Urban effects on groundwater recharge in Austin, Texas

by

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Dedication

To the memory of Bob Goldhammer,

the hugest geologist I have ever met.

Hasta luego Patrón.

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Abstract

Urban effects on groundwater recharge in Austin, Texas

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The University of Texas at Austin, 2004

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Cities and urban populations are growing at a high pace, but groundwater remains an underutilized resource in most urban areas. The general impacts of urban development on groundwater include overexploitation; subsidence; decreasing quality; salt-water intrusion; disruption of ecosystems; variations in the local climate; properties of the soil; natural drainage network; and the quantity, quality, and location of both recharge and discharge. The shallow urban underground is an intricate network of tunnels, conduits, utilities, and other buried structures comparable to a natural karstic system, except that "urban karst" is generated much faster. Urbanization also introduces new sources of water, resulting in an increase of groundwater recharge. These sources include irrigation of parks and lawns, leakage from water mains and sewers, and infiltration structures.

The areal extent of Austin, Texas, has grown steadily since 1885 but has increased five-fold since the 1960's. The difference between the amount of tap water treated in the City of Austin and the amount of sewage that arrives in the wastewater treatment plants (or excess urban water), represents the amount of urban water potentially available for recharge. A water balance shows that about 7% of the treated drinking water is estimated to be lost to leaks from the distribution network and 5% to leaks from sewers. The rest of the excess urban water is used in irrigation of parks and lawns, some of which will be evapotranspired and some will turn into recharge. Smaller fractions are recharged in septic tanks and other designed infiltration devices. Direct recharge from rainfall has decreased as a result of the introduction and expansion of impervious pavements, from 53 mm/a under preurban conditions to 31 mm/a in the year 2000. However urban sources of recharge contribute an average of 85 mm/a of excess urban water, resulting on an urban recharge of 63 mm/a, and a total recharge rate that could equal 94 mm/a.

Several hydrogeochemical parameters were tested as tracers of urban recharge in Austin. Chlorination by-products (trihalomethanes) were found in high concentrations in tap water and in low concentrations in wastewater. However, they were not detected in either surface water courses or groundwater. $\delta^{15}N$ is a commonly used indicator of leakage from sewers, but unusually low values were obtained. Finally, $^{87}Sr/^{86}Sr$ of dissolved strontium shows a strong trend that can be related to the degree of urbanization over the Barton Springs segment of the Edwards aquifer. Values of this ratio from the lesser urbanized wells indicate groundwaters close to equilibrium with the limestone, while samples from the more urbanized wells show higher values, which are closer to those of tap water.

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Chapter 1: Introduction

The impact of humans upon their environment has been long recognized (i.e., Sherlock, 1922; Legget, 1969; Hooke, 2000; Heiken et al., 2003). In fact, the magnitude of these impacts makes humans the major geologic agent on the land surface of the planet and anthropic effects are very severe where population concentrates, that is in urban areas. In 1900, only 10% of the world's population lived in cities (United Nations, 1991) compared to 50% today (United Nations, 2002). Figure 1.1 reveals that currently over 75% of the population of the more industrialized areas of the world are urban dwellers, compared to 40% in the less developed areas. Furthermore, in many instances, the rate of areal-growth for residential and commercial purposes is faster than the rate of population growth, a phenomenon known as "urban sprawl" (Office of Technology Assessment, 1995).

URBAN DEVELOPMENT AND THE WATER CYCLE

Urban development modifies the climate; the land surface and subsurface; and the quantity, quality, and regime of surface water and groundwater.

The covering and replacement of natural rocks, soils, and vegetation by pavements, foundations, builidings, metallic structures, dams, tunnels, and other structures have had, and will continue to have, profound impacts on the water cycle and hydrology of an area.

Because of the abundance of man-made or altered materials, Underwood (2001) even proposed a fourth major class of rocks: anthropic rocks.

Climate

Urbanization alters the surface temperatures, the albedo, precipitation, evaporation and transpiration rates, and the atmospheric energy balance in general. It may also have noticeable effects over the local climate (e.g., Changnon, 1976; Bornstein and Lin, 2000).

Water quantity

One of the main consequences of urban growth is the increase in population and subsequent water demand, and urban populations continue to grow at fast rates, especially in less developed countries (Table 1.1). Urban water demand often requires interbasinal water transfers, which affect the natural water budget in the area.

Approximately half of the world's urban population relies on groundwater as the main source of water supply (Table 1.1). In the USA, groundwater accounts for approximately 40% of the public water supply (Solley et al., 1998). Currently, San Antonio, Texas, is the largest city in the USA supplied from groundwater. However, groundwater is still an underutilized resource in many urban settings because of inadequate management, economy of scales, scientific uncertainties, and public policy promoting the usage of surface waters (Sharp, 1997). In the cases where groundwater is not a reliable source regarding its quantity or quality, it could still be used to balance or back-up the other sources of supply. For instance, low-quality groundwater could be used to clean streets, provide for fire suppression, flush toilets, or irrigate parks and lawns. Moreover, desalinization of brackish groundwaters may be more economical than using seawater, so that the use of poor quality groundwater is likely to increase.

It is interesting to note the differences in population estimates by different sources as shown by the inconsistency of Table 1.1. This illustrates the problems involved in understanding and quantifying the urban environment. Some of the differences are likely caused by people not accounted for in official census statistics, by rapid rates of growth, and by divergences on the definition of "city boundary" and "metropolitan area boundary".

Water quality

Water quality is a prime issue in urban water supply. Shallow aquifers and surface waters in urban settings are subject to pollution by runoff from paved surfaces, leaky storage tanks, surface spills and illegal dumping of hazardous waste, leaky sewage lines, and lack of sanitation facilities. With the increase of urbanized area, contamination of shallow aquifers is a major threat. In many developing nations, the installation of sewer systems lags behind population growth and the provision of mains for water supply. Only small areas in the centers of cities may be sewered. In the unsewered areas, more than 90% of domestic wastewaters may be released in pit latrines, cesspools, or septic tanks, which present significant potential sources of contamination (Mather et al., 1996). The contamination can be either point source or non-point source (e.g., aerosols, including motor vehicle exhaust and smelter emissions). In fact, there can be such a multitude of point sources in urban groundwater systems that contamination is diffuse and wide-spread so that it may be impossible to identify the precise sources (e.g., Lumsden, 1994; Mather et al., 1996; Van Metre et al., 2000; Wycisk et al., 2003).

Surface water

Urbanization affects the stream regime by modifying both base flow and flood discharge, bank erosion, sedimentation, land-sliding, declines in water quality, and flooding (Leopold, 1968, 1973). Garcia-Fresca and Sharp (in press) present some examples regarding the loss of surface courses to urban development, such as the long disappeared rivers of London (Barton, 1962; Sherlock, 1922), and Washington DC (O'Connor et al., 1999). Such buried channels may influence groundwater flow and affect wetlands, construction, and groundwater remediation.

Groundwater

Changes in surface water systems are commonly visible and apparent even to casual observers. Effects on groundwater systems may be equally significant but not always obvious. Human effects on groundwater in urban areas include overexploitation, subsidence, seawater intrusion, groundwater contamination, changes in recharge and discharge, alteration of the permeability structure, and destruction of important environmental resources, including wetlands and urban streams (e.g., Chilton et al., 1997; Garcia-Fresca and Sharp, in press; Howard, 2002).

The urban karst

The urban underground is an intricate and rapidly changing network of tunnels, buried utilities, garages, and other buried structures that disturb the natural structure of the ground and alter its porosity and hydraulic conductivity.

Based on the studies of porosity of karstic aquifers by Worthington (2003), and the volume of underground tunnels and installations catalogued for the Quebec City by Boivin (1990), Garcia-Fresca and Sharp (in press) conclude that the urban underground has secondary porosities and perhaps permeability distributions comparable to those of a karstic system (Table 1.2). The main difference between the natural and the urban system is that whereas the former takes millions of years to develop, the later can be emplaced in a few decades. Boivin (1990) did not provide estimates for the porosity created by smaller utility lines, trenches, pipes, and conduits. However, these "smaller pores" can dominate flow and transport in urban areas.

Reference to the influence of these shallow underground anthropogenic features is scarce in the hydrogeological literature, often limited to a vague sentence or two. Foster et al. (1994) pointed out that engineering structures can act as the principal sinks or discharge routes for the aquifer system or as barriers to shallow groundwater flow. In a study in the Ruhr valley in Germany, Coldewey and Meßer (1997) mention that the sand on which pipes are laid may contribute to increasing runoff and, thus, reduce groundwater recharge. Walton (1997) noted that sand-filled sewer trenches could serve as significant drainage pathways for excessive irrigation flows. In Sweden, localized groundwater declines have been related to higher permeability of the trench filling materials (sands) compared to the surrounding quaternary deposits (Norin et al., 1999); concrete or clay barriers were recommended to minimize these effects. Krasny (2002) inferred that urbanization increases the heterogeneity (of permeability and transmissivity). Marinos and Kavvadas (1997) studied groundwater table rise when flow is obstructed by shallow tunnels. They conclude the magnitude of the steady-state water table rise is proportional to the tunnel height and to the original hydraulic gradient in the direction normal to the tunnel axis, but independent of the hydraulic parameters of the aquifer. The water table rise is of the order of 1-10% the height of the tunnel, if located just below the original level of the water table, and smaller for tunnels below the original water table.

A comprehensive study of this issue is presented in Krothe (2002) and Krothe et al. (2002). Field data documented orders of magnitude increases in permeability along utility trenches. Finite-difference numerical simulations demonstrate that high permeability utility trenches alter groundwater flow. This can cause the development of complex or multiple solute plumes arising from a single point source. The utility trenches influence the direction and velocity of groundwater flow to the point of making it hard to predict.

Thus, the urban underground is comparable to a shallow karstic system (Sharp et al., 2001; Krothe et al., 2002; Sharp et al., 2003). The hydraulic conductivity of karstic aquifers is controlled by fractures and conduits (Halihan et al., 1999). The trenches in which the utility networks lie are analogous to naturally fractured systems. Larger underground openings, excavations, and tunnels are analogous to natural conduits, caves, and channels. The city thus becomes a pseudo-karst with highly variable permeabilities some of which can be exceptionally high; the permeability may be highly anisotropic and heterogeneous; there is internal drainage (storm drains that are analogous to dolines, swallets, and sink holes); rain water can be stored in the pseudo-epikarst; and recharge can be from both diffuse (natural and irrigation return flows) and discrete sources (i.e., leaky pipes and utility tunnels). This "urban karstification" is in continuous evolution as new structures are built over the older ones, buried structures are abandoned, and as existing geological structures, lithofacies, and other features are leveled and buried by further construction. However, as discussed above, the development of the urban karst takes place at much faster rate than natural karst. The oldest urban karsts are as old as human civilization and only date back a few thousands of years.

SCOPE

The goal of this study is to outline the effects of urban development on groundwater recharge, as well as the relevance of urban-enhanced recharge and its potential as a water resource. This is accomplished by means of an exhaustive literature review and is illustrated with the case study of the City of Austin, Texas (USA).

The current chapter presents the introduction to the topic and a summary of the effects of urban development on the local hydrology. Chapter two compiles a literature review of groundwater recharge in urban areas. First, world-wide examples illustrate the widespread of the phenomenon or urban-enhanced recharge. Then, the different mechanisms of recharge are discussed, as well as their relative relevance in urban areas. Finally methods for quantifying urban recharge are discussed. Chapter three describes some promising tracers of the effects of urbanization on groundwater. Chapter four compiles the physical description of the study area, i.e., the geography, climate, geology, hydrogeology, and history of the City of Austin. Chapter five portrays the case study of the City of Austin, Texas. Direct recharge is estimated under preurban conditions and under urban conditions for the year 2000. A water balance of the city is carried out for the same year, and several hydrogeochimical species are employed as tracers of urban recharge. In the final chapter some conclusions are drawn and lines of future work proposed.

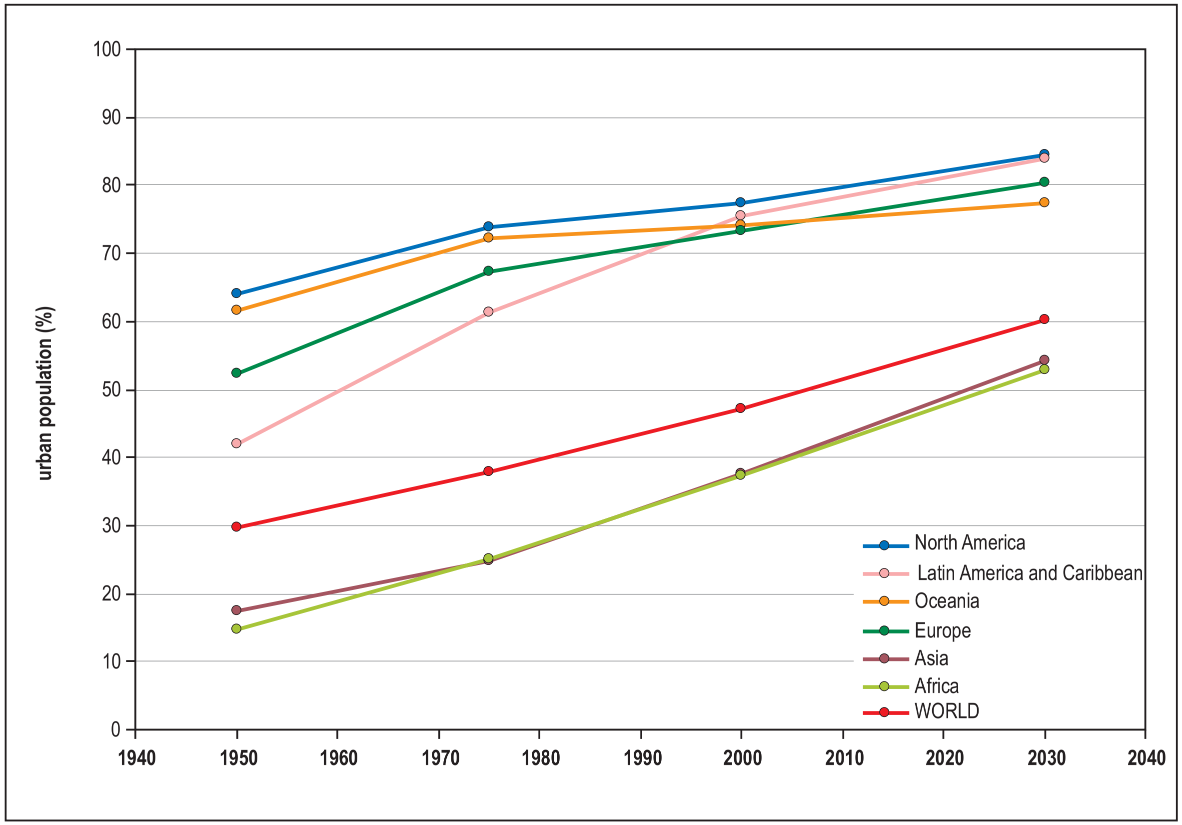


Figure 1.1: Percent of the world's population that is urban, for the world and by continent (after United Nations, 2002).

Table 1.1. The world's mega-cities. Groundwater dependent cities are noted in bold (after (1) United Nations, 2002, and (2) Morris et al., 2002). Discrepancies between sources (1) and (2) illustrate the problems involved in understanding and quantifying the urban environment, including estimating urban populations. Some of the differences are caused by varied definitions of "city boundary" and "metropolitan area," by people not accounted for in official censuses, and by rapid rates of growth.

1950 (1)		1975 (1)		2000 (2)		2001 (1)		2015* (1)	
city	Population (millions)	city	Population (millions)	city	Population (millions)	city	Population (millions)	city	Population (millions)
1 New York	12.3	1 Tokyo	19.8	1 **Mexico City**	25.8	1 Tokyo	26.5	1 Tokyo	27.2
		2 New York	15.9	2 São Paulo	24.0	2 São Paulo	18.3	2 **Dhaka**	22.8
		3 **Shanghai**	11.4	3 Tokyo	20.2	3 **Mexico City**	18.3	3 Mumbai	22.6
		4 **Mexico City**	10.7	4 **Calcutta**	16.5	4 New York	16.8	4 São Paulo	21.2
		5 São Paulo	10.3	5 Mumbai	16.0	5 Mumbai	16.5	5 Delhi	20.9
				6 New York	15.8	6 Los Angeles	13.3	6 **Mexico City**	20.4
				7 Seoul	13.8	7 **Calcutta**	13.3	7 New York	17.9
				8 **Teheran**	13.6	8 **Dhaka**	13.2	8 **Jakarta**	17.3
				9 Rio de Janeiro	13.3	9 Delhi	13.0	9 **Calcutta**	16.7
				10 **Shanghai**	13.3	10 **Shanghai**	12.8	10 Karachi	16.2
				11 **Buenos Aires**	13.2	11 **Buenos Aires**	12.1	11 Lagos	16.0
				12 Delhi	13.2	12 **Jakarta**	11.4	12 Los Angeles	14.5
				13 **Jakarta**	13.2	13 Osaka	11.0	13 **Shanghai**	13.6
				14 Karachi	12.0	14 **Beijing**	10.8	14 **Buenos Aires**	13.2
				15 **Dhaka**	11.2	15 Rio de Janeiro	10.8	15 **Manila**	12.6
				16 **Manila**	11.1	16 Karachi	10.4	16 **Beijing**	11.7
				17 **Cairo**	11.1	17 **Manila**	10.1	17 Rio de Janeiro	11.5
				18 Los Angeles	11.0			18 **Cairo**	11.5
				19 **Bangkok**	10.7			19 Istambul	11.4
				20 **London**	10.5			20 Osaka	11.0
				21 Osaka	10.5			21 Tianjin	10.3
				22 **Beijing**	10.4				
				23 Moscow	10.4				

* projected

Table 1.2: Porosity values for four karstic aquifers (after Worthington, 2003) and estimated porosity from human construction in Quebec City (after Boivin, 1990).

	POROSITY (%)		
	Matrix	Fractures	Conduits/channels
Smithville, Ontario	6.6	0.02	0.003
Mammoth Cave, Kentucky	2.4	0.03	0.06
Chalk, England	30	0.01	0.02
Nohoch Nah Chich, Mexico	17	0.1	0.5
Quebec City, Canada	n/a	unknown	0.06

Chapter 2: Groundwater recharge in urban areas

The hydrologic community has largely recognized that groundwater recharge can be inhibited in urban areas as impervious cover enhances runoff and limits infiltration (i.e., Leopold, 1968; Coldewey and Meßer, 1997). However urban development introduces new sources of recharge: leakage from water and wastewater distribution and collection systems, leaks from storm sewers, and irrigation return flow from lawns, parks, and golf courses (Lerner, 1986). Hutchinson and Woodside (2002) document a 350% increase in baseflow in the Santa Ana River in Orange County, California, that is attributed to increased discharge of wastewater. Christian (in preparation) use strontium isotopes to evaluate flow conditions in urban streams in Austin, and find a direct correlation between the isotopic composition of dissolved strontium and the degree of urbanization in the different watersheds comprised within the city. Their data indicate that at least for one stream, Waller Creek, which is located in the most urbanized section of the city, over 90% of the flow, under normal baseflow conditions, consists of treated water from the city's distribution systems. Discussed in more detail in a following section, the net recharge to urban areas commonly increases above natural recharge rates.

Numerous examples of significant water table-rise and increase on recharge to the groundwater have been reported in the last decade (e.g., Foster et al., 1994; Chilton et al., 1997; Chilton, 1999). Figure 2.1 portrays groundwater recharge for various cities as a function of the aridity of the cities' climate, expressed by mean annual rainfall. The table is adopted from Foster et al. (1994), who suggested ranges of natural recharge for non-urban environments, probable minimum recharge rates for comprehensively sewered and drained cities, and probable maximum recharge rates for unsewered and undrained cities which have been revised after adding nineteen data points to Foster et al.'s (1994) original four. In all cases, except for Birmingham, UK, the total recharge to the groundwater is increased by urban sources of recharge. For the exception of Birmingham, Lerner (1997) estimates a 4% loss in recharge, and is expressed as a pointing-down arrow in Figure 2.1. Urban-enhanced recharge is most significant in arid climates and in cities in developing countries. In a broader sense, urbanization introduces new sources and pathways of recharge (Lerner, 1986) and affects water quality.

RECHARGE MECHANISMS

Estimating recharge in natural areas is not an easy and straightforward task. Recharge to aquifers is a complex process that involves climate, vegetation (or lack thereof), soil properties, the vadose zone above an aquifer, and the hydrogeologic characteristics of the aquifer itself. Understanding all the mechanisms and processes involved is difficult. For instance, soil properties vary with the amount of moisture; secondary porosity may dominate the direction and velocity of flow; and calculating evapotranspiration is difficult. Hydrogeologists employ different methods to estimate recharge. Methods vary depending on the resolution of the study, the geologic environment, and legal, economic, social, and political constraints.

Estimating groundwater recharge is even more difficult in urban environments because the water balance is often altered by interbasinal transfers and the karstic nature of the shallow urban underground. According to Lerner (1990a), water in urban environments follows two networks of pathways that are often interconnected. The *natural network* is related to rainfall, evapotranspiration, runoff, infiltration, recharge, and groundwater flow. The *urban network* consists of leakage from the water distribution system, on-site water treatment devices, and sewers, as well as irrigation return flows. Because of water imports and exports, the hydrologic cycle may not be in balance locally.

