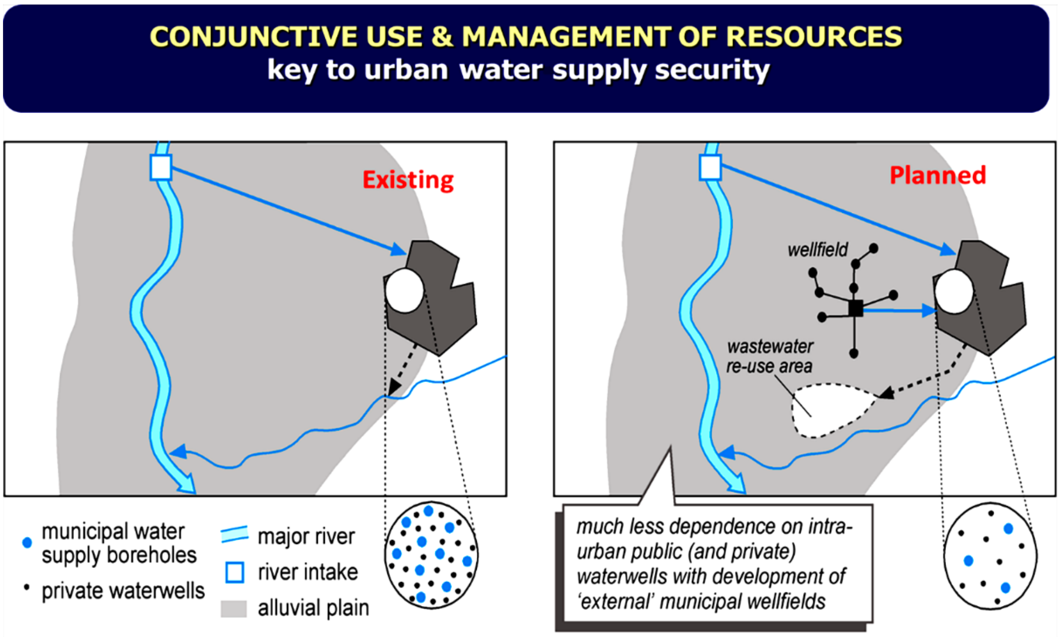

Figure 130 - Example of a conjunctive use scheme for urban water supply (modified after Foster et al., 2010d).