Simmers (1998) describes three types of recharge, according to the processes involved and their spatial distribution. These are:

Direct recharge: vertical percolation of rainwater through the unsaturated zone. Direct recharge depends on evapotranspiration, the antecedent moisture content, and the vertical hydraulic conductivity of the unsaturated zone.

Indirect recharge: water losses from surface water bodies, such as rivers, lakes, reservoirs, and from water and sewage distribution systems.

Localized recharge: percolation through preferential pathways (desiccation cracks, burrows, lithologic contacts, faults, fractures, and karstic features).

To Simmers' classification, one more type must be added: *artificial recharge*, which is sourced from water intentionally applied by humans, such as return flow from irrigation of parks and lawns, and infiltration of runoff by means of different runoff detention and infiltration systems.

Quantifying groundwater recharge in natural systems is challenging, because of the uncertainties related to indirect and localized recharge, as well as the complexity of the processes that take place in the vadose zone. The urban environment is yet more complex because a large variety of land uses coexist within the city –a relatively small area–, and the heterogeneity of the shallow underground. In any case, the total recharge in a city will be the sum of the direct, indirect, localized, and artificial components. The uncertainties intrinsic to quantifying these sources make it desirable to simplify by means of a water balance based on the amount of groundwater abstractions, imports, water use, and wastewater outflows (Lerner, 1990a). The estimation of groundwater recharge should be an iterative process that is continuously reevaluated by data collection and monitoring (Simmers, 1998).

Below, the four mechanisms of recharge are discussed in detail, and quantification methods proposed for each. However simple the above categories of recharge may seem, when examined in detail, they may overlap and are not mutually exclusive.

DIRECT RECHARGE

Direct recharge in cities takes place by percolation in unpaved areas and, to a lesser extent, through paved surfaces that are not always perfectly "impervious". Lerner (2002) proposes a proportion of the impermeable area to be treated as permeable in recharge calculations.

Direct recharge is less important as the aridity of the local climate increases. It also decreases as the amount of "impervious" cover increases.

Direct recharge can be estimated by assessing the amount of pervious cover in the city. Precipitation and potential evapotranspiration data are transformed into effective precipitation (e.g., Lerner et al., 1993) by means of a daily soil moisture balance. This method uses root constants and wilting points to account for different crops and soil types.

A proportion of the impervious cover should be treated as permeable, as some infiltration does take place through asphalt, concrete, bricks, and other "impervious" materials. According to Lerner (2002) roughly 50% of the impervious cover should be accounted for as permeable.

INDIRECT RECHARGE

Indirect recharge is the sum of the recharge coming from seepage from surface water bodies, leakage from water mains, wastewater and storm sewers, and septic tanks. This study does not discuss groundwater-surface water interactions, but focuses instead on urban sources of recharge.

Recharge from losing streams in urban areas is changed as the stream flows are altered by urbanization. A decline in aquifer heads caused by overexploitation will alter the hydraulic gradients between surface and the aquifer and between adjacent formations. This could also enhance recharge. However, in the following, we do not address these recharge processes, which are site specific and chiefly dependent upon the local and regional hydrogeologic settings.

Although it is common practice to consider that all leakage becomes recharge, this is not correct. Some of this water will be lost to evapotranspiration; some may infiltrate into wastewater and storm sewers, and some may discharge to streams as interflow.

A simple way to assess the water available for recharge is to make a balance of the water served versus the wastewater treated (Lerner et al., 1993). Yang et al. (1999) quantified the recharge in the city of Nottingham, UK, by means of a calibrated groundwater flow simulation supplemented by calibrated solute balances for chloride, sulfate, and nitrogen. They concluded current recharge to the aquifer is less than prior to urbanization; however, mains leakage is the main current source of recharge in Nottingham. Barrett et al., (1999) review a broad variety of marker species for identifying urban recharge sources, and selected the most promising: trihalomethanes for mains water, isotopic composition of sulfur and oxygen in sulfate for both precipitation and mains water, and a number of potential markers for sewage. They conclude no ideal markers exist and recommend a multi-component approach.

Leakage from water mains

Water mains must be pressurized to avoid infiltration of contaminants into the mains as well as to insure distribution to the far reaches of the water system. Pressure is the cause of the high leakage rates in water distribution systems. A review of the literature shows that the most efficient cities report 10% water loss from the distribution system. Typical values in developed countries are around 20 to 30%, and 30 to 60% in the less developed countries (Table 2.1). In arid climates, the amount of water distributed in a city is often significantly greater than rainfall (Foster et al., 1994). Thus, mains leakage is a consistent source of indirect groundwater recharge.

Lerner et al. (1990) propose several indirect methods to estimate leakage from water distribution networks. One way is to assume a certain percentage of the water supplied is leakage. Thornton (2002) suggests that about 60% of the unaccounted for water can be attributed to leakage. Other methods are mass balances of inputs and outputs to the network. External losses on consumers' premises (on the "consumers' side of the meter") are not accounted by water supply authorities. They may be reflected as legitimate use per property, when in fact they can be the most leaky parts of the system. Leakage rates will vary spatially depending on the pressure of the water, the age and the material of the pipes, and the maintenance of the system.

Leakage from wastewater sewers

Reports of groundwater contamination by sewage or wastewater are numerous worldwide (e.g., Eiswirth and Hötzl, 1997, Hiscock et al., 1997, Ramaraju et al., 1999, Blarasin et al., 1999). These reports indicate that leakage from sewers is quite common. When sewer lines are located below the water table, they may infiltrate groundwater, and when located above the water table, they may leak (Lerner, 2002). Because flows in these pipes are not under pressure, it is reasonable to assume they leak less than water mains. A number of corporations exist that provide services for reaming sewer lines to clean out roots that have penetrated them. However, leakage from sewage lines is generally not an immediate economic loss so that repairs are delayed. Many cities lack sewer networks and rely on septic tanks or similar systems to dispose of waste water. In this cases, most of the supplied water turns into recharge to the subsurface (Foster et al., 1994).

Reports quantifying wastewater leakage from sewers are scarce in the literature. The few published estimations seem to agree on a leakage rate of 5% of the sewage flow through the network: Barcelona (Vázquez-Suňé, 2003), Nottingham (Yang et al., 1999), Munich (Lerner, 1997), Dresden (Grischek et al., 1996), and several other German cities (Foster et al., 1994). Rieckermann et al. (2003) indicate typical losses are below 5% of the sewage flow. Giudici et al. (2001) report 20% losses from the sewage network in Milan, Italy, with losses from the drinking water network around 10%. This is due to the fact that many Milan water users have private supplies through wells.

Some of the most recent methods to quantify the leakage from sewage systems consist of adding artificial tracers on the system and analyzing the composition downflow in order to make a mass-balance of the introduced solutes (Rieckermann et al., 2003).

Leakage from storm sewers

Recharge from storm water happens under transient high-flow conditions and it is very difficult to measure and model. Lerner (2002) proposes two methods to account for recharge from storm water: 1) an empirical approach, and 2) the assumption that some proportion of the surface of the city is not impermeable. Both approaches are, however, uncertain at best. Methods are yet needed to quantify the hydrogeologic effects of storm sewers.

Septic tank infiltration

On-site wastewater treatment systems can be assumed to recharge all the water they receive, except for some small losses to evapotranspiration and, perhaps, stream baseflow. Thus, about 90% of the water supplied in unsewered cities can recharge the groundwater (Foster et al., 1994).

LOCALIZED RECHARGE

Localized recharge takes place through faults, fractures, and cracks in the rock outcrops and therefore depends mainly on the geologic materials and structures as well as the soil types in each area. As defined above, localized recharge is not directly related to urbanization, although it can be affected by it.

Numerous approaches exist for modeling flow through fractures and conduits (e.g., Sharp, 1993; Zahm, 1998; Halihan et al., 1999). This problem is not exclusively related to urbanization and is not specifically addressed in this study.

ARTIFICIAL RECHARGE

Artificial recharge consists on the water intentionally applied to the underground and includes devices designed to enhance infiltration, as well as irrigation water in excess of plant needs.

Designed infiltration structures

A variety of man-made structures are constructed to reduce flooding, relieve the sewerage networks, and promote groundwater recharge. Such structures include recreational lakes and ponds, soakways, runoff detention ponds, retention basins, artificial infiltration ponds, spreading basins, recharge ditches, and injection wells.

It can be assumed that infiltration structures recharge all the water they receive, except for some losses to evapotranspiration and stream interflows, as is the case of septic tanks. The importance of such recharge sources depends on their abundance in a city, their location with respect to the aquifers and the particular design characteristics of each device. Maintenance plays an important role. When clogging takes place, infiltration structures may become ineffectual and minimize recharge.

Irrigation return flow

The water directly applied in parks and lawns, in excess of the plant requirements, will percolate and recharge the groundwater, except for some losses to evaporation and interflow. What makes this source of recharge different from effective precipitation is the intentionality of its application, as well as the uncertainties related to its quantification.

This source of recharge can be especially significant in arid and semi-arid climates. La Dell (1986) and Lerner (1990a) illustrate this with the example of Doha (Qatar), where the water table rise is directly related to the excessive irrigation of parks and lawns.

Recharge from excess irrigation can be quantified by mass balancing water supply, water use, the physical properties of the soils, and evapotranspiration (e.g., Berg et al., 1996). In arid and semi-arid areas variations in these parameters should be obvious when comparing dryer and wetter months.

HYDROCHEMICAL IMPLICATIONS OF URBAN RECHARGE

Table 2.2 summarizes the different sources of groundwater contamination in urban areas. Different sources of recharge have different effects on the quality of groundwater, some of which are undesirable and may present health hazards. Examples of groundwater contamination by leaky wastewater sewers and septic tanks are abundant in the literature. Salameh et al. (2002) report the occurrence of chlorination by-products (trihalomethanes) in groundwater in Amman, Jordan, derived from the leaky distribution network and seepage from cesspools. Ellis (1997) describes several pollutants present in urban stormwater and the potential of pollution to the groundwater from infiltration of runoff. Van Metre et al. (2000) studied sediment cores in urban lakes and reservoirs –ultimate runoff collectors– and suggested a direct relationship between the increasing loads of polycyclic aromatic hydrocarbons (PAH) in the sediments and vehicle traffic. More recently PAHs have been detected in alarmingly high concentrations in coal tar-based parking-lot sealant products which are easily mobilized by runoff (Mahler et al., 2004). Pesticides and polychlorinated biphenols are also common. These legacy pollutants pose a future threat to water quality in urban areas.

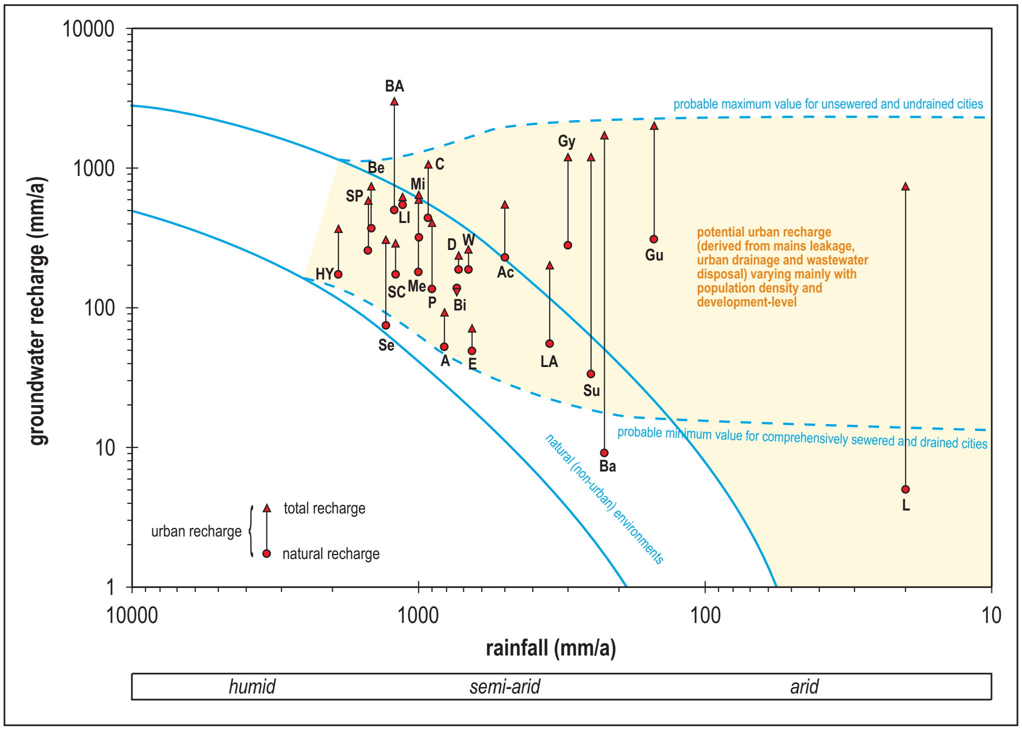


Figure 2.1: Urban-enhanced groundwater recharge in twenty three cities around the world (modified from Foster et al., 1994).

HY: Hat Yai, Thailand (Foster et al., 1994); **SP**: São Paulo, Brazil (Menegasse et al., 1999); **Be**: Bermuda, UK (Lerner, 1990b); **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 and Alvarado Rivas, 1999); **P**: Perth, Australia (Appelyard et al., 1999); **A: Austin (Texas), USA**; **Bi**: Birmingham, UK (Knipe et al., 1993); **D**: Dresden, Germany (Grischeck et al., 1996); **W**: Wolverhampton, UK (Hooker et al., 1999); **E**: Évora, Portugal (Duque et al., 2002); **Ac**: Aguascalientes, México (Lara & Ortiz, 1999); **LA**: Los Angeles (California), USA (Geomatrix, unpublished): **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).

Table 2.1: Compilation of water main or distribution systems losses in various cities of the world. Some general rates are denoted in italics.

City	Water main losses [%]	
Hull, UK	5	Chastain-Howley, pers. comm.
Los Angeles, USA	6 – 8	Geomatrix, 1997, unpub.
Hong Kong, China	8	Lerner, 1997
San Antonio, USA	8.5	Austin American Statesman, 1998
Évora, Portugal	8.5	Duque et al., 2002
Milan, Italy	10	Giudici et al, 2001
Austin, USA	12	City of Austin, 2003, pers. comm.
N Auckland, NZ	12.3	Farley and Trow, 2003
Toronto, Canada	14	City of Toronto, 2001, pers. comm.
Calgary, Canada	15	Grasby et al., 1997
US average	*16*	Thornton, 2002
Dresden, Germany	18	Grischeck et al., 1996
São Paulo, Brazil	16	Menegasse et al., 1999
UK general rates	*20 - 25*	Lerner, 1997
Göteborg, Sweden	26	Norin et al., 1999
Round Rock, USA	26	Austin American Statesman, 1998
Tomsk, Russia	15 - 30	Pokrovsky et al., 1999
Amman, Jordan	30	Salameh et al., 2002
Kharkiv, Ukraine	30	Jakovljev et al., 2002
Sana'a, Yemen	30	Alderwhish and Dottridge, 1998
Brushy Creek, USA	33	Austin American Statesman, 1998
Calcutta, India	36	Basu and Main, 2001
San Marcos, USA	37	Austin American Statesman, 1998
St. Petersburg, Russia	~ 30	Vodocanal 2000, unpub.
Developing countries	*30 - 60*	Foster et al.,1998
Lusaka, Zambia	45	Nkhuwa, 1999
Mérida, México	~ 50	Foster et al., 1994
Lima, Perú	45 - 60	Lerner, 1986
Cairo, Egypt	> 60	Amer and Sherif, 1997
Some Italian systems	*> 80*	Farley and Trow, 2003

Table 2.2. Sources of groundwater contamination in urban areas (modified from Howard, 2002)

POINT SOURCES	NON-POINT SOURCES
Municipal waste sites and landfills	Effluent from latrines and cesspits
Industrial discharges, leaks and spills	Oil and chemical pipelines
Leaks from underground storage tanks containing non aqueous phase liquids (NAPL) and brines	Lawn, garden and parkland fertilizers, herbicides and pesticides
Snow dumps	Road deicing chemicals
Spills from road and rail transport of chemicals	Oil, grease and aerosol emissions from motor vehicles
Stockpiles of raw materials and industrial wastes	Wet and dry deposition from smoke stacks
Design infiltration devices	Fill material containing construction waste

Chapter 3: Groundwater tracers in urban areas

Some chemical species can be good indicators of the different sources of recharge existing within a city. However, many of these tracers are not conservative, unique, or universal, and the signature of different sources may overlap. After the recharge sources have been identified, local circumstances, natural and anthropic, should define the appropriate tracers for each study area. However the use of some species as tracers is widespread in the study of urban groundwater. Several species can be used to identify sewage inputs to the groundwater, such as species present in animal waste (i.e., Cl^-, total nitrogen, metabolites, and pharmaceuticals) or detergents (i.e., sulphate, boron, and phosphate). Water supply inputs can be potentially identified with isotopes of hydrogen and oxygen in water, as well as disinfection by-products (trihalomethanes). Some species identify atmospheric precipitation, the presence of fertilizer, the density of traffic, or local manufactures. Besides the ions, different isotopes of some elements can also be studied. Throughout descriptions and applications of a variety of urban tracers are given by Vázquez-Suňé (2003), Barrett et al. (1999), Clark & Fritz (1997), among others.

Three basic tracers were tested in this study to evaluate the potential urban effects and sources of recharge to the groundwater:

$^{87}Sr/^{86}Sr$ of dissolved strontium, $\delta^{15}N$ of dissolved nitrate, and disinfection byproducts (trihalomethanes).

ENVIRONMENTAL ISOTOPES

Different uses of environmental isotopes for the study of groundwater have been reported in the literature. Jones and Banner (2000) used the $\delta^{18}O$ of groundwaters to determine the seasonal and spatial variations in recharge to the Pleistocene limestone aquifer in Barbados, West Indies.

Oetting (1995) analyzed major and trace elements and $^{87}Sr/^{86}Sr$ values from surface and fresh groundwaters for the Edwards aquifer. His results indicate processes of fluid-rock interaction and fluid-mixing taking place between groundwaters and carbonate and evaporite rocks in the aquifer. This study also characterized the evolution of saline groundwaters (also called "badwater") as the result of fluid-rock interactions and fluid mixing in the aquifer, confirming earlier results by Clement and Sharp (1988) and Clement (1989). Jørgensen and Holm (1994) combined the analyses of the isotopes of oxygen, hydrogen, and strontium in order to identify details in the mechanisms of salinization of groundwater and in differentiating the sources of salinity. $^{87}Sr/^{86}Sr$ ratios have also been used in studies of soils (e.g., Banner et al., 1994), and as indicators of climatic fluctuations in cave water geochemistry (Banner et al., 1996). Musgrove and Banner (2004) describe the geochemical and isotopic variations in vadose groundwaters from multiple caves within the Edwards aquifer; this study addresses the sources of dissolved constituents in groundwater, water-rock interaction pathways, changes in vadose flow routes, and groundwater residence time. Figure 3.1 presents a compilation of $^{87}Sr/^{86}Sr$ values of different geologic materials and waters. Limestones present much lower values than clastic sedimentary rocks, which are rich in clay minerals. Rainwater and surface waters in central Texas generally present higher values than groundwaters.

Fertilizers and animal waste are the two main sources of nitrogen in groundwater. The distinct $\delta^{15}N$ signature of dissolved nitrogen between different sources allows to identify, and sometimes quantify, such inputs to the groundwater (Mariotti, 1984; Heaton, 1986). Figure 3.2 presents a compilation of typical values of $\delta^{15}N$ of the different sources of nitrogen. Examples of the use of $\delta^{15}N$ as a tracer of sewage in groundwater are common in the literature (e.g., Kreitler & Browning, 1983; Mariotti, 1984; Barrett et al. 1997; Hiscock et al., 1997; Whitehead et al., 1999).

TRIHALOMETHANES

The disinfection of drinking water by chlorination produces a range of organic substances including trihalomethanes. Trihalomethanes form by the reaction of excess chlorine reacting with residual organic matter, especially humic substances. Trihalomethanes constitute a potential health risk as their carcinogenicity has been confirmed by several studies (Reuber, 1979; Velema, 1987). If detected in groundwater, they indicate the presence of water from the public supply, most likely leaked from mains. However, trihalomethanes are very highly volatile compounds and are also present in smaller concentrations in wastewater.

Salameh et al. (2002) report the presence of trihalomethanes on all groundwater samples in Aman City, Jordan. Concentrations range from 0.2 to 31.88 μg/ℓ, and the probable sources are chlorinated surface water at a treatment plant, the leaky water-distribution network, and sewage seepage from cesspools.

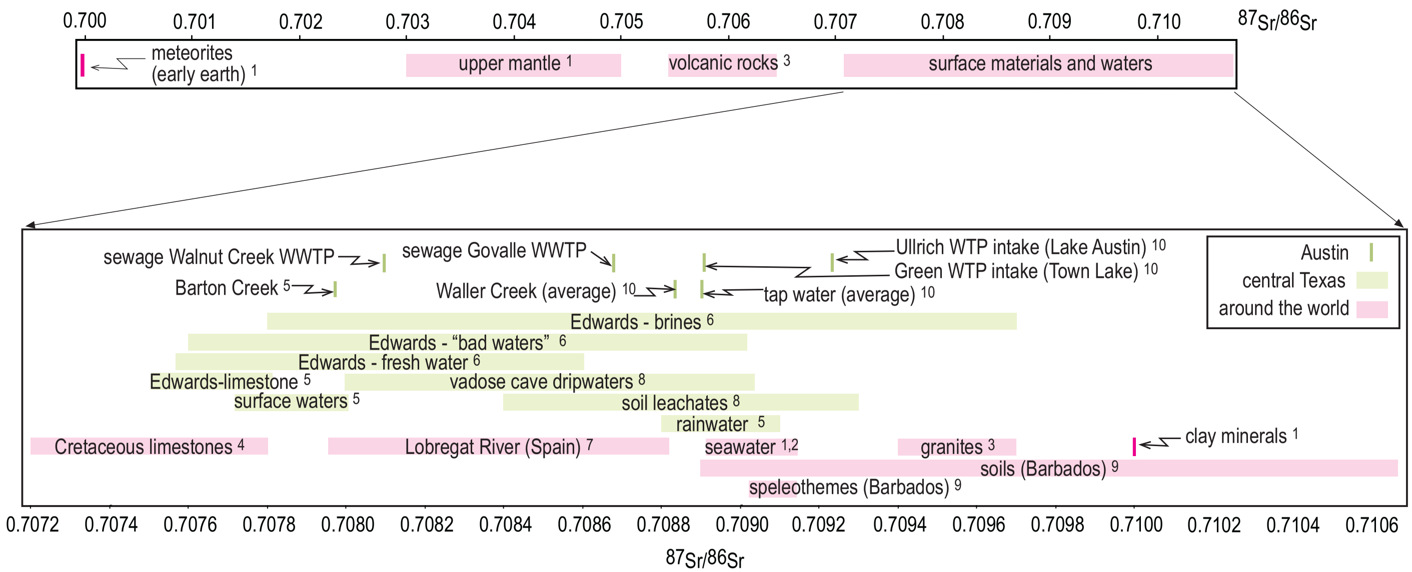


Figure 3.1: Compilation of $^{87}Sr/^{86}Sr$ values in different geologic materials and waters around the world, after 1) Banner, 2004; 2) Capo and de Paolo, 1990; 3) Neumann and Dreiss, 1995; 4) Burke et al., 1982; 5) Oetting 1995; 6) Oetting et al., 1996; 7) Soler et al., 2002; 8) Musgrove, 2000; 9) Banner et al., 1996; and 10) Christian, in preparation.

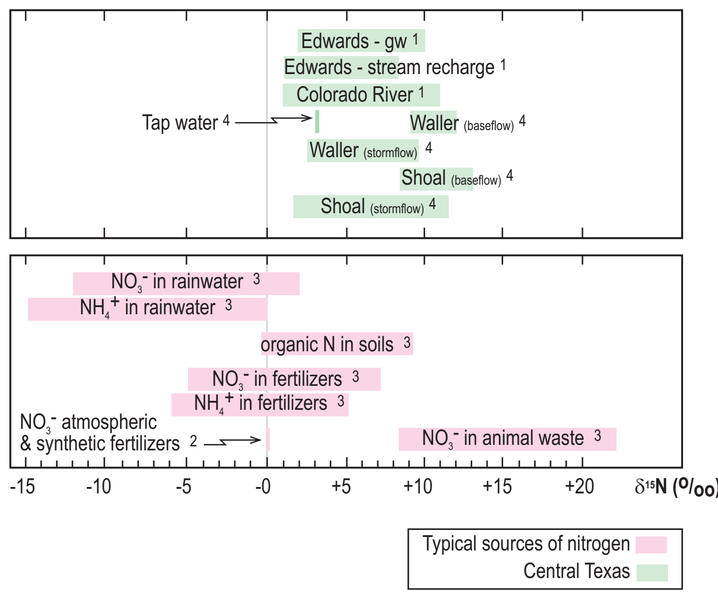


Figure 3.2: Typical values of $\delta^{15}N$ of different sources of nitrogen, after 1) Kreitler and Browning, 1983; 2) Mariotti, 1984; 3) Heaton, 1986; and 4) Ging et al., 1996.

Chapter 4: Study area

The concepts described in chapters 1 through 3 are applied to the City of Austin, Texas (USA). In this section the study area is described in terms of the geography, climate, geology, hydrogeology, and history of the urban development.

GEOGRAPHIC CONTEXT

Austin is located in central Texas, 260 km west of Houston, 315 km south of Dallas, 130 km north of San Antonio, and 930 km east of El Paso (Figure 4.1). Elevations within the metropolitan area vary from 120 m to nearly 300 m above sea level.

The city lies across the Balcones Escarpment, mostly on the physiographic region developed within the Balcones Fault Zone (Garner and Young, 1976) (Figure 4.2). Other parts of the urban area rest on the Edwards Plateau (or Hill Country) to the west and the Blackland Prairie, to the east. Physiographic differences on either side of the escarpment are substantial. The Hill Country is a limestone plateau carved into hills by numerous rivers and creeks. Soils are generally unsuitable for cultivation, and the most common vegetation are grasses, juniper and live oak. In the Blackland Prairie thick and fertile soils overlay clastic materials, and groundwater can be deep and occasionally of low quality.

The Austin urban area used to be situated in the center of Travis County, but today it sprawls into the neighboring counties of Hays, Bastrop and Williamson. Most of the areal expansion has taken place since the 1980s. The city is roughly dissected in north and south halves by the Colorado River and is entirely located within this basin. The main tributaries completely or partly contained within the city limits include Shoal, Waller, Boggy, Tannehill, Fort, Buttermilk, Little Walnut, Walnut, Decker, Onion, Carson, Blunn, East Bouldin, West Bouldin, Williamson, Slaughter, Barton, West Bull, and Bull Creeks and a number of smaller streams (Figure 4.3). Numerous springs are found in the city and the vicinity, including Barton Springs, one of the most important spring systems in Texas.

CLIMATE

The climate of Austin is humid subtropical with hot summers. Winters are mild, with below freezing temperatures occurring on an average of about 25 days each year. Rather strong northerly winds, accompanied by sharp drops in temperature, frequently occur during the winter months in connection with cold fronts. The cold spells are usually of short duration and seldom last more than a few days. Daytime temperatures in summer are hot, but summer nights are usually pleasant (National Oceanic and Atmospheric Administration, NOAA, online). The mean-annual temperature is 20°C, the mean-maximum temperature for July is 32°C, and the mean minimum temperature for

January is 3°C for the period of record, 1854-2003 (National Weather Service, NWS, online). Below freezing temperatures are generally constrained between the months of November through March, providing an average growing season of 273 days (NOAA, online).

Mean-annual precipitation is about 813 mm, although it ranges from 279 to 1650 mm during the period of record, 1856-2002 (NWS, online). Precipitation is fairly evenly distributed throughout the year, but the heaviest amounts occur in late spring. A secondary peak of rainfall occurs in September because of tropical cyclones from of the Gulf of Mexico. The Balcones Escarpment is the first significant topographic barrier for the moist and warm air masses coming from the Gulf of Mexico. These air masses cool down as they gain elevation and, thus, can produce large storms that can lead to flooding. Precipitation from April through September usually originates from thunderstorms, and fairly large amounts of rain can fall within short periods of time. Thunderstorms and heavy rains may occur in all months of the year, but most winter precipitation consists of light rain. Snow is insignificant as a source of moisture and usually melts rapidly (NOAA, online).

The mean-annual pan evaporation for 1916-79 was 1,880 mm (Farnsworth, 1982). Prevailing winds are southerly, however, in winter northerly winds are about as frequent as those from the south (NOAA, online).

GEOLOGY

The Balcones Escarpment is a topographic and geologic feature that extends throughout central Texas from Del Rio on the Mexican border nearly to Dallas. It is a crustal discontinuity that reflects several superimposed geologic events, namely 1) the Ouachita orogeny (Pennsylvanian); 2) the continental margin stage during the opening of the Gulf of Mexico (early and middle Mesozoic); and 3) Miocene age faulting, which created the Balcones Fault Zone (Figure 4.2). This fault system consists of coastward-dipping normal faults that act as the hinge between the stable continent and the subsiding Gulf Coast (the subsidence started in the Cretaceous and continues to the present) (Hentz, online; Young, 1972). In central Texas the general strike of the faults in the Balcones system is around N40E (Dunaway, 1962). Relative displacements of the footwall and hanging wall along the fault-plane are typically 3 to 45 m, with the maximum being about 180 m at Mount Bonnell fault.

The Balcones Escarpment separates the Edwards Plateau (or Hill Country) to the west from the Blackland Prairie to the east (Figure 4.2). On the continental scale, this feature is the physical frontier between the Great Plains Province and the Coastal Plain.

The different formations are described in detail by several researchers (e.g., Rose, 1972; Sharp, 1990; Scanlon et al., 2001), and in the Geologic atlas of Texas, Austin sheet (Bureau of Economic Geology, 1974).

Garner and Young (1976) described the following main rock types exposed in the Austin area: Cretaceous marine limestones, dolomites and clays, Tertiary sandy clays, and Quaternary terrace and alluvium deposits from the Colorado River (Figure 4.4). Figure 4.5 shows the footprint of the City of Austin for reference. The stratigraphy of the Lower Cretaceous in the study area can be organized into the Trinity, Edwards, and Washita Groups (older to younger) as depicted in Figure 4.6.

The extensive faulting creates a mosaic of different rock types, which results in varied hydrogeologic properties and regimes, different soils, flora, fauna, and perhaps even microclimate. Peter T. Flawn (cited by Woodruff, 1994) describes the Balcones Fault System and the Colorado River as "the fundamental geological constraint in the Austin area". The fault system extends NE-SW while the river flows NW-SE, dividing the study area into different geological environments. South of the Colorado River the Balcones Fault System juxtaposes the Kainer and Person Formations downward on the east ("soft" limestone), against the lower half of the Glen Rose Formation on the west ("hard" limestone). North of the river, the fault system transposes the Lower Cretaceous units (Glen Rose, Walnut and Edwards Limestone Formations) against the Upper Cretaceous (Del Rio Clay, Buda Limestone, Eagle Ford Shale, and Austin Chalk). Quaternary alluvial and terrace deposits are mainly found to the east of the fault.

HYDROGEOLOGY

A feature of great hydrogeological relevance in the Austin area is the Edwards aquifer (Figure 4.7). The Edwards aquifer is one of the most productive aquifers in North America (Sharp and Banner 1997). It is the sole source of drinking water for more than 1.5 million people along the Balcones Escarpment, supports farming and ranching to the west, provides habitats for several endangered and endemic species, and provides stream flow to several rivers in Texas. This karstic aquifer extends along the Balcones Escarpment, from Del Rio on the Mexican border (it extends beyond the Rio Grande into Mexico), to Bell County. It is divided into several hydraulically-differentiated segments: the San Antonio, the Barton Springs, and the Northern segments.

The City of Austin spreads over two segments of the Edwards aquifer outcrop: the Barton Springs segment and the Northern segment (Johns and Woodruff, 1994) (Figure 4.8). The former is located south of the Colorado River and east of the main faults, it has an unconfined and a confined portion, and supplies groundwater to several small towns and private well owners in Travis and Hays counties. Because the main discharge point –Barton Springs– enters Town Lake, from where 20% of Austin's drinking water is drawn, this segment of the aquifer also contributes to the City's water supply. The Northern segment of the aquifer lays north of the river, crops out west of the fault system, it is confined to the east of the faults, and has a minor relevance as a water supply. West of the fault the Glen Rose Limestone is found with its classic stair-step geomorphology. The Glen Rose crops out throughout the contributing zone, where runoff is directed to the recharge zone of the Barton Springs segment of the Edwards aquifer.

The geology, stratigraphy and hydrogeology of the Edwards aquifer are synthesized by Senger and Kreitler (1984), Sharp (1990), and Scanlon et al. (2001). The aquifer consists of all members of the Edwards Group and Georgetown Formation (lowest member of the Washita Group) (Figure 4.6). An erosional hiatus occurred between deposition of the rest of the aquifer and the Georgetown Formation. The aquifer is confined below by the Upper Glen Rose Formation, and above by the Del Rio Formation. Other members of the Trinity Group form the deeper and regionally broader Trinity Aquifer.

The effective porosity of the Edwards aquifer is mainly due to fracturing and karstification of the limestone that provides discrete groundwater flow through fractures, caves, and conduits. However, the intense faulting isolates different blocks of the aquifer, juxtaposing stratigraphic units of different hydraulic properties, and thus inhibiting flow between adjacent blocks. Nonetheless groundwater flow can occur through the very fractures that isolate the blocks (Figure 4.9).

Natural and urban sources of recharge to the aquifer are discussed in chapter 4, and discharge takes place through wells and springs. The most important discharge points in Austin are Cold Springs and the Barton Springs system, which discharge into Town Lake on the Colorado River. Other minor springs are described by Brune (1981). Major Texas springs have historically dried during periods of drought (Sharp and Banner, 1997), and many others have declined and disappeared, often related to shifting from farm and ranch land uses to urban land uses (Brune, 1981). Spring discharges are expected to decrease as water demand and pumping increases.

The Quaternary deposits provide small amounts of water through wells but do not constitute a resource comparable to the Edwards aquifer, and thus their hydraulics and hydrochemistry have not been as extensively studied. However, small springs drain these materials, and numerous wells are completed in these units. It is possible that they play a role in the recharge to the confined portions of the underlying Edwards aquifer. The location of Quaternary deposits is shown in Figure 4.10.

HISTORY OF URBAN DEVELOPMENT

Springs at the base of the Balcones Escarpment are the key to understanding the development of human settlements and land use patterns in Texas since prehistoric times. Human inhabitation of the Balcones Escarpment by Paleo-Indians can be tracked to at least the late Pleistocene, around ~9200 B.C. (Hester, 1986). Some examples of their presence near Austin are the Levi Rockshelter and the Wilson-Leonard site. Pre-Columbian Americans preferred spring water over river water and settled near springs or spring fed creeks. First European explorers entering Texas were often guided by Indians from one spring to another over well-worn trails (Brune, 1981). Like the Indians, early Texas settlers preferably located in the proximity of springs. The earliest permanent Spanish settlement in Texas was established downstream from San Pedro Springs in 1718, in present day San Antonio.

Austin was first settled by English-speaking people in the 1830s. The site was chosen because of the reliable water supply from the Colorado River and from several springs which drain the Edwards aquifer. The location offered the advantage of prime farmland to the east and grazing land to the west. The Austin Chalk provided a firm base to build upon as well as construction material. The navigability of the Colorado River was also considered an asset (Palmer, 1986). The first documented settlement of the area dates at 1835, when Jacob Harrell and his family camped near the present site of the Congress Avenue bridge (Bear, online). In 1837 Texas declared its independence from Mexico, and the settlement was named Waterloo. In 1839 it was renamed Austin and became the capital of the Republic of Texas. Texans voted to join the United States of America and were admitted as the 28th state of the Union in 1845. Austin was selected permanently as the seat of government in 1872.

By the last part of the 19th Century the Blackland Prairie soil fertility declined and required chemical fertilizers, and the hills to the west were under advanced stages of overgrazing (Palmer, 1986). These trends continued throughout the 20th century.

Prior to the 1970s, the major industries between San Antonio and Austin were limestone quarries (Palmer, 1986). With the boom of the Texas economy during the 1980s both industry and people relocated to the region, which created a land rush which has caused the increase of urban growth and sprawl ever since. Heated controversy resides on the direction in which this development must follow. Opponents to developing the hills to the west argue conservation of wild habitats and protection of the Edwards aquifer, while preservation of prime farmland is a strong argument against sprawling eastward.

A compilation of old maps and drawings of Austin, Texas, allows the assessment of the evolution of the areal growth of the urban area (Figure 4.11). The quality of some of the figures, as well as scaling problems, results in questionable accuracy on the calculated areas; however, it provides a good approximation to the process. For the purpose of this study, only the Full Jurisdiction area of the city was considered (Figure 4.12). The images used in this exercise are compiled in Appendix 1.

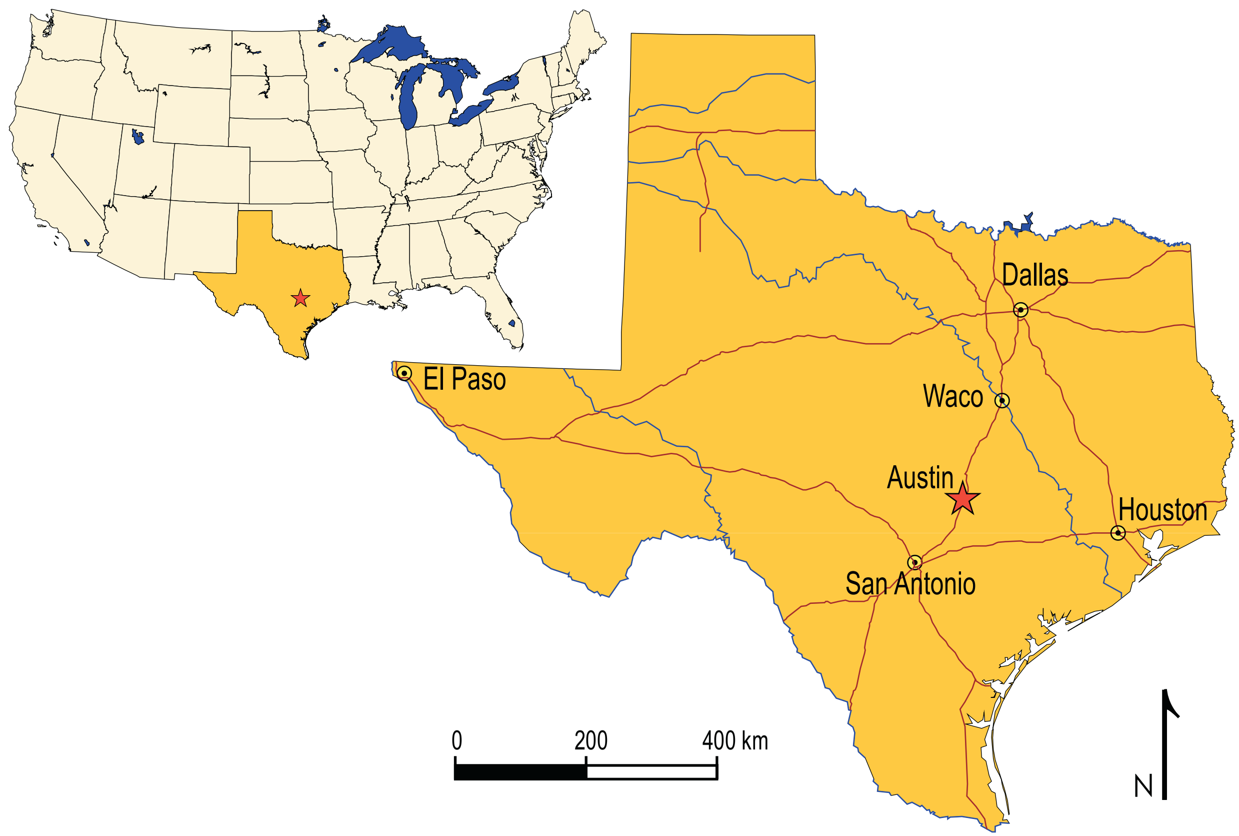


Figure 4.1: Geographic location of of Austin, Texas (USA)

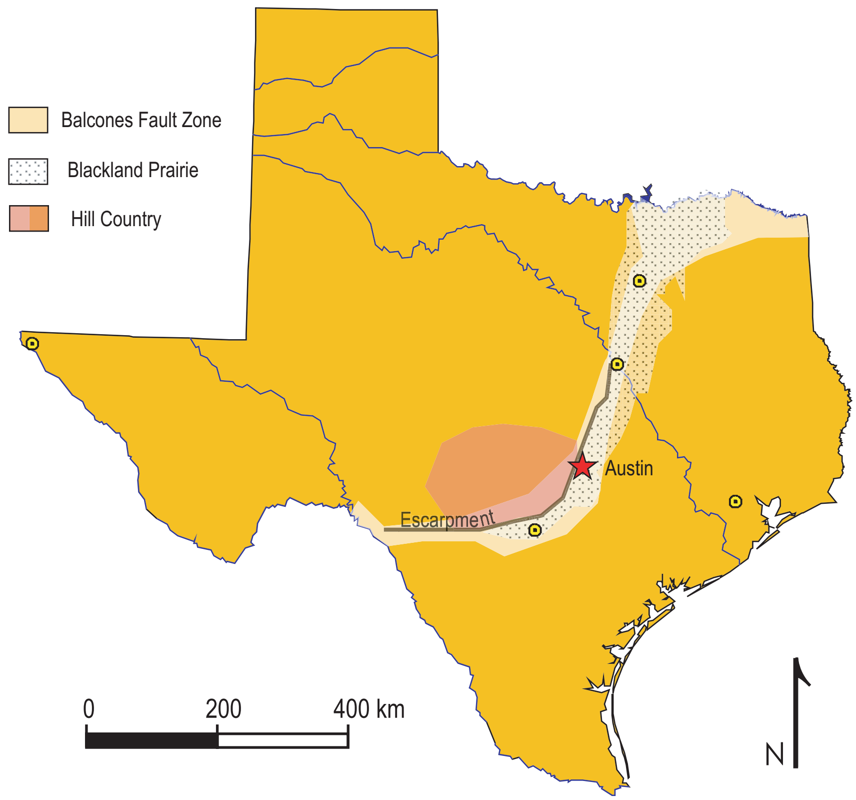


Figure 4.2: The Balcones Escarpment and main geomorphological features in central Texas.