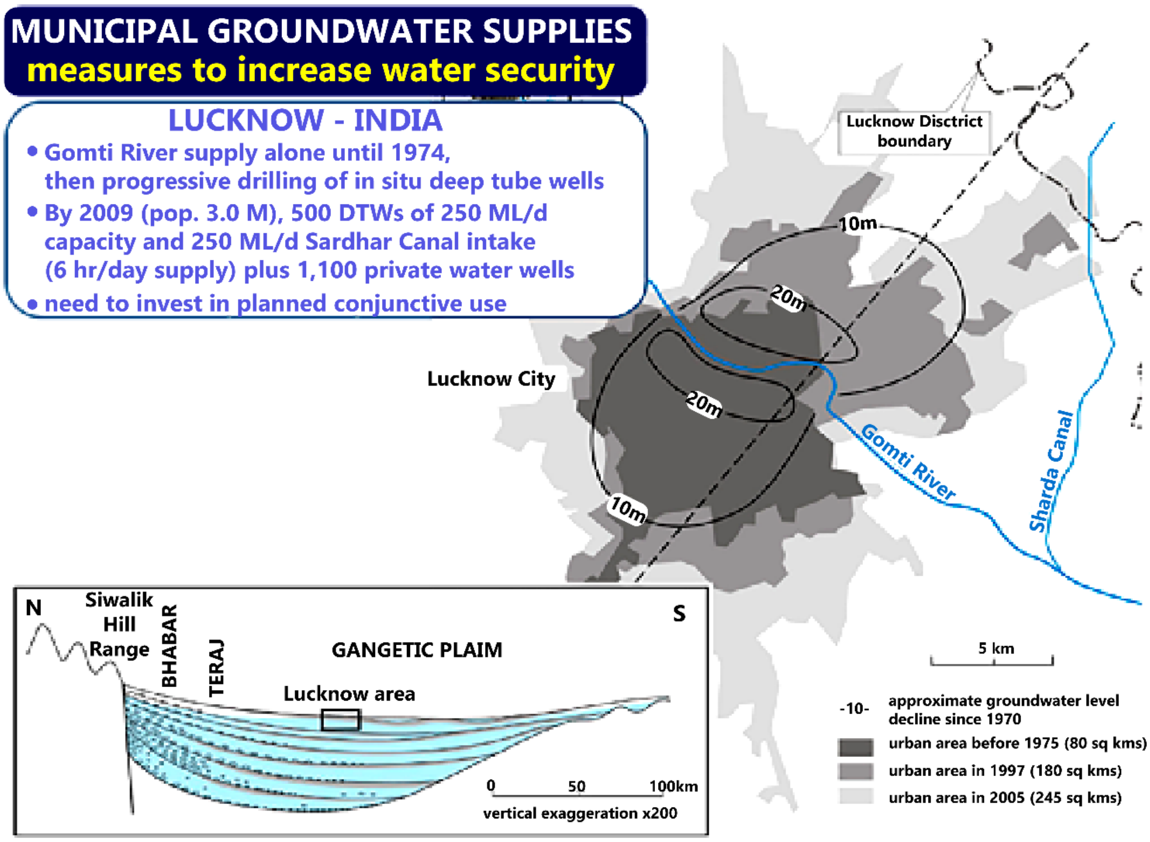

Figure 131 - Lucknow, India, began a conjunctive use scheme in 1973 but needs to invest further in its development. 100 km ≈62 miles, 10 m ≈6.2 miles (modified after Foster et al., 2010d).

7.5 Exercises Related to Section 7

Exercises related to Section 7 are available at this link↴.

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8 Key Takeaways and Priority Data Needs

Large cities drive the world's economy, but serious doubts have been raised concerning their long-term sustainability, given rapid urban population growth in many parts of the world and an escalating demand for water. Sustainable cities require sustainable water supplies, and many cities regularly report severe water shortages. Compounding the problem is the unpredictability of climate change, which makes resource planning extremely difficult.

The only certainty is groundwater, representing over 97 percent of the world's fresh, readily accessible water, which must and will play a major role as cities across the globe increasingly address water supply issues. Sensibly managed—and with the support of sound science and new, creative technologies—groundwater resources will be key if the world's urban sustainability challenge is to be resolved.

The gravity of the problem cannot be overemphasized. Urban growth, industrial development, and irrigated agriculture have always imposed an enormous strain on water resources. While increased demand for water is the primary concern, the problem is exacerbated in many cities by the release of urban-sourced contaminants that compromise water quality and thereby seriously limit its utility. Moreover, urban development and the intensification of infrastructure such as buildings, roads, and tunnels seriously impact the entire urban water balance, causing profound changes to key hydrologic components such as evapotranspiration, surface runoff, and aquifer recharge. The overall effect can be to change groundwater flow volumes and directions that fundamentally affect the nature and degree of interaction between water bodies—groundwater, river water, lakes, wetlands, and the oceans. Understanding such changes and managing their effects continues to be a difficult task for practitioners working with urban groundwater.

Unfortunately, much of the world's urban growth occurs in low- and middle- income countries where relatively prosperous, well-serviced urban cores are often surrounded by an expanding sprawl of underserviced suburbs and peri-urban slums. In many cases, suburban and peri-urban areas support most of the urban population but are seriously neglected when it comes to proactive land use planning and the provision of adequate sanitation and drinking-water services. Frequently, developed-world solutions for dealing with growing cities are not readily transferable to less-developed countries where there is often a serious disconnect between water managers and urban planners. In many cases, there is no urban planning at all. Without the planning of urban space and infrastructure, opportunities to provide adequate water and sanitation services are seriously compromised; the classic centralized approach to delivering water services will often fail.

Despite the daunting scale of the problem, progress continues to be made, and there are signs the urban sustainability challenge should be viewed with cautious optimism, particularly with respect to the role of groundwater. The science behind urban

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groundwater is very strong, and there is a growing recognition that urban water needs to be managed holistically by considering the urban water cycle in its entirety. While such principles have been embedded for decades in IWRM (Integrated Water Resource Management) and IUWM (Integrated Urban Water Management), the role of groundwater has frequently been ignored.