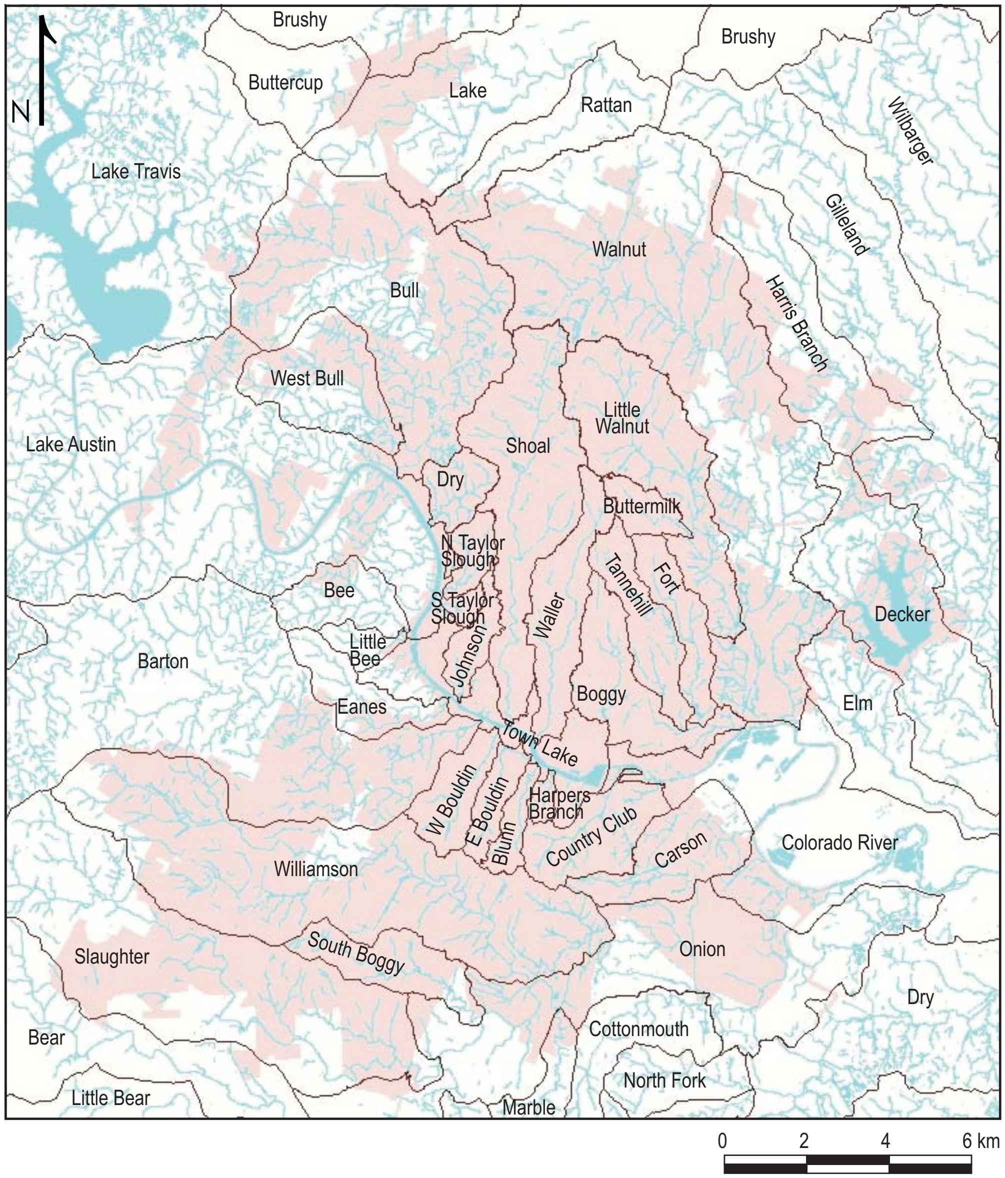


Figure 4.3: Colorado River tributaries and their watersheds in the vicinity of the City of Austin, Texas (City of Austin GIS data, online).

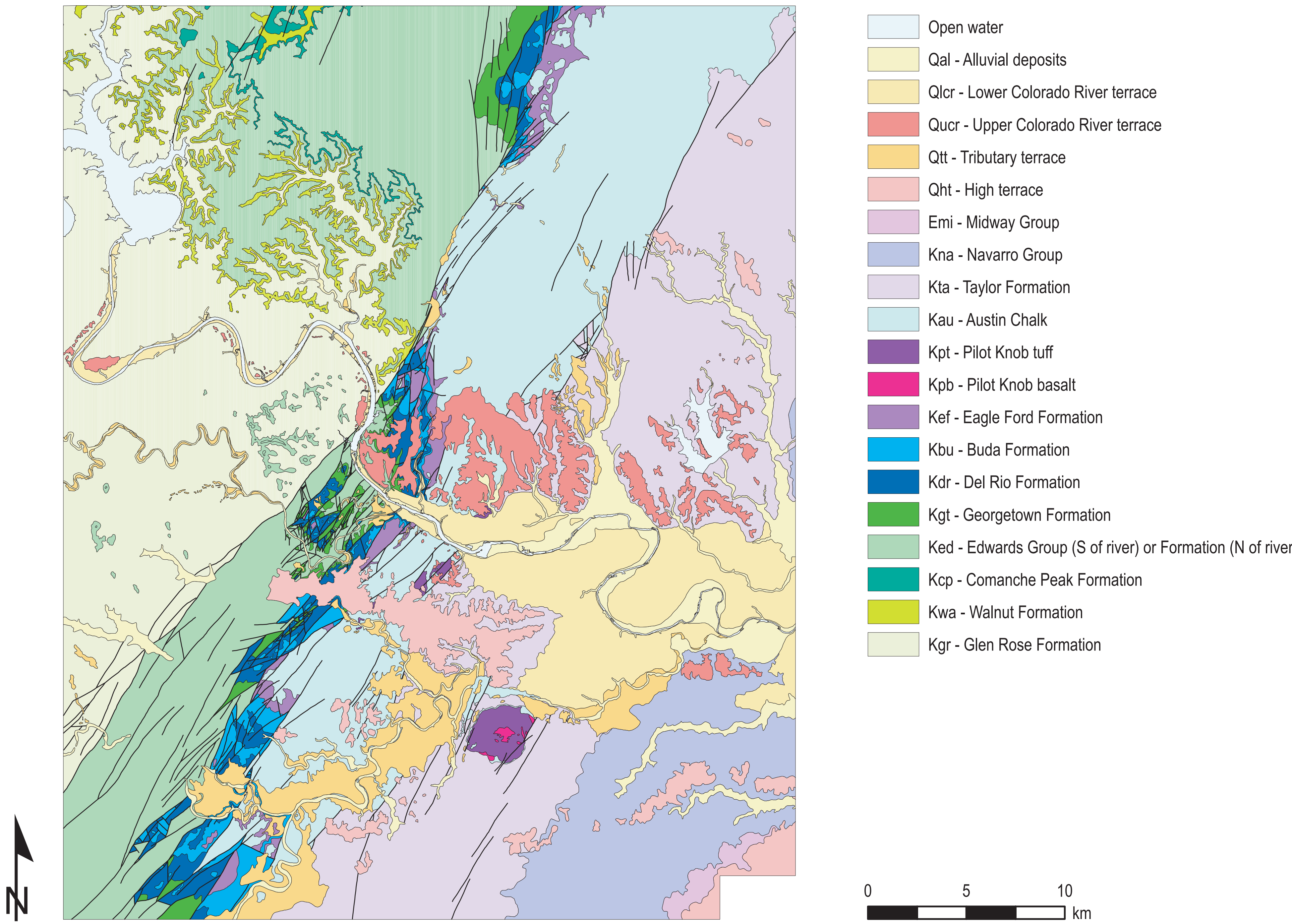


Figure 4.4: Geology of the Austin area (modifyed from Tremblay and Andrews, 1997; after Garner and Young, 1997).

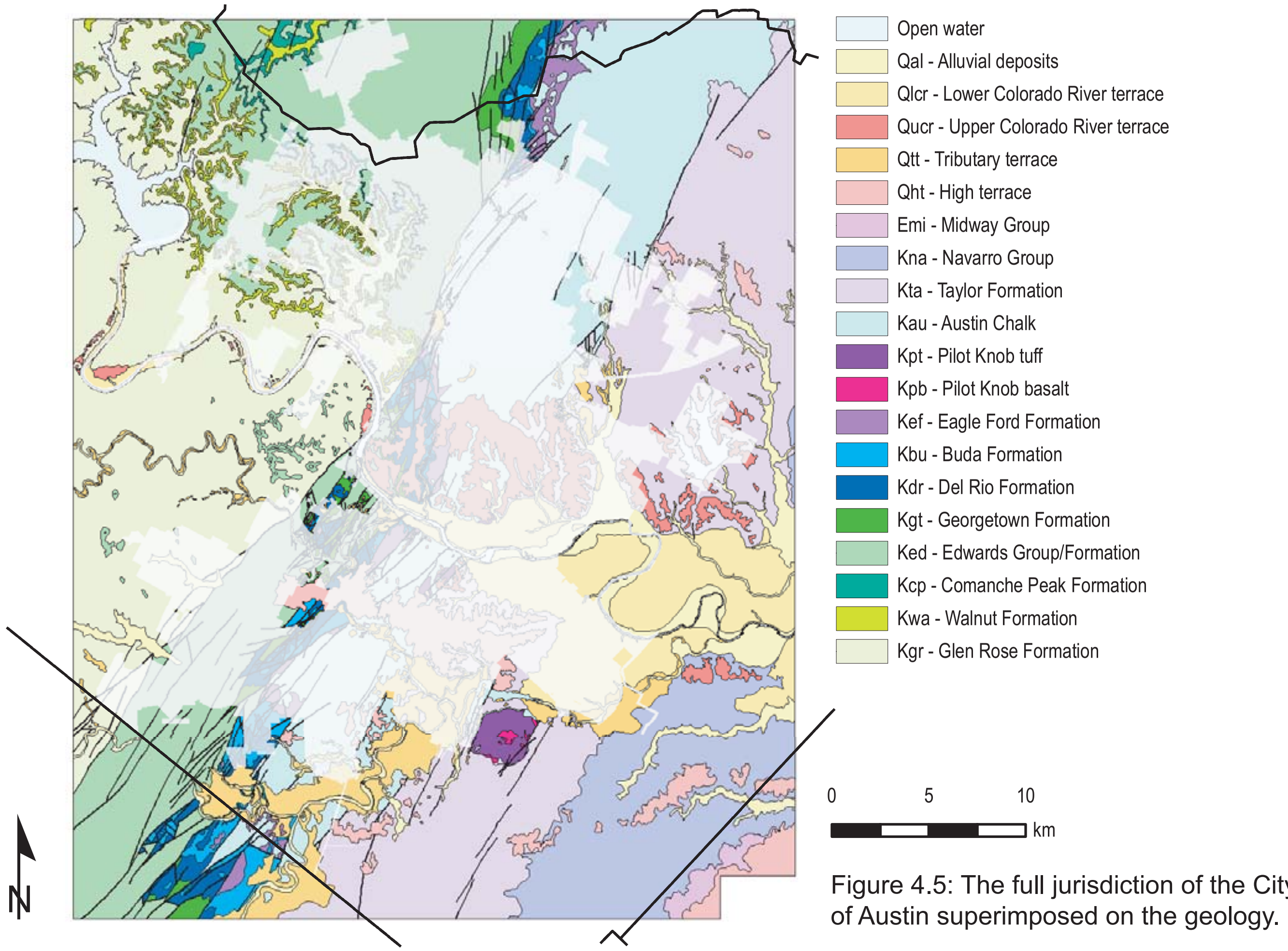


Figure 4.5: The full jurisdiction of the City of Austin superimposed on the geology.

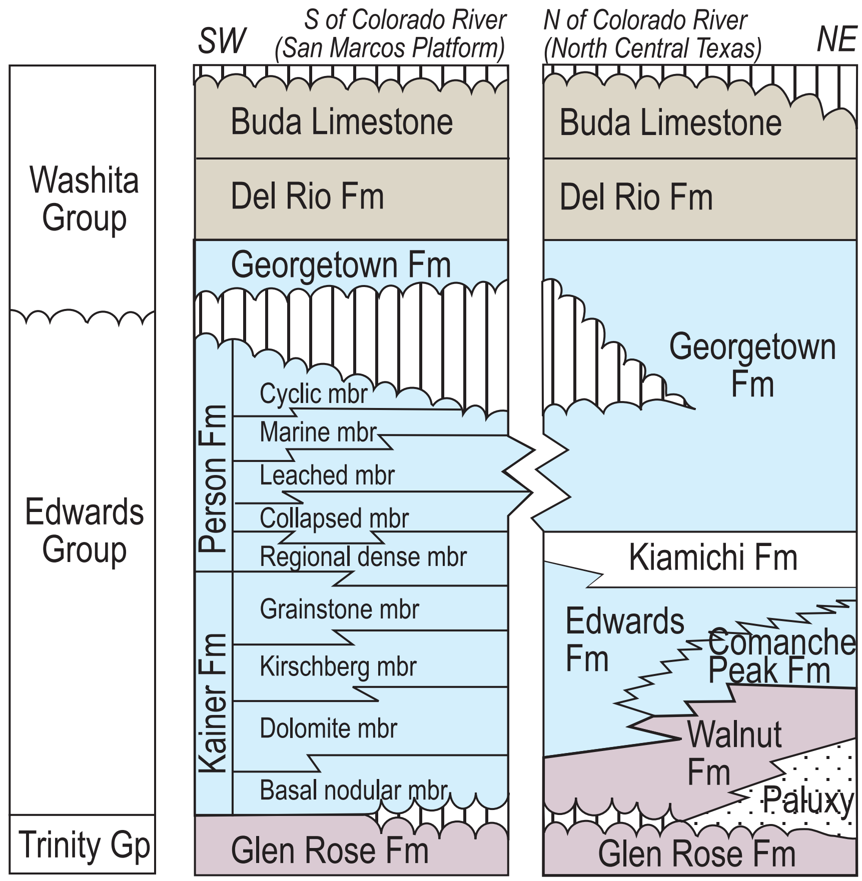


Figure 4.6: Hydrostratigraphy of the Lower Cretaceous materials in the Austin area (after Rose, 1972; Hauwert et al., 1998; Sharp, 1990; and Scanlon et al., 2001).

Figure 4.7: The Edwards and Edwards-Trinity aquifers of Texas
Source: TNRIS, online).

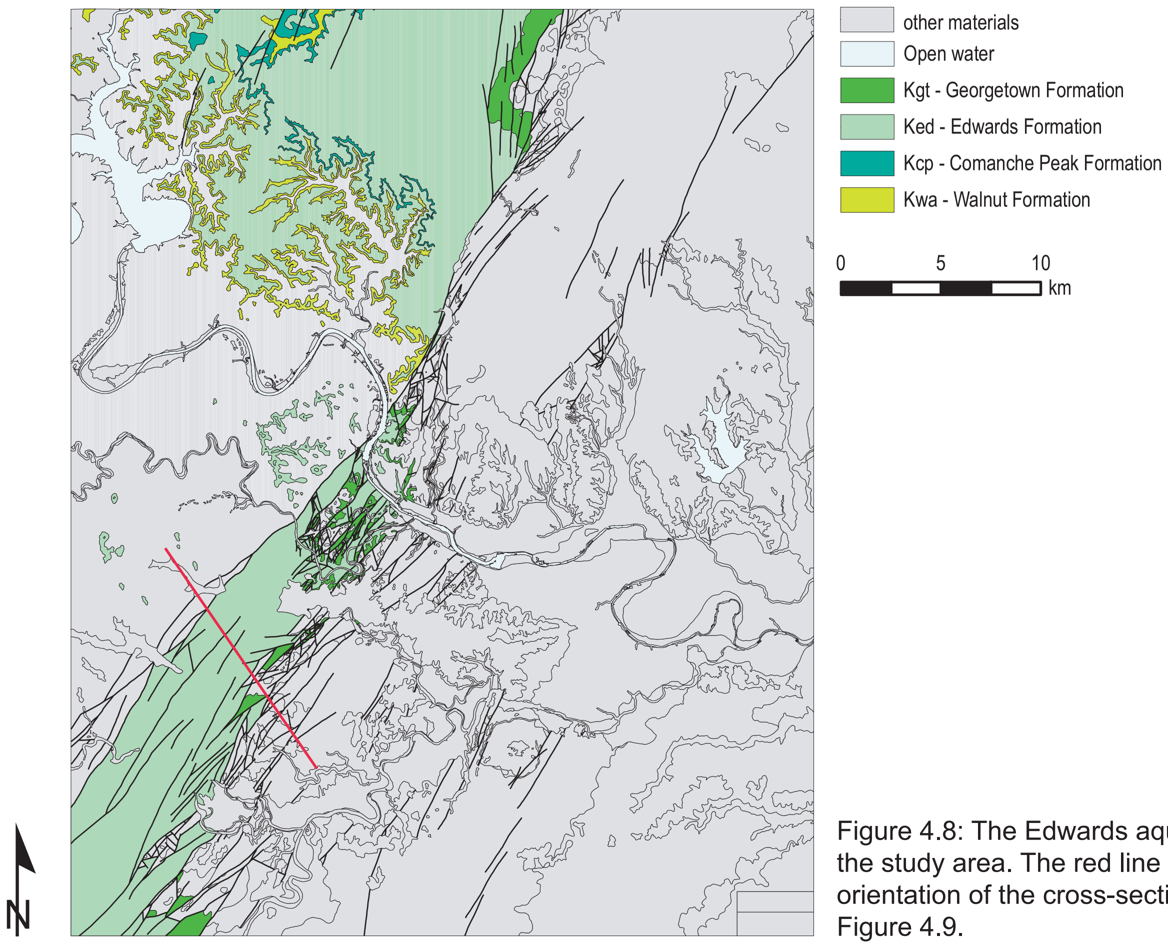


Figure 4.8: The Edwards aquifer within the study area. The red line indicates the orientation of the cross-section shown in Figure 4.9.

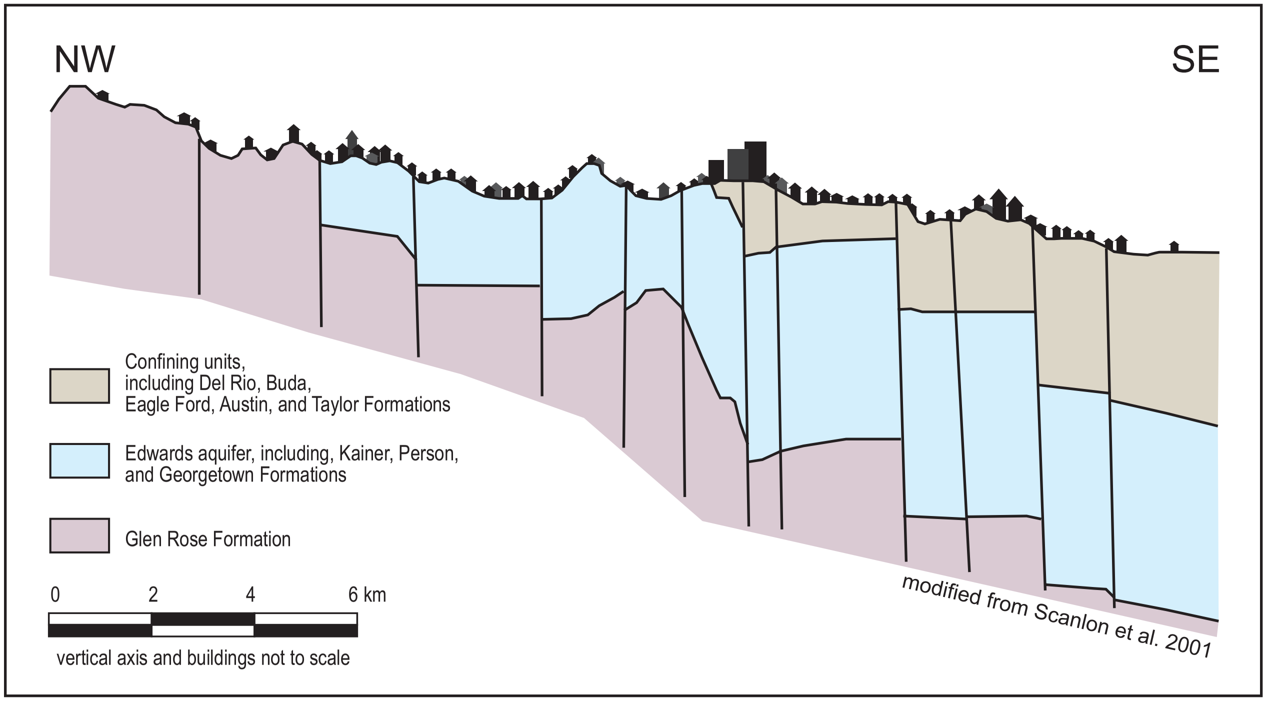


Figure 4.9: Simplified cross-section of the Edwards aquifer within the City of Austin.