This needs to change, and there are early signs that it will, especially in low- and middle-income countries where urban water stress is most evident. Good progress has been made in recent years on the importance of responsible water governance, most notably with respect to groundwater, and there is growing appreciation of the benefits of developing and implementing groundwater resource management plans rooted in sound science and including a strong quantitative understanding of the hydrogeologic setting—yet fully involving the interests of all stakeholders in decision making and implementation.

Time will tell if progress on urban water governance can be maintained. Certainly, the science is well advanced, and urban water managers have all the tools they need to move forward. If there is one major impediment, it can be summed up in one word: DATA. The management of urban groundwater is currently reactive and needs to become proactive. Proactive management of groundwater will never become a reality without comprehensive, good quality data. However, for most the world's urban areas, including those in high-income countries, reliable data are in very short supply.

There is an urgent need for cities to make greater use of advanced urban water modeling tools to provide proactive guidance on the management of urban water resources, especially groundwater. Where used, such tools can prove invaluable for understanding water quality and water quantity changes over time, including the potential effects of changing climate and land use (e.g., urban growth). However, without good data to calibrate these models and provide baseline information, this opportunity is entirely lost. The models are available and are based on good science, but they need to be adopted and fully utilized. Essential data needs begin with sound geological information and a thorough knowledge of the urban infrastructure (including information on its location, geometry, and materials involved). Also critical are a reliable urban water balance and comprehensive records of potentiometric levels, contaminant sources, and groundwater quality.

8.1 The Urban Aquifer Framework

All studies must begin with a sound understanding of the geology and the extent to which it has been modified by urban development and related infrastructure. The geometry of the system must be reliably established together with a full understanding of the materials involved. Deep foundations and similarly engineered structures require mapping in detail. Without this comprehensive range of knowledge, it is impossible to resolve the three-dimensional nature of the urban aquifer system and determine likely directions of groundwater flow.

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8.2 The Urban Water Balance

Urban development can radically change the water cycle and introduce new water balance components, many of which are not well quantified. From a groundwater perspective, urgent data needs for most cities include:

- monthly or annual estimates of water withdrawals (municipal use, industrial use, private use, and groundwater removed for construction dewatering);
- sewer exfiltration and infiltration rates;
- exfiltration and infiltration rates for rail and vehicle tunnels;
- leakage rates from pressurized water supply networks;
- water level and streamflow monitoring to assess inequities in the water balance and provide essential input to urban groundwater modeling; and
- stormwater infiltration rates, especially where stormwater is managed by releasing excess surface water to the shallow subsurface, often with the assistance of purportedly green infrastructure.

8.3 Potentiometric Levels

Monitoring of groundwater levels, where it takes place, tends to focus on aquifers used for supply. Within city limits, this normally means deeper confined aquifers where water quality is often preferable.

For proactive management of groundwater, water levels need to be monitored in all aquifer units, thus allowing the potential for the transfer of water from one aquifer unit to another to be identified and estimated. This is particularly important for urban multi-layer aquifers where contaminated groundwater that is often present in the uppermost aquifer threatens water quality in lower aquifers.

8.4 Contaminant Sources

Potential sources of groundwater contamination in urban areas are often known but rarely quantified. Chemical audits documenting the mass of contaminants released to the subsurface in urban areas should be obligatory for all urban areas. Commonly lacking is reliable quantitative information on septic system discharge and leakage from sewer pipes, landfills, and USTs.

8.5 Groundwater Quality

The quality of urban groundwater is rarely monitored, especially in cities where urban aquifers are not used for supply. To provide an early warning of impending threat and as an input to predictive modeling tools, urban groundwater requires comprehensive monitoring with a focus on shallow groundwater and, where possible, the unsaturated zone. Monitoring should be intensified in the vicinity of potential sources of contamination.

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This includes coastal areas, where sentry monitoring wells are essential for controlling the risk of seawater intrusion.

The availability of good, reliable data will guarantee the best possible decisions can be made for cities that rely on groundwater for all or part of their water supply. However, in the interests of protecting groundwater dependent ecosystems (GDEs) and preventing the flooding of tunnels, underground parking facilities, and domestic basements with groundwater—frequently polluted by urban-sourced contaminants—such data are also essential for cities wholly reliant on surface water. Urban groundwater needs to be managed proactively, whether it is used as a resource or not.

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9 Exercises

9.1 Exercises Related to Section 2

9.1.1 Exercise 1

What are the two main influences on the water table level during the evolution of a city? Describe how they influence the water table and groundwater quality underneath the city in the four phases of development described in Figure 6.

Click for Solution to Exercise 1⤵

9.1.2 Exercise 2

Compare urban and rural hydrogeology (i.e., in what major ways they are similar and different?). Describe four characteristics (one from each theme) unique to urban hydrogeology and how they affect groundwater.

Click for Solution to Exercise 2⤵

Return to Where Text Linked to Exercises for Section 2⤴

9.2 Exercises Related to Section 3

9.2.1 Exercise 3

a. Describe direct and indirect recharge.
b. What is impervious cover and how does it contribute to recharge? Provide examples.
c. What are some strategies to promote indirect recharge? Describe the risks and important considerations when promoting indirect recharge.
d. Consider Figure 17 and estimate the percent increase in total recharge compared to natural recharge for Buenos Aires.

Click for Solution to Exercise 3⤵

9.2.2 Exercise 4

Describe urban karst: What is it? What does it influence?

Click for Solution to Exercise 4⤵

Return to Where Text Linked to Exercises for Section 3⤴

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9.3 Exercises Related to Section 4

9.3.1 Exercise 5

a. What is a point source of contamination? How does it differ from distributed sources of contamination?

b. Classify the following contamination sources as point, line, or distributed:

- Septic surface sewage canal
- Concentrated pesticide application (e.g., from aerial spraying)
- High density of septic systems
- Landfill leachate

Click for Solution to Exercise 5⤵

9.3.2 Exercise 6

Complete the following table:

Contamination Source	Main contaminants (inorganic or organic? One example contaminant)	Risks/Concerns (mobility, toxicity)
Underground storage tanks		
Road de-icing		
Stormwater (urban environments)		

Click for Solution to Exercise 6⤵

Return to Where Text Linked to Exercises for Section 4⤴

9.4 Exercises Related to Section 5

9.4.1 Exercise 7

What are the three main global challenges related to groundwater faced by cities throughout the world. Provide some examples of cities that face these challenges.

Click for Solution to Exercise 7⤵

9.4.2 Exercise 8

a. What is the saline water wedge, and what are the two main factors that largely define the position of the saline water wedge?

b. What is an overdeveloped aquifer and how does it affect the saline water wedge?

c. What strategy was applied to stop the advance of seawater in Los Angeles (California), USA?

Click for Solution to Exercise 8⤵

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9.5 Exercises Related to Section 6

9.5.1 Exercise 9

What are the three essential elements needed to meet the world's increasing demand for urban water supplies? What are some practical measures to achieve them?

Click for Solution to Exercise 9⤵

9.5.2 Exercise 10

What are the advantages and disadvantages of UGROW compared with AISUWRS?

Click for Solution to Exercise 10⤵

Return to Where Text Linked to Exercises for Section 6⤴

9.6 Exercises Related to Section 7

9.6.1 Exercise 11

What is the failure of IWRM/IWM/IUWM?

Click for Solution to Exercise 11⤵

9.6.2 Exercise 12

Why is stakeholder involvement considered an essential instrument of good water governance, especially in urban settings?

Click for Solution to Exercise 12⤵

Return to Where Text Linked to Exercises for Section 7⤴

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11 Exercise Solutions

11.1 Exercise Solutions Related to Section 2

11.1.1 Exercise Solution 1

Two influences: water demand and urban pollution

Phase 1: Water table **lowers** due to pumping in shallow wells; wastewater discharged to ground.

Phase 2: Water table **lowers** due to increased pumping; groundwater pumped from deeper wells; wastewater discharged to ground causing pollution of shallow groundwater.

Phase 3: Pumping outside of city and wastewater discharge causes water table to **rise;** water table lowers at periphery due to increased pumping in semi-urban areas.

Phase 4: Water table continues to **rise** due to continued wastewater discharge and lack of pumping in city (pumping outside of the city/water imports).