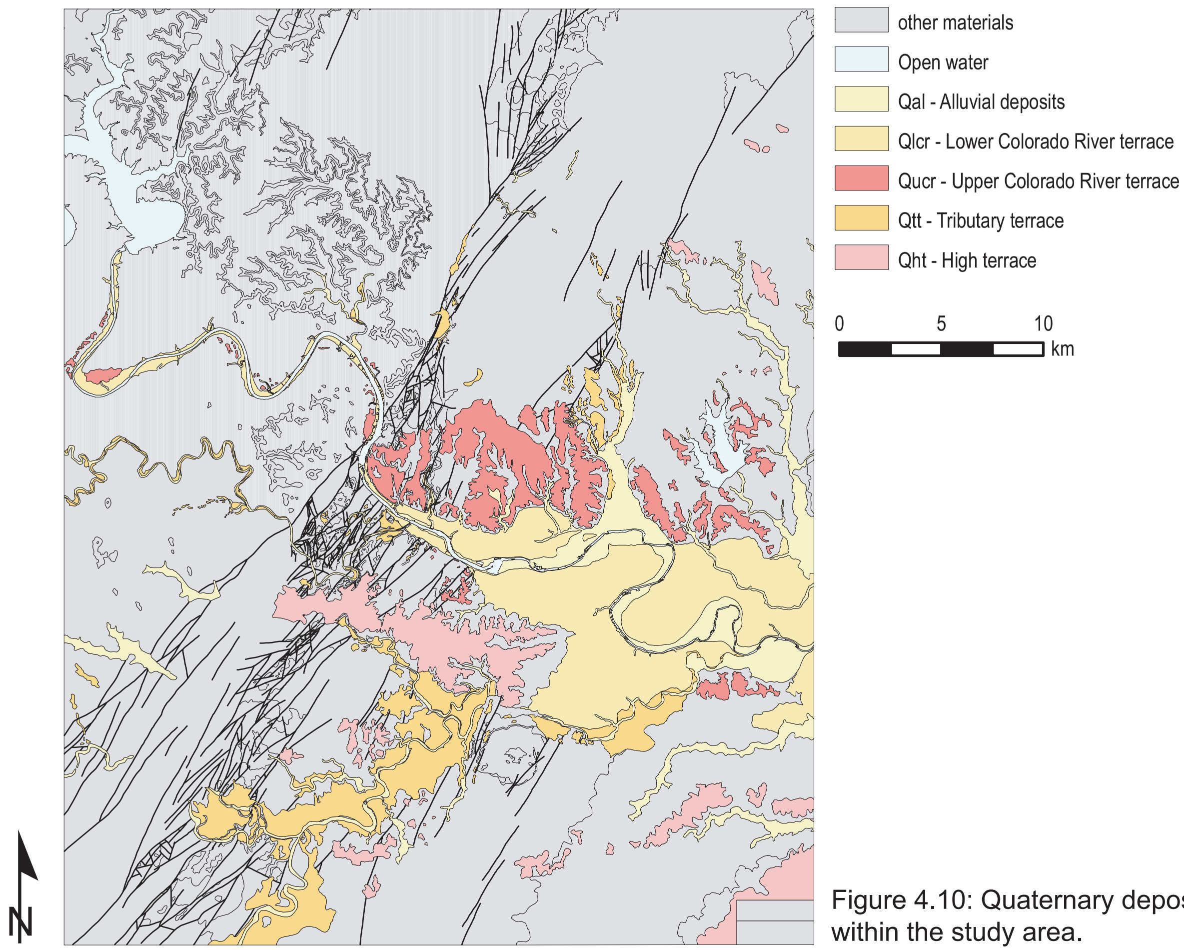


Figure 4.10: Quaternary deposits within the study area.

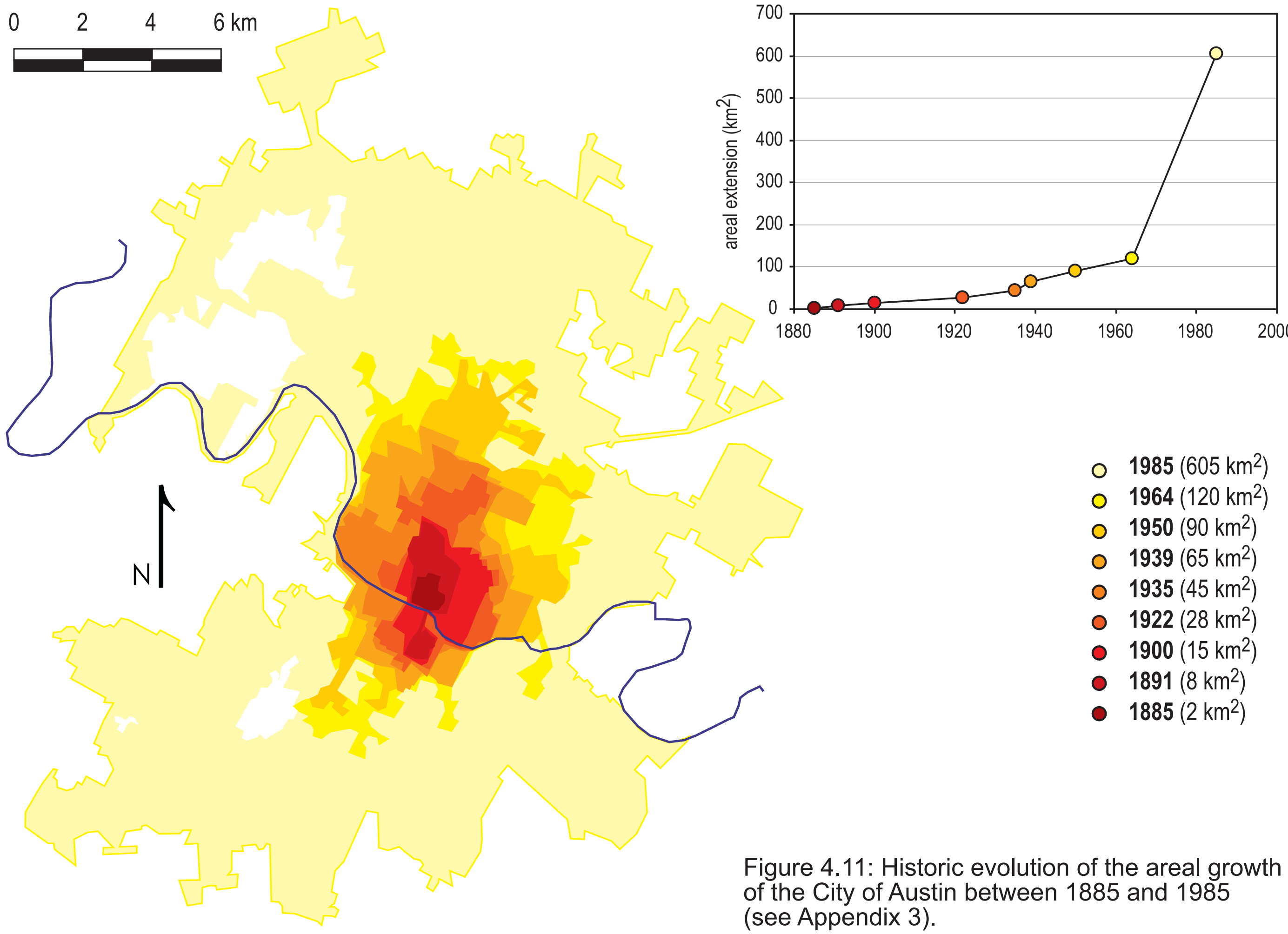


Figure 4.11: Historic evolution of the areal growth of the City of Austin between 1885 and 1985 (see Appendix 3).

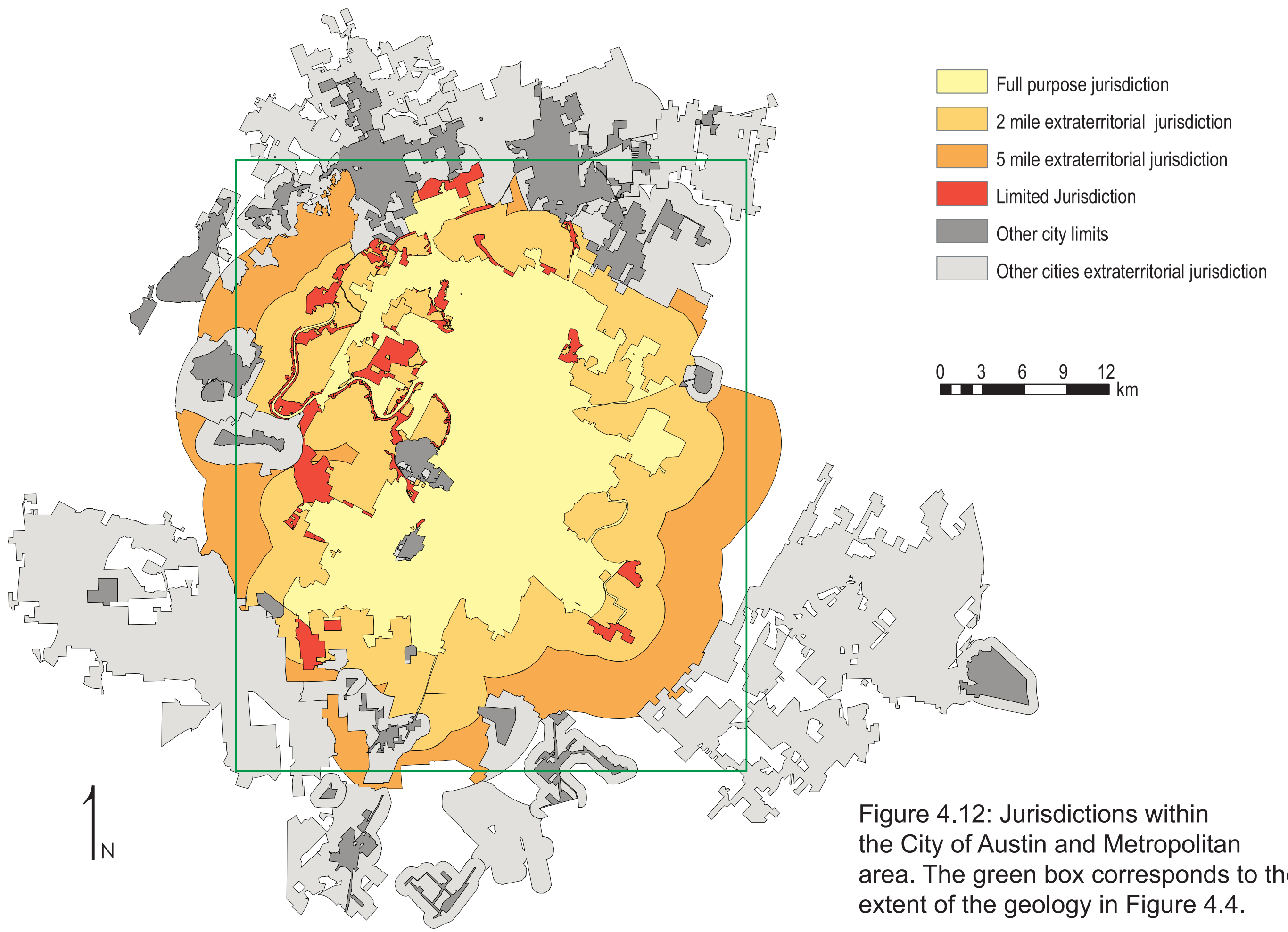


Figure 4.12: Jurisdictions within the City of Austin and Metropolitan area. The green box corresponds to the extent of the geology in Figure 4.4.

Chapter 5: Recharge in Austin

This chapter presents a study of recharge to groundwater in the City of Austin area. Prior to urbanization, the main source of recharge was infiltration of precipitation. With the introduction of impervious surfaces, recharge from precipitation is expected to decrease, but new sources of recharge of urban origin are introduced.

First, a pre-urban assessment of direct recharge was carried out, based on hydrogeologic analyses. Direct recharge was then reassessed to account for land use and impervious cover in the year 2000. A water balance was calculated for the city for the same year to estimate the amounts of urban water available for recharge and the indirect and artificial components of recharge (from leakage and irrigation, respectively). Water sampling campaigns were conducted between the summers of 2002 and 2003 and have been divided into two groups: 1) sampling pertinent to assessing urban influences on natural recharge and 2) sampling aimed at recharge from strictly urban sources.

DIRECT RECHARGE

In order to understand the relevance of urbanization on groundwater recharge, an assessment of direct recharge from rainfall prior to and after urban development is necessary.

Preurban direct recharge

Under preurban conditions, recharge takes place mainly by direct infiltration of precipitation. Thus, preurban direct recharge can be estimated by assuming zero impervious cover and by studying the spatial distribution and properties of the outcropping rocks. The tasks are significantly simplified by using the geographic information systems (GIS) ArcView and ArcGIS.

For this exercise, the area of the city was redefined in order to match the coverage of the water and wastewater networks (Figure 5.1). Then the geology of the area was overlain and trimmed to match the water and wastewater service area (Figure 5.2). Each geologic material was isolated or grouped with adjacent materials of similar hydrogeologic characteristics (Figures 5.3-5.12). Thus, Quaternary materials, Edwards aquifer limestones, and the Taylor and Navarro Formations were grouped. The Glen Rose, Del Rio, Buda, and Eagle Ford Formations; the Austin Group; and the Pilot Knob Tuff were analyzed separately. The surface area covered by each hydrostratigraphic unit was measured using a GIS (Table 5.1). A recharge coefficient, as the percentage of precipitation that turns into recharge, was assigned to each rock formation, except for the Barton Springs segment of the Edwards aquifer, where it was back-calculated from spring discharge. These coefficients may be adjusted as new information is acquired.

Mean annual precipitation data for Austin was obtained from the National Oceanic and Atmospheric Authority's public records (online).

Recharge to each geologic division was calculated from precipitation and the recharge coefficient. For the Quaternary materials, a recharge coefficient of 9% was determined, based on a hydrogeologic study of similar terrace deposits in Ellis County, Texas, by Wickham (1991). The coefficient assigned to the Austin Group was 1%, based on Mace (1998). For the Northern segment of the Edwards aquifer a recharge coefficient of 20% was applied as determined by Jones (2003). The other coefficients were subjectively assigned, relatively to the known values and the hydraulic properties of the materials. The Del Rio and Eagle Ford Formations were considered impermeable (recharge coefficient, 0%); the Glen Rose, Buda, Taylor and Navarro Formations were assigned the same coefficient as the Austin Group (1%); the Pilot Knob Tuff was assigned a coefficient of 5%. The accuracy of the former is highly uncertain, however the limited outcrops of this unit makes this fact nearly irrelevant.

For the Barton Springs segment, a different calculation was used, as recharge can be directly related to the discharge from the aquifer, and such discharge is a relatively well known parameter (e.g., Sharp, 1990; Scanlon et al., 2001). The calculation is as follows:

Recharge = Discharge + ΔStorage

$R = D$ (assuming $\Delta S \approx 0$)

$R = D_{springs} + D_{wells}$

$D_{springs} = D_{Barton} + D_{Cold} = 1{,}800$ l/s

$D_{wells} = 0.1 \times D_{Barton}$ (Scanlon et al., 2001)

$D_{Barton} = 1{,}500$ l/s (Slade et al. 1985)

$R = 1{,}800 + 0.1 \times 1{,}500 = 1{,}950$ l/s $= 61{,}495{,}200{,}000$ l/a

$A_{recharge} = 233\ km^2$

$A_{contributing} = 684\ km^2$

$A_{r+c} = 233 + 684 = 917\ km^2$

$R = 61{,}495{,}200{,}000$ l/a / $(917\ km^2 \times 10^6) = 67.1$ mm/a

About 85% of the natural recharge to the Barton Springs segment of the Edwards aquifer takes place from losing streams that cross the outcrop, and the rest from direct rainfall over the outcrops (Sharp, 1990). Thus, in order to compute the direct recharge of rainfall over the outcrop, the calculation is as follows:

$R = D_{springs} + D_{wells}$

$R = R_{rain} + \cancel{R_{streams}} + D_{wells}$

$R_{rain} = 0.15 \times D_{springs}$

$R_{streams} = 0.8 \times D_{springs}$

$R = (0.15 \times 1{,}800) + (0.1 \times 1{,}500) = 420$ l/s $= 13{,}245{,}120{,}000$ l/a

$R = 13{,}245{,}120{,}000$ l/a / $(233\ km^2 \times 10^6) = 56.8$ mm/a

Whether only direct recharge or both direct and indirect (from losing streams) recharge are considered, the results are similar: 56.8 and 67.1

mm/a respectively (Table 5.1). As a comparison, Table 5.2 presents the estimated annual recharge for Barton Springs segment of the Edwards aquifer, based on the numerical model of Slade et al. (1985). Recharge values are highly variable from year to year and do not seem to be closely related to the amount of rainfall but, perhaps, to the spatial and temporal distribution of the rain throughout the year and, particularly to the intensity of rainfall events.

Recharge was converted from rate (mm/a) to volumetric flux (ℓ/a) using the respective surface areas covered by each rock type. The sum of all volumes was computed to obtain the overall volume of recharge for the entire city. For the Barton Springs segment of the Edwards aquifer, the volume of infiltrated precipitation alone was used, and recharge from losing streams ignored. Finally, the service area defined above was used to transform the total recharge from volume back to a rate to normalize or average recharge over the area of the whole city.

The direct recharge to groundwater prior to urbanization in the Austin area is estimated to be 53 mm/a, most of which takes place over the Quaternary deposits and the Edwards aquifer outcrop.

Direct recharge under urban conditons

In the previous section, direct recharge before urban development was estimated. However, direct recharge from infiltration of rainfall is reduced by addition of "impervious" cover: roads, sidewalks, buildings, parking lots, and other urban surfaces. For this exercise a conservative point of view was adopted by ignoring the fact that not all urban surfaces and pavements are completely impervious.

A coverage of the landuse in the city was obtained from the City of Austin public GIS datasets (City of Austin, online). The landuse dataset was trimmed to the shape of the previously defined water and wastewater service area (Figure 5.13), as well as the shapes of the different rock outcrops (Figures 5.14 - 5.23). The areas designated with the same landuse within each rock-type were summed, and percentages of impervious cover were assigned to each landuse (Table 5.3). The amount of pervious cover left within each rock-type was calculated and used to recalculate direct recharge in a similar manner as in the previous section (Table 5.4).

The average annual direct recharge to the groundwater after urbanization in the Austin area is estimated to be 31.3 mm/a, which is a decrease of over 20 mm from preurban conditions. The remaining precipitation is surface runoff and evapotranspiration. This estimate is rather conservative if we consider the fact that not all manmade urban surfaces are impervious.

Urban effects on direct recharge

The previous sections demonstrate that urban development affects the rate and location of recharge to groundwater. Groundwater samples were collected from the Barton Springs segment of the Edwards aquifer in order to assess such effects and determine any potential urban geochemical signature. Both well and spring waters were collected and analyzed for the stable isotopes of water, $^{87}Sr/^{86}Sr$ of dissolved strontium, and common major and minor elements.

Sampling

Locations along the Barton Springs segment of the aquifer were selected, as shown in Figure 5.24, and *a priori* classified as *urban* or *rural*, depending on their location with respect to the urbanization density of Austin. The term *rural* is not used here to designate a undeveloped or agricultural area, but a much less urbanized area. Water in *rural* areas is more likely to infiltrate relatively homogeneously through soils and continue downward through tortuous pathways before it reaches infiltration features (karstic conduits and fractures) and the water table. Because urban areas have increased impervious cover, infiltration through soils is reduced, and runoff is more likely to rapidly reach the infiltration features and flow discretely into the phreatic zone. We hypothesize that two modes of infiltration and flow through the vadose zone should be reflected in the geochemical signature of the water samples. It is also important to bear in mind the complexity of this system as a karst. The general flow pattern in the Barton Springs segment is SW to NE and takes place through fractures and preferential conduits (Halihan et al., 1999). The possible role of soils and the host aquifer rock will also be discussed.

Analytical methods

Variations in radiogenic isotopes of strontium, trace elements, and stable isotopes of oxygen and hydrogen were integrated to define the constraints of groundwater evolution in the Barton Springs segment of the Edwards aquifer and the implications of increased urbanization.

Thirteen samples were collected from well and springs within the Barton Springs Segment on 2/25/2001 and 3/1/2001 at locations shown in Figure 5.24. Temperature, pH, conductivity, and turbidity were measured in situ. Eight different bottles were collected at each site with the purpose of carrying out different chemical analyses. In general, wells were purged for several minutes before taking the sample.

Titration was carried out at The University of Texas Department of Geological Sciences (UT DoGS) facilities and the alkalinity of the samples determined.