Return to Exercise 1↥

11.1.2 Exercise Solution 2

The hydrogeological and hydrochemical processes in urban and rural hydrogeological systems are mainly the same. What makes urban systems unique are the urban features that influence the groundwater flow system and chemistry of the groundwater. The same is true for rural systems. While urban features can be found in rural systems, they usually have only minor effects in rural systems (and vice versa).

Four characteristics and their effects are found in Table 2 (choose one from each theme).

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Return to Where Text Linked to Exercises for Section 2↥

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11.2 Exercise Solutions Related to Section 3

11.2.1 Exercise Solution 3

a. **Direct recharge:** Water that infiltrates into the soil immediately following its arrival as incident precipitation and passes directly to the underlying aquifer.
   Indirect Recharge: Normally occurs when water infiltrates following transport across the land surface as runoff. It most commonly occurs in vegetated swales, stream channels, or beneath shallow temporary ponds that form following intense periods of rain.

b. **Impervious cover:** Material overlying natural cover (e.g., soil) that does not allow water to infiltrate. Impermeable cover includes roofs, sealed roads, parking lots, and pedestrian pathways. In practice, it may also include sports fields and parks where underground drains have been installed to intercept infiltrating water. Large areas of impermeable cover will cause a significant rise in the surface water runoff component.

c. **Strategies to promote indirect recharge:**
   i) downspout disconnection programs to divert water destined to enter storm sewers,
   ii) permeable pavement,
   iii) the use of soakaways and recharge basins to allow drainage to the sub-surface with minimal evaporation, and
   iv) Use injection wells to recharge stormwater into underlying aquifers.

   Risks:
   i) production of urban contaminants,
   ii) clogging of infiltration lagoons, recharge wells, and the aquifer, and
   iii) excess recharge leading to rising groundwater levels (e.g., flushing of contaminants).

   Important considerations:
   i) estimating the amount of water entering the aquifer via indirect recharge relative to the capacity of the aquifer, which can be difficult, and
   ii) estimating the amount of recharge that reaches the aquifer via "sealed" surfaces.

d. **Percent Increase in recharge in Buenos Aires:**
   Natural recharge: ≈660 mm/a
   Total recharge: ≈3100 mm/a
   Difference between Total and Natural recharge = 2440 mm/a
   Percent increase = (2440/660) 100 percent ≅ 370 percent increase

Return to Exercise 3↥

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11.2.2 Exercise Solution 4

Urban karst is secondary permeability that develops in association with subsurface construction activities. Any enhancement of permeability or storage created by urbanization, e.g., trenches and drains lined with permeable material.

Urban karst influences the water balance and how the water table responds to changes in the water balance because it increases the storage space available in the aquifer. It can also influence the rate and direction of contaminant transport as well as the nature by which urban-sourced chemicals are released to urban streams.

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11.3 Exercise Solutions Related to Section 4

11.3.1 Exercise Solution 5

Point source: Originates at a specific location and significantly impacts groundwater quality close to source; impact is constrained to the contaminant plume that results. **Distributed source:** High density of point and line sources combine to pollute the resource over a large area and can be a more serious threat to an aquifer than a point source.

- Septic surface sewage canal - line
- Concentrated pesticide application (e.g., from aerial spraying) - point
- High density of septic systems - distributed
- Landfill leachate – point

Return to Exercise 5↥

11.3.2 Exercise Solution 6

Contamination Source	Main contaminants (inorganic or organic? One example contaminant)	Risks/Concerns (mobility, toxicity)
Underground Storage Tanks	Organic, e.g., gasoline hydrocarbons (such as BTEX)	Mobile, can migrate as an LNAPL or contaminant plume, threat to human health
Road de-icing	Inorganic (e.g., Cl, Na, Mg, K, or other salts from Table 17↥)	Cl = high mobility, can directly impact water quality (e.g., Cl limits are based on taste, Na is linked to hypertension and hypernatremia) or indirectly by mobilizing trace elements such as cadmium, copper, lead, and zinc
Stormwater (urban environments)	Both (any examples included in Table 20↥)	Variable due to variable contaminants; contaminants tend to be mobilized during storm events, issues related to physical habitat changes, water quality changes, public health concerns and aesthetic degradation

Return to Exercise 6↥

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11.4 Exercise Solutions Related to Section 5

11.4.1 Exercise Solution 7

Three main groundwater challenges for cities throughout the world

i) finding sustainable supply for ever growing megacities and peri-urban areas (e.g., New Delhi, India)
ii) land subsidence due to over-development (e.g., Mexico City, Mexico; Tokyo, Japan; and Bangkok, Thailand)
iii) coastal cities face the threat of sea-level rise and seawater intrusion (e.g., Los Angles, California and Chennai, India)

Return to Exercise 7↥

11.4.2 Exercise Solution 8

a. The saline water wedge (Figure 62a) is normally separated from the freshwater body by a narrow transition zone of variable density. Its position is largely defined by the **freshwater potential** and **local hydraulic gradient**.
b. An aquifer is overdeveloped when freshwater pumping rates exceed natural aquifer recharge rates. An overdeveloped aquifer can result in a lowering of the freshwater potential throughout the area with progressive and extensive invasion of the aquifer by seawater (Figure 62c↥).
c. Sets of closely spaced injection wells were drilled to inject high-quality freshwater into the aquifer, thus creating hydraulic pressure ridges or “barriers” (Figure 63↥).

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11.5 Exercise Solutions Related to Section

11.5.1 Exercise Solution 9

Three essential elements needed to meet the world's increasing demand for urban water supplies (followed by some measures that can achieve them) are:

a. Increasing urban water supply
   i) new groundwater resources and resource mining
   ii) offshore reserves
   iii) aquifers beneath lands reclaimed from the sea
   iv) Recharge Management Using Artificial Recharge (MAR)
   v) treatment and desalination
b. Reducing water demand (demand management)
   i) water conservation
   ii) controls on water accessibility
   iii) water pricing—cost structuring
c. Using available water more efficiently
   i) recycling waste water
   ii) capturing and infiltrating storm water

Return to Exercise 9⤴

11.5.2 Exercise Solution 10

Advantages of UGROW compared to AISUWRS:

UGROW is considered as a complete and fully integrated urban water systems model with a dedicated finite element groundwater flow model as part of the modeling package, while AISUWRS uses a coupling approach that ultimately places reliance on an independently developed groundwater flow model.

Disadvantages of UGROW compared to AISUWRS:

UGROW is unable to incorporate more than one aquifer system, while AISUWRS is coupled with a groundwater flow model so it can handle multi-layer aquifer systems.

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11.6 Exercise Solutions Related to Section

11.6.1 Exercise Solution 11

In many applications of IWRM, little recognition or weight is given to the vital function that groundwater plays in the global water cycle and the immense benefits that could be derived from the improved management of groundwater. Groundwater is either ignored or it is lumped together with surface water, even though groundwater and surface water operate on distinctly different scales of time and space.

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11.6.2 Exercise Solution 12

Reasons that stakeholder involvement is essential include the following

a. Top-down management decisions taken unilaterally by government agencies without broad social agreement tend to be impossible to implement; stakeholders need to feel a sense of ownership for decisions that are made.
b. Essential groundwater management activities such as data collection, policing, and collection of water payments can be carried out more efficiently and economically when performed in a cooperative manner.
c. Involvement of stakeholder groups can strongly facilitate the integration and coordination of decisions that relate to all aspects of land use, the groundwater resource, and the management of waste.

Return to Exercise 12↥

Return to Where Text Linked to Exercises for Section 7↥

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12 About the Author

Dr. Ken Howard is Professor of Hydrogeology at the University of Toronto - Scarborough, Canada, where he is Director of the Groundwater Research Group. He has over 45 years of global research experience in all matters related to groundwater but has maintained a special interest in urban groundwater issues. He has published over 130 journal articles and six books. Professionally, he has been certified by the American Institute of Hydrology (PHG) since 1987, chartered by the British Geological Society UK (Chartered Geologist, Fellow of the Geological Society) since 1990, and registered as a Professional Geoscientist with the Association of Professional Geoscientists of Ontario (Professional Geoscientist, Fellow of the Geological Society) since 2002. In 1997, he was elected Chair of the International Association of Hydrogeologists (IAH) Commission on Groundwater in Urban Areas and joined the IAH Board of Directors in 2000 as IAH Vice President for North America. He was elected IAH President in 2012 (four-year term) and continued to serve on the IAH Board of Directors until the end of 2020.

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