Oxygen isotope composition was determined by CO_2 equilibration (Epstein and Mayeda, 1953) in a Micromass MultiPrep automated sample preparation system at 40 degrees C. The isotopic composition of the CO_2 was measured on a VG PRISM Series II mass spectrometer (a standard gas source mass spectrometer). UT DoGS lab internal standard samples BEVO (-2.64 ‰) and BTW (-12.95 ‰) were run before, between, and after the samples in order to assess and correct the drift on the measurements associated with this equipment. Results from two runs were averaged. The oxygen isotope values are given as

$\delta^{18}O$, in ‰ with respect to SMOW. The $\delta^{18}O$ values of all the BEVO samples were averaged for adjustment of the measurements. For the sample from well 58-42-914 only one value was obtained, and for sample 58-50-742 the difference between both analyses was too big to be considered.

Hydrogen isotope composition, δD, for ten of the thirteen samples were successfully determined by converting H_2O into H_2 by zinc-reduction (Coleman et al., 1982) and then determining the isotopic composition on a VG SIRA 12 mass spectrometer. SMOW (+2 ‰) and SLAP (Standard Light Antarctic Precipitation; -425 ‰) standards were run before, during, and after the analysis to assess the magnitude of the corrections needed.

Uncertainty for $\delta^{18}O$ is approximately ±0.15 ‰, and ±2 ‰ for δD (2-sigma external reproducibility for the lab internal standards and are equivalent to the 95% confidence limit).

$^{87}Sr/^{86}Sr$ values for all 13 samples were also determined at the UT DoGS facilities in two different sessions. A general description of the method is described by Banner (2004). Two ml of sample were allowed to evaporate from a Teflon vial. The precipitate was dissolved in strong nitric acid and loaded onto ion exchange columns using a strontium specific synthetic resin. The strontium was extracted by adding nitric acid of varying strengths. The pure strontium was dissolved in a very small amount of phosphoric acid and placed on zone-refined rhenium filaments, which are coated with a Tantalum oxide (Ta_2O_5) with the purpose of oxidizing the strontium. The analyses were carried out by dynamic multicollection on a Finnigan MAT 261 thermal ionization mass spectrometer. A standard sample (NBS 987) was analyzed to allow for interlaboratory comparisons. Values for the standard $^{87}Sr/^{86}Sr$ ratio, internal, and external precisions are compiled in Appendix 2. Blank samples are regularly run to monitor the level of contamination.

The analyses to determine the concentrations of some major and minor elements in the samples were carried out at the University of Minnesota laboratory by ThermoElemental PQ ExCell quadrupole ICP-MS. The method is based on the EPA standard method for ICP-MS analyses 200.8. The standard deviation was calculated for each elemental analysis of each sample. Values for which the concentration of the sample was less than three times the standard deviation are considered to be below the detection limits (BDL). For detection limits and analytical uncertainty on these analyses see Tables A2.3 and A2.4 in Appendix 2.

Results and discussion

The results of the analyses of isotopes of hydrogen and oxygen in the water, isotopic ratio of dissolved strontium, and major and minor constituents are presented in Table 5.4. Regional data from soil water, cave dripwater, surface water, and groundwater is compiled from the literature, as it provides a perspective of groundwater evolution processes and pathways for infiltration in soil, though the vadose zone, and into the phreatic zone.

Isotopes of hydrogen and oxygen

$\delta^{18}O$ and δD values of global meteoric waters lay on a straight line called the Global Meteoric Water Line (GMWL), which was first defined by Craig (1961). Figure 5.25 shows the $\delta^{18}O$ and δD values of nine of the thirteen samples and the GMWL. Urban and rural samples plot scattered and values in general lay on or are very close to the GMWL. The $\delta^{18}O$ values are within the range of values measured by Oetting (1995) and Musgrove (2000) for groundwaters of the Edwards aquifer. Because all the sampling took place on the same day, no temporal variability can be determined; thus, no inference on the amount of recharge can be done, as proposed by Jones and Banner (2000). Figure 5.26 presents a model for the evolution of the evolution of $\delta^{18}O$ and $^{87}Sr/^{86}Sr$ of vadose waters in the Edwards aquifer (Musgrove, 2000; Musgrove and Banner, 2004). Samples in the present study have similar values of $\delta^{18}O$ but lower values of $^{87}Sr/^{86}Sr$. The isotopic composition of oxygen suggests the water-rock interaction has not been enough for a drift on $\delta^{18}O$ towards higher values to happen. Groundwaters and aquifer rocks are not in equilibrium with respect to oxygen. A sample from well 58-50-207 presents lower $\delta^{18}O$ than the general trend and actually resembles values of dripwaters from caverns in the westernmost side of the Edwards aquifer (Musgrove, 2000). This suggests groundwater follows a pathway of relatively short travel-time into this well.

Isotopes of strontium

Groundwater acquires most of its dissolved Sr through interaction with different soils, other waters, rocks, and diagenetic processes along its path. The $^{87}Sr/^{86}Sr$ value reflects the evolution of such interactions and the relative contributions of the different sources of Sr and can, therefore, be used to delineate groundwater flow routes and variations on recharge rates (Banner et al., 1996).

The $^{87}Sr/^{86}Sr$ values determined for the thirteen samples are within the ranges found by Oetting (1995) and Musgrove (2000) for the Edwards aquifer groundwater and for waters in central Texas in general (Figure 5.27). The samples present $^{87}Sr/^{86}Sr$ values close to those of limestone and in some cases close to equilibrium with respect to Sr, indicating waters that have traveled longer and show greater degree of water/rock interaction than soil leachates and dripwaters in the vicinity of the aquifer. The minimum $^{87}Sr/^{86}Sr$ value corresponds to sample 58-57-3ES and is very close to that of Edwards Limestones, suggesting that these groundwaters have undergone extensive interaction with the host rock. The maximum value, sample 58-50-207, is close to that of rainwater, soil waters, vadose water, saline groundwater, and local tap water. Rainwater is not a likely source of strontium in groundwater because the relatively low strontium content of this source (Banner et al., 1994).

Rural wells have lower $^{87}Sr/^{86}Sr$ values relative to the *urban* ones (Figure 5.27). Many samples show values between 0.7079 and 0.7080. As discussed above, this could be due to different modes of infiltration for the more and less urbanized areas. A slower and more diffuse recharge is more likely to happen in areas with little or no impervious cover directing the runoff towards infiltration features.

The lowest $^{87}Sr/^{86}Sr$ values for urban wells are displayed by samples 58-50-225, 58-42-914 (Barton Springs), and 58-50-201. The explanation to this effect may relay on the complexity of the karstic nature of the study area. According to the results of several tracing tests carried out by the Barton Springs/Edwards Aquifer Conservation District in this portion of the aquifer, wells 58-50-225 and 58-50-201 do not lie upon any of the flowpaths defined (Hauwert et al., 1998; and Hauwert, personal communication). Rainwater infiltrating around these two wells travels through the soils and vadose zone before intercepting the conduit network. Water in this segment of the aquifer travels through complex pathways that channel the water towards Barton Springs. Thus, the sample from this location could have the mixed signature of different water facies. The location presenting the highest $^{87}Sr/^{86}Sr$ value (well 58-50-207) is at the starting point of one of these groundwater pathways, likely has not traveled as long through the tortuous vadose paths, and has experienced little interaction with the limestone of the aquifer.

Christian (in preparation) describes the $^{87}Sr/^{86}Sr$ value on surface waters in Austin, increases with increasing urbanization, expressed as the percentage of impervious cover. Recharge from losing streams could also explain the higher values of the ratio found in the *urban* samples.

Minor and major elements

Some of the major and trace elements are represented on a Schoeller diagram in Figure 5.28. A sample from well 58-50-915 stands out by having relatively high sodium and potassium, magnesium, rubidium, and lead, and, especially, high calcium and iron contents. Sodium and potassium are also high in the sample from well 58-50-201, which also has relatively high values of the other elements.

In Figure 5.29, sodium is represented against the $^{87}Sr/^{86}Sr$ value and compared to local dripwaters (Musgrove, 2000). *Rural* groundwaters from the Barton Springs segment of the Edwards aquifer have lower $^{87}Sr/^{86}Sr$ values. The samples from the *rural* wells have sodium concentrations similar to dripwaters, and lower than values for *urban* samples. Elevated concentrations of sodium could indicate pollution of groundwaters in the urbanized areas. However, saline groundwaters known as "badwaters" are present in deeper parts of the aquifer, and define its easternmost boundary (Oetting, 1995; Scanlon et al., 2001). The badwaer line is depicted in Figure 5.24. Badwaters are likely to migrate upward in the aquifer along fractures and faults and mix with the fresh water. *Urban* samples are closer to the badwater line and are more likely to show some mixing. This could explain the higher sodium concentrations found on *urban* samples, but this possibility requires further study. In contrast, dripwaters are not likely to experience contamination from this source.

In order to understand the possible geochemical evolution pathways of these groundwaters, the Sr/Ca and Mg/Ca ratios have been determined and plotted against each other and against the $^{87}Sr/^{86}Sr$ value (Figures 5.30-32). Linear trends of positive slope on the Sr/Ca-Mg/Ca space indicate different water-rock interaction pathways. *Urban* and *rural* samples cover a similar range of Sr/Ca versus Mg/Ca ratios (Figure 5.30). However, a greater amount of scatter on the *urban* samples relative to the *rural* ones may be due to different water-rock interaction processes. Curves in figures 5.31 and 5.32 delineate the evolution of the different ratios for a fluid progressively recrystallizing either calcite or dolomite from an initial fluid based in the composition of soil leachates (Musgrove, 2000; Banner et al., 1989; and Banner and Hanson, 1990). As water-rock interaction increases, so do the Sr/Ca and Mg/Ca ratios. There is an apparently consistent shift towards higher $^{87}Sr/^{86}Sr$ values on the *urban* samples, which is consistent with Christian's (in preparation) findings. Deviations from these models for some urban samples indicate other processes besides dissolution of dolomite and recrystallization of calcite are taking place.

Figure 5.31 shows that *rural* samples evolve following the paths where dolomite is being dissolved and calcite is being precipitated. Samples 58-50-742, 58-42-914 (Barton Springs), 58-42-915, and 58-50-201 lie out of this evolution path. Such chemical signatures could represent the mixing of evolved groundwaters with small fractions of badwaters, which migrate upward through faults and fractures (Oetting, 1995) and is also in agreement with the fact these samples have high sodium concentrations.

Figure 5.32 portrays the Mg/Ca ratio against the $^{87}Sr/^{86}Sr$ ratio. The Mg/Ca ratio is directly proportional to the degree of water-rock interaction and, therefore, also to the residence time of groundwater in the aquifer. Most samples follow the model proposed by Musgrove (2000), Banner et al. (1989), and Banner and Hanson(1990) except for those of *urban* wells 58-42-915, 58-50-211, 58-50-207, and perhaps also 58-42-921 (Upper Barton Springs).

Sample 58-58-121 has a very high lead content (> 20 ppb). Lead levels in both wells 58-58-121 and 58-50-915 are above the EPA's *Criterion Continuous Concentration* for this toxic pollutant (~2.5 ppb) indicating groundwater from these wells may be polluted, which was evident from the Schoeller diagram for well 58-50-915 (Figure 5.28).

Different land uses favor different infiltration modes and therefore different degrees of water-rock interaction. Water is less likely to reach equilibrium with the host rock when water is directed quickly into karstic recharge features, compared to following natural more tortuous pathways into the features.

URBAN RECHARGE

The study of the urban sources of recharge to groundwater in Austin consists of a water balance and the analysis of indicator species of water mains and sewage leakage.

Urban recharge in Austin consists mainly of the indirect and artificial components. Water demand, municipal water uses, and a comparison of the amounts of water served versus the wastewater treated is an estimate the amount of treated water available for recharge (denominated "excess urban water" or EUW below). Next, leakage rates from the water and sewage networks are computed (indirect recharge). Then, the amount of water applied for irrigation (artificial recharge) and evapotranspiration rates are computed. Finally, the total urban recharge is determined.

Trihalomethanes are produced as a consequence of chlorination of tap water during disinfection treatment. Thus, leakage from mains constitutes the sole source of these compounds in groundwater. The "usual suspect" sources of nitrate in waters include fertilizers, sewage, animal wastes, the atmosphere, and the decay of organic matter in the soil. $\delta^{15}N$ of dissolved nitrate was analyzed for tap water, groundwater, surface water, and wastewater from the City of Austin. The distinct $\delta^{15}N$ signatures of the different end-member sources provide a tool to identify the sources of nitrate in groundwater originating from leakage from sewage lines and infiltration of wastewater from on-site treatment systems.

Urban water balance

The main two elements of the urban water budget are the amount of drinking water provided in Austin and the volume of sewage handled by the municipal wastewater treatment plants. The difference between these two parameters provides insight on the potential amount of urban water available for recharge to groundwater (Figure 5.33). The amount of unnacounted for water in the city was analyzed in order to estimate the leakage rates from the water and sewage networks, the urban irrigation rates, and the effect of evapotranspiration. The water balance and additional water-related urban statistics are presented in Table 5.6. Water volumes were converted to millimeters per year (mm/a) in order to normalize them with respect to the areal differences of the water and wastewater service areas and to allow comparison with rainfall and recharge rates.

The population of Austin in the year 2000 was 656,562 people. This is a relatively low population density when compared to other cities in the US and Europe (Figure 5.34). In the same year, the City of Austin

Water and Wastewater Utility had a water service area of 710 km^2, serving 738,229 people (Dan Pedersen, City of Austin Water and Wastewater Utility, personal communication). The wastewater service area is somewhat smaller because some users have on-site sanitation systems (septic tanks) and other political/economical considerations on the districting of the utility services. Austin has a temperate climate with moderate precipitation (813 mm/a). The rates of direct recharge under preurban (DRP) and urban (DRU) conditions are analyzed above and estimated to be around 53 and 31 mm/a respectively. In the year 2000 the Utility served an average of 541,000 m^3/d, which for the period of one year and over the 710 km^2 of the service area represents 278 mm/a. In the same time period an average of 317,000 m^3/d of wastewater were treated, which for one year and the service area (601 km2) represents 193 mm/a. The difference between the amount of water treated for consumption and the sewage that arrives in the wastewater treatment plants reveals the amount of water potentially available from recharge from strictly urban sources (Figure 5.33):

EUW = W – WW = 278 – 193 = 85 mm/a

where

EUW: excess urban water

W: served drinking water

WW: treated wastewater

This figure increases dramatically when the maximum capacities of the water and wastewater treatment plants are compared (207 mm/a); however, the facilities are not currently functioning at maximum capacity. The EUW applied to the environment can be subdivided as leakage from utilities (mainly water and wastewater networks) and irrigation of parks and lawns.

Figures 5.35 through 5.37 show water uses and statistics for the City of Austin. Water demand has increased in the city in the last decade (Figure 5.35), while the amount of gross unbilled treated water has just fluctuated around a 12% (Figure 5.36). The City of Austin Water and Wastewater Utility determines the amount of drinking water lost from the distribution system as the difference between served water and billed consumption. The average water loss for the interval is 11.23% (Dan Pedersen, personal communication), but for the water balance of the year 2000, a gross unbilled loss of 12% was adopted (Austin American Statesman, 1998). The Utility breaks the gross unbilled water into "unbilled uses" and "losses" (Figure 5.37). Unbilled uses represent 6.8% of the total treated water and include fire fighting water, thefts, municipal swimming pools, leakage, and water mains breakages. The last two represent less than 2.01% of the total treated water, which, when added to the 5.7% of the water that is simply "lost", results on a maximum leakage rate of 7.7%, or expressed as mm/a:

$$WL = W \times 0.077 = 278 \times 0.077 = 21 \text{ mm/a}$$

where

WL: drinking water leakage rate

A 5% leakage is assumed for the wastewater collection network. Thus the original amount of water that should have reached the wastewater treatment plants can be calculated, and a sewer leakage rate of 10 mm/a determined as follows:

$$WWL = (WW / 1 - 0.05) \times 0.05 = 10 \text{ mm/a}$$

where

WWL: wastewater leakage rate

The third pathway for the water that never reaches the WWTP is irrigation of parks and lawns, which is assumed to take place over the whole non-impervious (i.e., pervious) fraction of the area of the city. The amount of water applied to watering in Austin is determined as the difference after subtracting the leakage rates and adjusting for the pervious area:

$$I = (EUW – WL – WWL)\ PA = (85 – 21 – 10)\ 725/437 = 54 \times 725/437 = 90 \text{ mm/a}$$

where

I: irrigation rate

PA: pervious-area ratio (see Table 5.3)

A fraction of the water leaked or applied as irrigation will be lost to evapotranspiration and the rest will become groundwater recharge. Evapotranspiration (ET) is a measurement of the total amount of water lost from the soil into the atmosphere as the result of direct evaporation and respiration (transpiration) of plants, both processes often impossible to separate. Assessing the role of evapotranspiration is a difficult task that involves observations of the surface temperature, atmospheric humidity, wind speed and soil moisture conditions, as well as of land use and land cover.

As different plants have different water requirements, and thus different ET rates, a standard rate referred to as the reference evapotranspiration (ET_0) is used. ET_0 is the potential ET for a cool season grass, 4-inches tall, in a deep soil, and under well watered conditions (TexasET, online). ET_0 depends on the climate and varies from location to location. The water requirements of specific crops and turf grasses can be calculated as a fraction of the ET_0, known as the crop coefficient (Kc) or turf coefficient (Tc). Crop coefficients vary depending on the type of plant and its stage of growth (FAO, online). For warm season grasses, such as St. Augustine, the Tc is 0.6 throughout much of the year, while for cool season grasses, such as rye, it is 0.8. However, park and lawn irrigation rarely accounts for full plant requirements in order to maintain a healthy, attractive turf with as little water as possible and to reduce grass clipping production. Thus, we reduced the water requirement by an allowable stress coefficient (AS), which ranges from 1 for no stress conditions to 0.4 for very high stress. The plant water requirement (PWR) is estimated as follows:

$$PWR = ET_0 \times Tc \times AS$$

Daily Potential Evapotranspiration of a Grass Reference Crop (ET_0) data for the Austin area was obtained from the Texas Evapotranspiration Network (TexasET, online). The times series spans from 2000 to the present, but the record is only complete for the year 2003.

PWR for the period of one year was calculated for the non-impervious fraction of the city area under three scenarios: 1) a vegetative cover with the minimum evapotranspirative capability (Kc = 0.2) under very high water-stress conditions (AS = 0.4); 2) a highly evapotranspirative vegetation with no water restrictions (Kc = 0.8; AS = 1); and 3) a vegetative cover comprised of different grasses, shrub, and trees under normal stress (Kc = 0.6; AS = 0.6). The resulting PWR after adjusting for the pervious fraction of the city area (437/725 ratio) were 49, 219, and 548 mm/a for each scenario respectively (Table 5.7). The later and the former represent extreme PWR values, while the middle one is an intermediate value more in agreement with the study area. In cities, a fraction of the PWR will be satisfied by precipitation and some by urban irrigation; however, it is virtually impossible to discern the relative contribution to evapotranspiration from each of these sources. Precipitation and irrigation add up to the total water applied to the non-impervious surfaces in Austin as follows:

$$SWA = P + I = 813 + 90 = 903 \text{ mm/a}$$

where

SWA: surface water application

P: precipitation

Thus, precipitation contributes 90% of the surface water application, and irrigation contributes the other 10%. We can assume they contribute to evapotranspiration in the same proportions and determine the fraction of the PWR satisfied by irrigation (PWRi) to be 5, 22, and 54 mm/a for each scenario. Then irrigation return flow or irrigation recharge (IR) amounts to 49, 32, and -1 mm/a, depending on the scenario, as determined by:

IR = EUW – WL – WWL – PWRi = 85 – 21 – 10 - PWRi

where

IR: irrigation return flow, or irrigation recharge

PWRi: plant water requirement satisfied by irrigation

Finally, the potential recharge rate (R) for the city of Austin can be computed as:

R = DRU + EUW – PWRi

where

R: total recharge

DRU: direct recharge – urban conditions

Hydrochemistry

The potential sources of groundwater recharge in Austin are precipitation, tap water (through leakage or irrigation), and sewage.

Thus, groundwater, surface water, water treatment plant outflows, and wastewater treatment plant inflows were sampled.

Groundwater samples from the Barton Springs system were collected on 8/30/02. The individual discharge points are known as the Barton Springs Pool, Eliza Springs, Old Mill Springs, and Upper Barton Springs.

Water samples were collected from water and wastewater treatment plants (WTP and WWTP respectively) on 9/6/2002. Finalized treated water samples were collected at the three local WTPs: Green, Ullrich, and Davis. Samples of the untreated flow arriving into Green and Ullrich WTPs were also collected. Green WTP draws water from Town Lake, and Ullrich WTP draws water from Lake Austin. Both lakes are reservoirs created by damming along the Colorado River within the city limits. At Govalle WWTP a sample of the raw inflow was collected after the removal of large solids. At the Walnut Creek WWTP, 4 samples were collected: raw influent sewage, before chlorination, after chlorination, and final treated effluent before it is dumped into the Colorado River, downstream of Austin.

Seven surface water samples were collected on 7/15/2003 along Waller Creek to be analyzed for trihalomethanes, as the baseflow of this creek could be mainly originated from leakage from mains (Ging et al., 1996; and Christian, in preparation). The sampling points from downstream to upstream are: Waller Creek at 3th Street, two samples at 9th Street, at 24th Street, at the small tributary flowing through Adams-Hemphill Park, at the Hancock Golf Course (near 41st Street), and at Skyview Road. One sample was collected from a small buried tributary of Shoal Creek we call "Little Shoal", near the confluence at 4th Street.

Analytical methods

Trihalomethane analyses were carried out at the Environmental and Water Resources Laboratory of the Department of Civil Engineering at The University of Texas. The method employed is a variation of the US EPA method 551.1 (revision 1.0) for the determination of disinfection byproducts, chlorinated solvents, and halogenated pesticides/herbicides in drinking water by liquid-liquid extraction with pentane and gas chromatography. Samples were analyzed for chloroform, bromodichloromethane, dibromochloromethane, and bromoform. All four species were summed to represent total trihalomethanes.

δ15N of dissolved nitrate was analyzed at Coastal Science Laboratories, Inc., in Austin, Texas. The nitrate was obtained by reducing it to ammonia with DeVarda's Alloy (Cu:Al:Zn) under alkaline conditions, so that the NH_3 is distilled off into an acid trap (0.003N HCl, pH ~2.5). After quantitative trapping, the pH of the trap is raised to ~3.5, and the NH_4^+ is trapped on a special molecular sieve (artificial zeolite, Union Carbide W-85). The sieve is filtered, dried (~60C for a couple of days), and run by continuous flow mass spectrometry. The mass spectrometry setup starts with an elemental analyzer (Carlo-Erba NA1500) operating under normal conditions. The zeolite sample is combusted at 1020° C, and combustion products are passed over a chromium oxide oxidizing bed and then an elemental copper reduction segment. The produced pure CO_2 and N_2 are separated by internal GC. The effluent stream (UHP He carrier gas) is fed into the isotope ratio mass spectrometer (VG, SIRA-10) via an open split capillary. Instrument software is used to calculate the isotopic ratio of the N_2 relative to a reference gas. Laboratory standards and NIST and IAEA reference materials are run daily for calibration of the mass spec reference gas.

Some additional $^{87}Sr/^{86}Sr$ ratios were obtained for the raw water inflow to two municipal water treatment plants, and raw sewage from two wastewater treatment plants (Figure 5.27). The analytical method is described above. Two blank samples were run during this period of the study, which had 3.8 (6/19/03) and 17.1 (8/8/03) picograms of Sr, respectively.

Major and minor elements were analyzed at the UT DoGS facilities. Major and minor cations were analyzed by ICP-MS, and anions were analized by ion chromatography.

Results

The results of the analyses of total trihalomethanes, isotopes of nitrogen, and major and minor ions from this portion of the study are summarized in Table 5.8.

Although individual species of trihalomethanes were analyzed, only total trihalomethanes, the sum of all trihalomethane species, are discussed here. However, the results of individual trihalomethane species are collected in Appendix 3.

As summarized in Figure 5.38, no total trihalomethanes were detected in surface water courses nor at the lake-intakes of the water treatment plants. Finalized treated water at all thee water treatment plants was found to have the highest levels of total trihalomethanes, all above 30 $\mu g/\ell$. Raw sewage with no further treatment than the separation of large solids presented values lower than 10 $\mu g/\ell$. Total trihalomethanes was neither detected in surface water nor in groundwater from the Barton Springs system.

Values of $\delta^{15}N$ of the dissolved nitrate were obtained for samples from surface water, sewage, tap water, and groundwater. The results are contrasted to the compilation of values published in the literature in Figure 5.39.

The lowest $\delta^{15}N$ values correspond to water from Town Lake and Lake Austin, -1.3 and -1.6 respectively. Water treatment plant outflow (tap water) presented values of 0.7 and 1.2 $^0/_{00}$. Groundwater from the

Barton Springs system showed values between 2.0 and 3.4 $^0/_{00}$. The highest values were found for sewage, 7.5 and 7.9 $^0/_{00}$.

$^{87}Sr/^{86}Sr$ ratios of drinking water prior to treatment, and raw wastewater are also presented in Table 5.8 and Figure 5.27.

Discussion

Austin has a temperate climate with moderate precipitation, yet the amount of water supplied (W = 278 mm/a) seems to be small compared to annual mean rainfall (P = 813 mm). This can be explained by the low-density urban style of Austin compared to other cities in the US and Europe (Figure 5.34).

A significant amount of the supplied drinking water never reaches the sewage treatment plants (EUW = 85 mm/a, on average). The fate of the lost water must be one of the following:

1. Leakage from mains and sewers
2. Irrigation of parks and lawns
3. Infiltration of septic tank effluent
4. Consumption (drawn out of the water cycle) by:
    a. Human and plant metabolic fixation of water
    b. "Virtual water" – water incorporated into industrial products, mainly food. In the case of Austin, this type of water may constitute a source rather than a sink, but there's no significant food industry in the city
    c. Diagenetic processes

For the City of Austin, 4.a, b and c are assumed to be practically negligible.

The City claims a loss of about 12% of the served water. A comparison of this value with main loss rates around the world indicates that Austin has one of the most efficient distribution systems (Table 2.1). Some of the factors contributing to the small loss of water in Austin include:

1. Underestimation of water losses by the municipal service.
2. Underestimation of breakage and leakage rates by the municipal service.
3. A presumably outstanding maintenance of the network by the municipal service.
4. The relatively young age of a large portion of the network.

Mains leakage was determined to be around 7.7% of the treated water. Thornton's (2002) Water Loss Control Manual indicates 60% of the unaccounted for water by a municipal service can be attributed to leakage from the mains, so that for a 12% unaccounted for water, the leakage rate would be 7.2%, which is in close agreement with our estimate. This mains leakage rate is also conservative because it does not account for leakage "on the other side of the meter", which occurs within the user's premises and is billed.

Leakage from wastewater pipes is generally less than 5%, as a review of the literature demonstrates, which in Austin represents 10 mm/a of potential recharge.

On-site sanitation systems infiltrate most of the water they receive. Unfortunately the amount of such devices existing in Austin has not been determined, but the population densities of both water and wastewater service areas are similar, 1040 and 1141 people served per km^2 of service area, respectively. Thus I infer that the effect of septic-tanks is small.

The treated water that does not reach the sewage treatment plants (EUW) and is not lost to leakage from pipes equals the amount of water used on the irrigation of parks and lawns, and is determined by subtracting the leakage rates from the EUW. Irrigation rates range from 54 to 176 mm/a if we consider average water and sewage treatment rates, or maximum treatment capacities, respectively. A fraction of the water applied as irrigation will turn into recharge to the groundwater (IR), some of it will be lost to evapotranspiration, and some will be lost to interflow. Plant water requirements for the study area were determined based on local reference evapotranspiration rates, amounting to 49, 219, and 548 mm/a for three different evapotranspiration scenarios. This allowed the determination of the amount of urban irrigation water used by plants (PWRi), and, using that information, for the determination of irrigation return flow rates (IR).

Finally, the potential recharge to the groundwater is computed as the sum of the direct recharge from precipitation (DRU), leakage rates (WL and WWL), and irrigation return flow (IR).

In summary, recharge has increased significantly with the advent of urban development. Direct recharge from rainfall has decreased from 53 to 31 mm/a because of the increased impervious cover. However, additional sources of recharge have been introduced as the result of urban activity, accounting for approximately 85 mm/a. Of these 85 mm/a approximately 31 are lost to leakage from mains and sewers, and the rest is used for irrigation. A conservative estimate renders total groundwater recharge under urban conditions at around 94 mm/a, which doubles that prior to urbanization, 32 of which originate from urban irrigation, making this the principal urban source of groundwater recharge in Austin. This is not surprising in a city of not very humid climate, composed mainly of single-family houses with yards and lawns where high water-demanding grass species are common. The next most relevant source of recharge is direct recharge from rainfall (DRU = 31 mm/a), followed by leakage from the drinking water distribution (WL = 21 mm/a), and the sewage network (WWL = 10 mm/a).

The absence of trihalomethanes in groundwater may be due to dilution of mains leakage into the groundwater and the detection resolution of $1\mu g/\ell$. However, it could alternatively be attributed to the high volatility of these compounds.

Septic tanks are designed to infiltrate sewage; however, they do not seem to have contributed to the degradation of the quality of the groundwater of the Edwards aquifer in the past (St. Clair, 1979). Kreitler and Browning (1983) did not find $\delta^{15}N$ values elevated enough to suggest the presence of animal waste in the Edwards aquifer. However, the City of Austin has detected higher values in more recent years (David Johns, City of Austin Watershed Protection Department, personal communication). In addition, a review of the literature indicates that $\delta^{15}N$ values of dissolved nitrate in sewage are expected to be higher than 10 $^0/_{00}$. In the same respect, the $\delta^{15}N$ of Town Lake and Lake Austin should be similar to those of the Colorado River reported by Kreitler and Browning (1983). However, our results are significantly lower, which may cast doubt upon the reliability of the results and thus, no mass balance between the different sources has been attempted.

$^{87}Sr/^{86}Sr$ ratios of wastewater are lower than those of drinking water and rainwater, rendering this geochemical parameter a potential tracer of urban recharge. Questions arise regarding the potential contribution of drinking water and wastewater to the groundwater samples from the Barton Springs segment of the Edwards aquifer collected on the first sampling campaign.

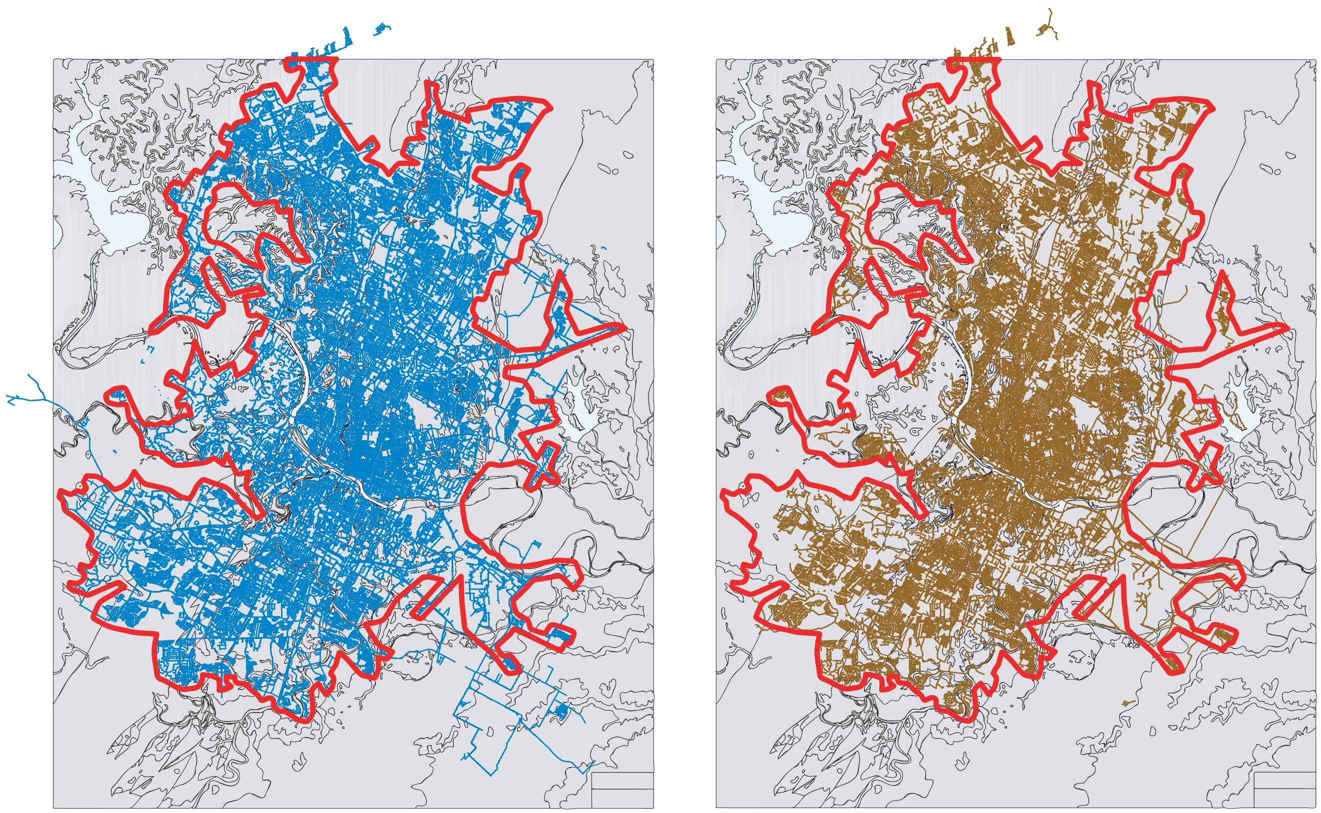
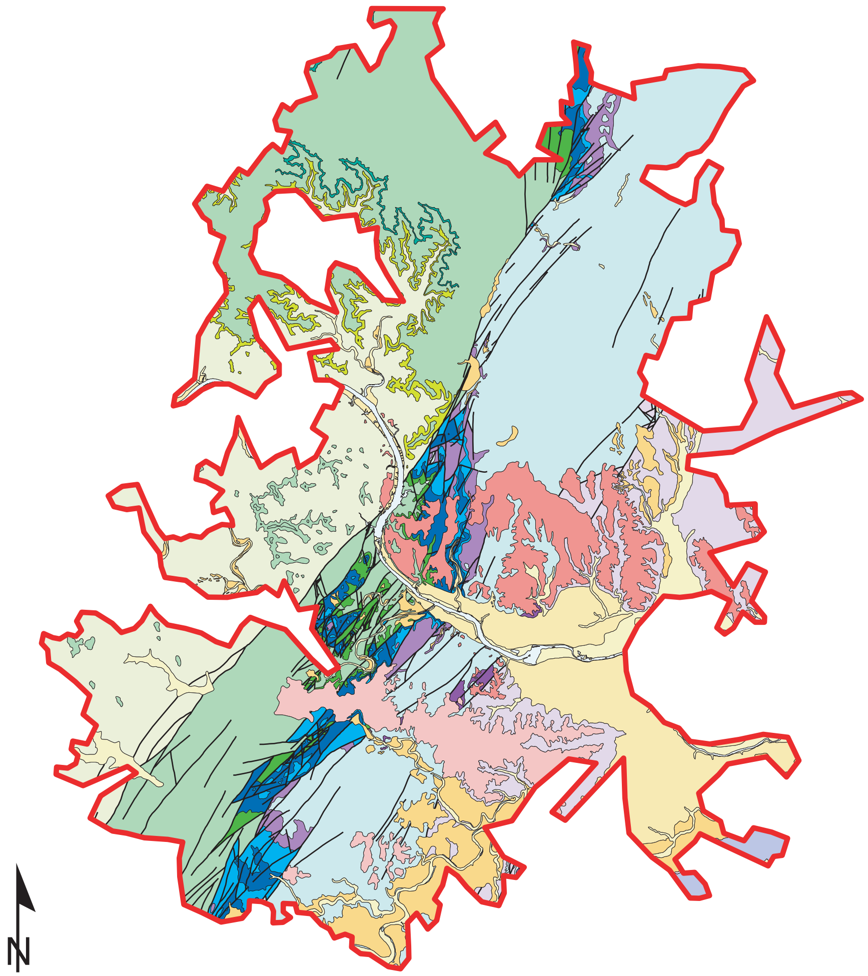


Figure 5.1: Water and wastewater service area defined by the spatial distribution of mains and sewers in Austin, Texas.

Figure 5.2: Geology within the service area.



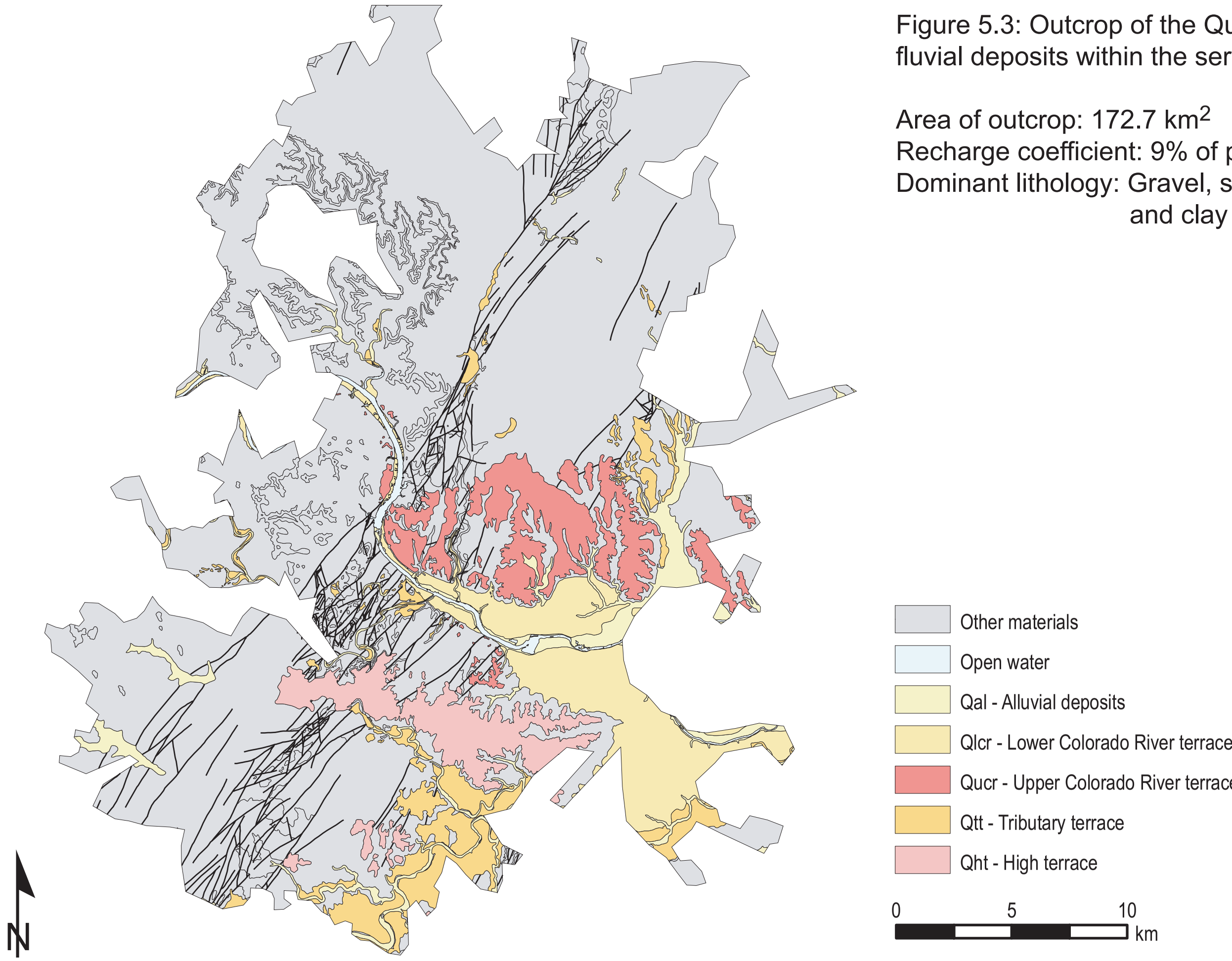


Figure 5.3: Outcrop of the Quaternary fluvial deposits within the service area.

Area of outcrop: 172.7 km^2
Recharge coefficient: 9% of precip.
Dominant lithology: Gravel, sand, silt, and clay

Figure 5.4: Outcrop of the Taylor and Navarro Formations within the service area.

Area of outcrop: 48.6 km^2
Recharge coefficient: 1% of precip.
Dominant lithology: Clay

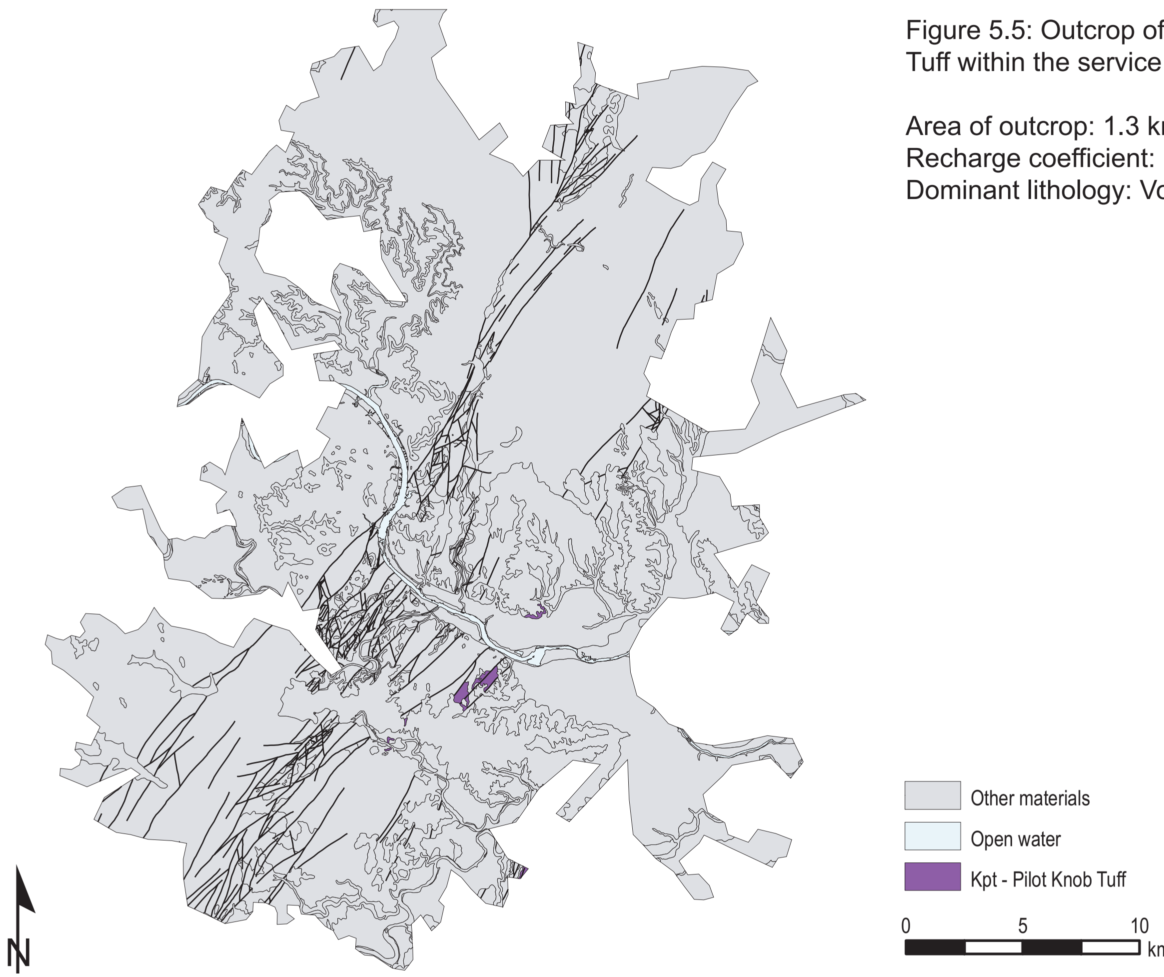


Figure 5.5: Outcrop of the Pilot Knob Tuff within the service area.

Area of outcrop: 1.3 km^2
Recharge coefficient: 1% of precip.
Dominant lithology: Volcanic Tuff

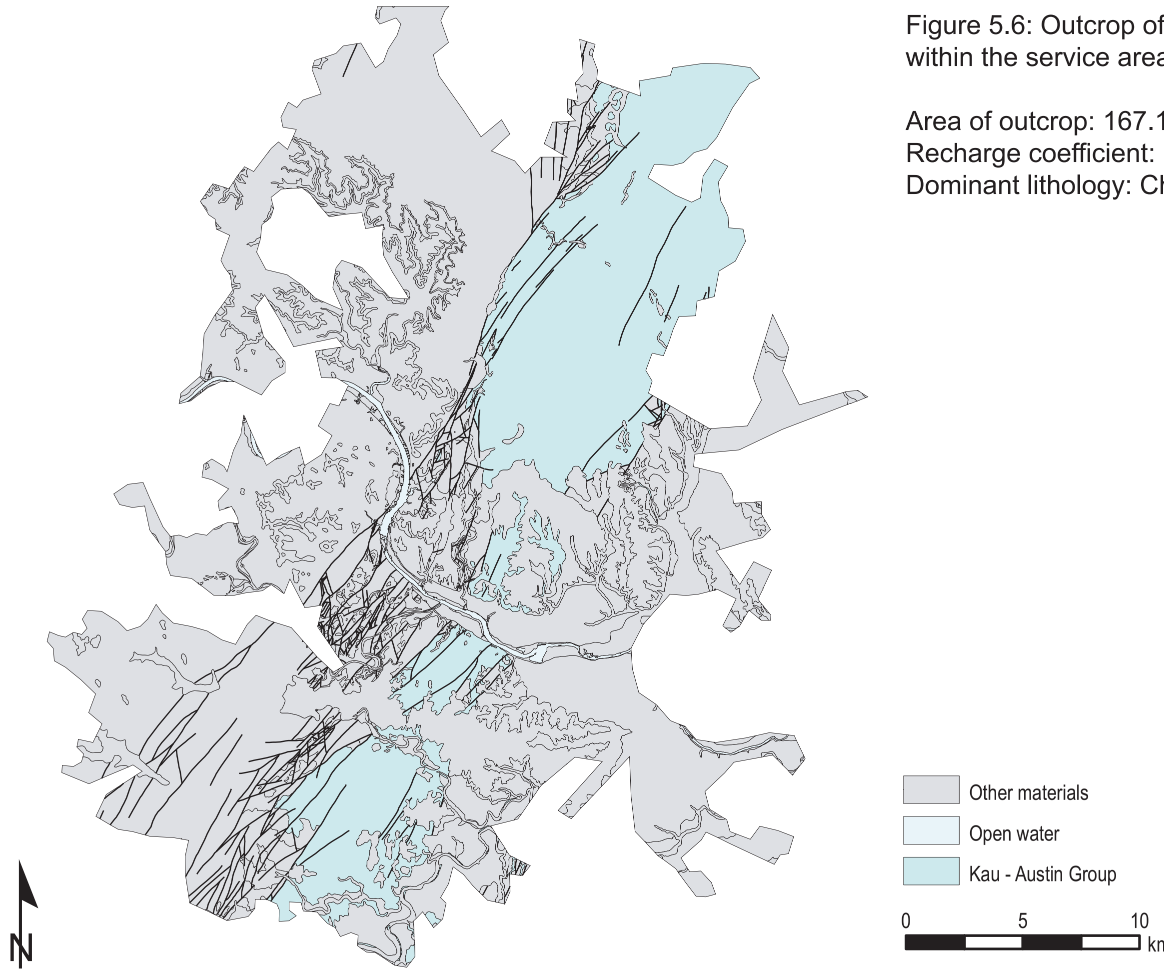


Figure 5.6: Outcrop of the Austin Group within the service area.

Area of outcrop: 167.1 km^2
Recharge coefficient: 1% of precip.
Dominant lithology: Chalk

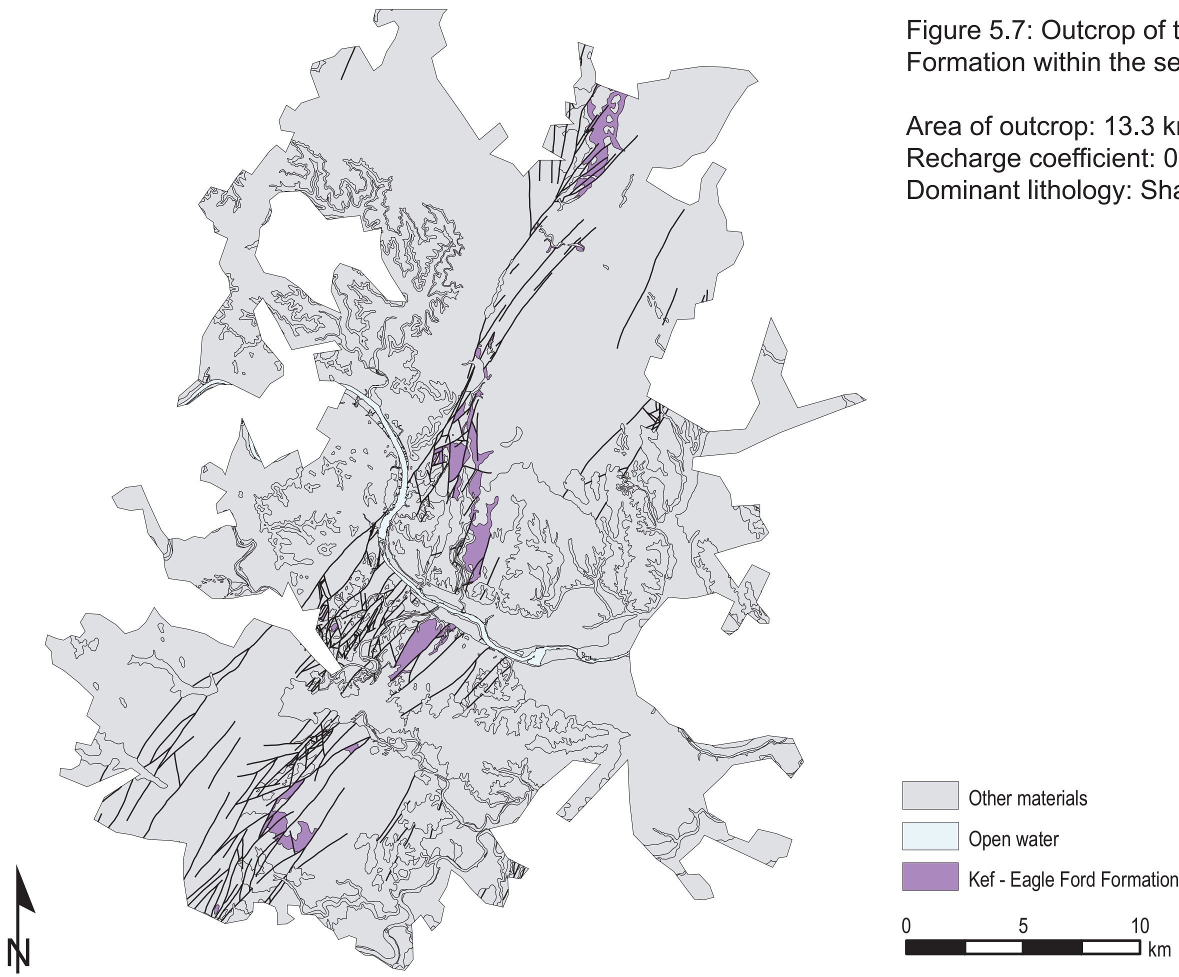


Figure 5.7: Outcrop of the Eagle Ford Formation within the service area.

Area of outcrop: 13.3 km^2
Recharge coefficient: 0% of precip.
Dominant lithology: Shale

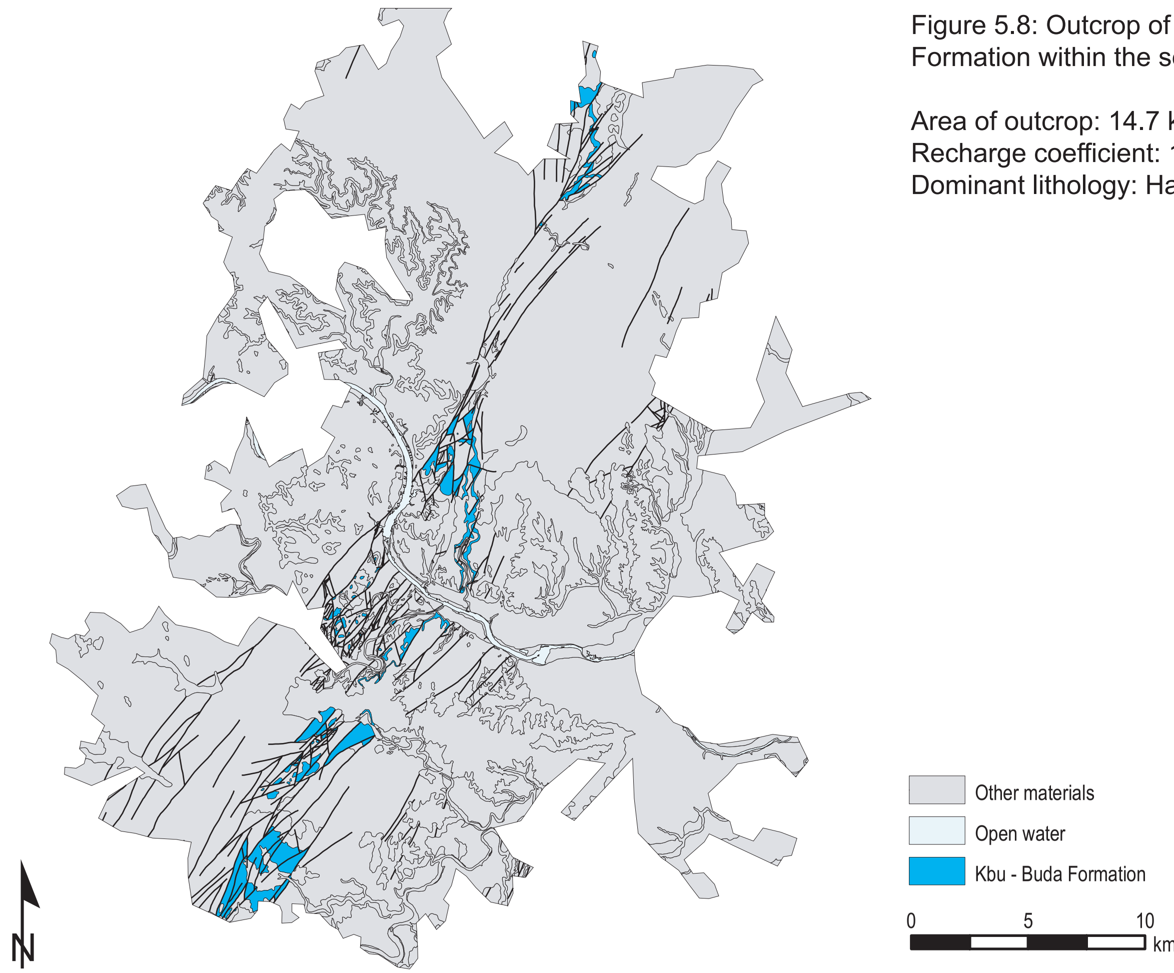


Figure 5.8: Outcrop of the Buda Formation within the service area.

Area of outcrop: 14.7 km^2
Recharge coefficient: 1% of precip.
Dominant lithology: Hard limestone

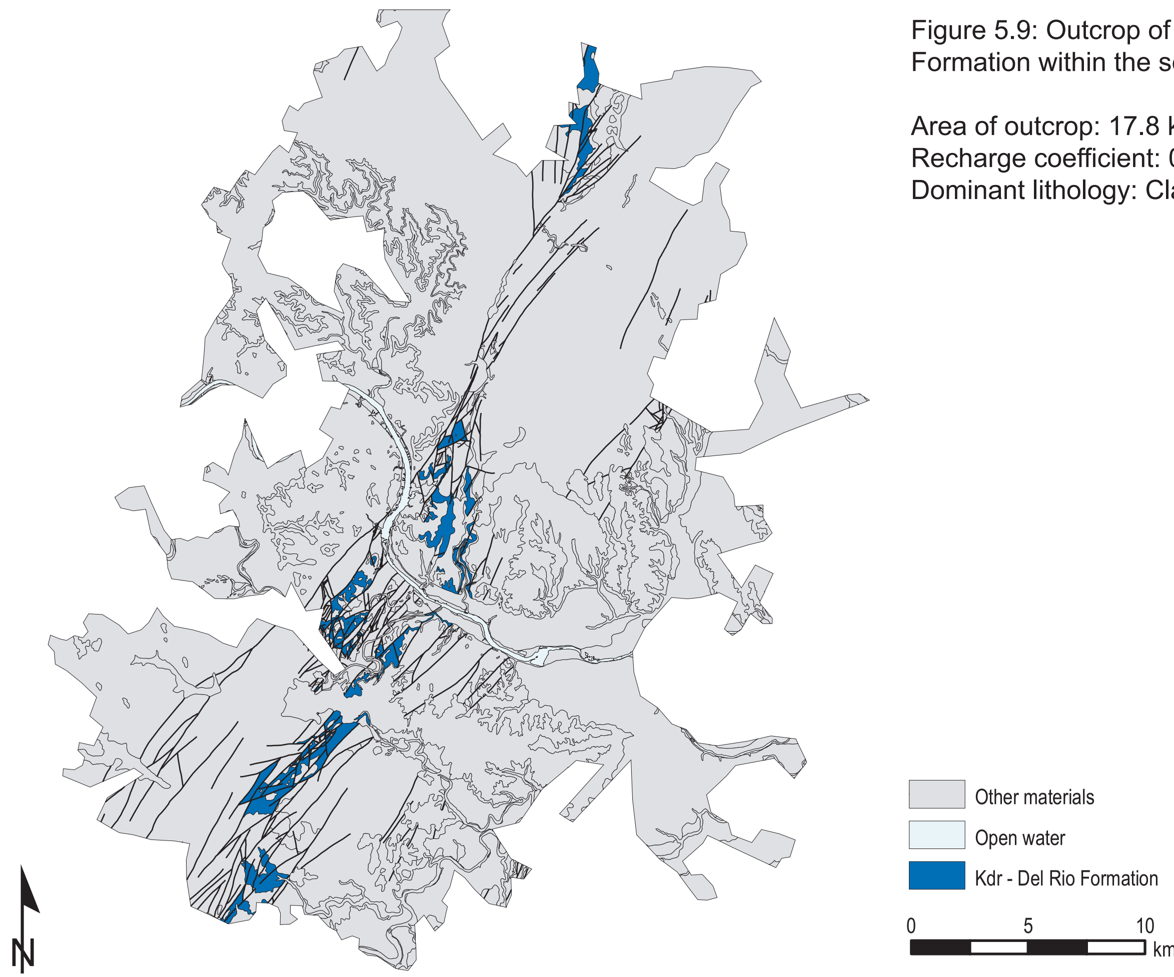


Figure 5.9: Outcrop of the Del Rio Formation within the service area.

Area of outcrop: 17.8 km^2
Recharge coefficient: 0% of precip.
Dominant lithology: Clay

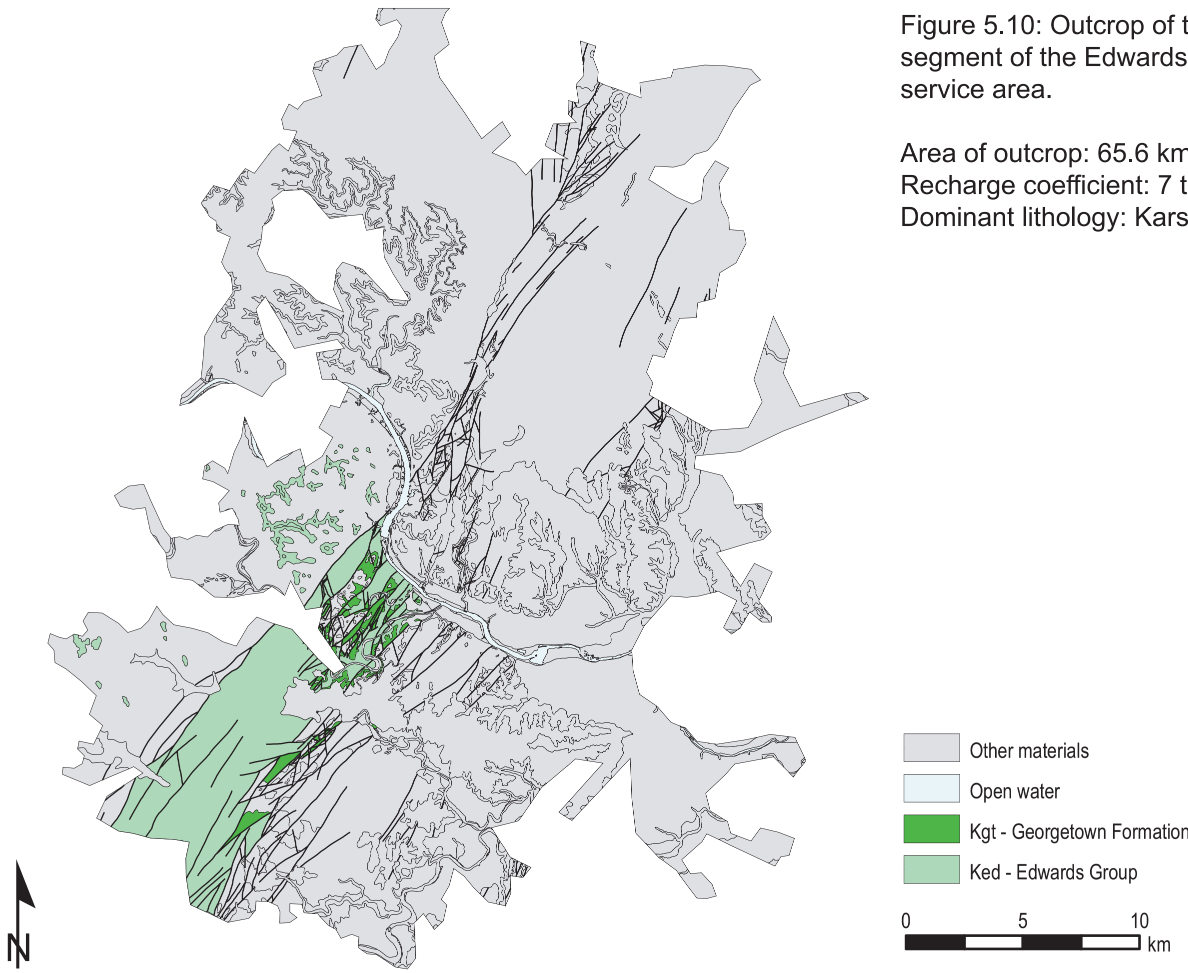


Figure 5.10: Outcrop of the Barton Springs segment of the Edwards aquifer within the service area.

Area of outcrop: 65.6 km^2
Recharge coefficient: 7 to 8.2% of precip.
Dominant lithology: Karstic limestone

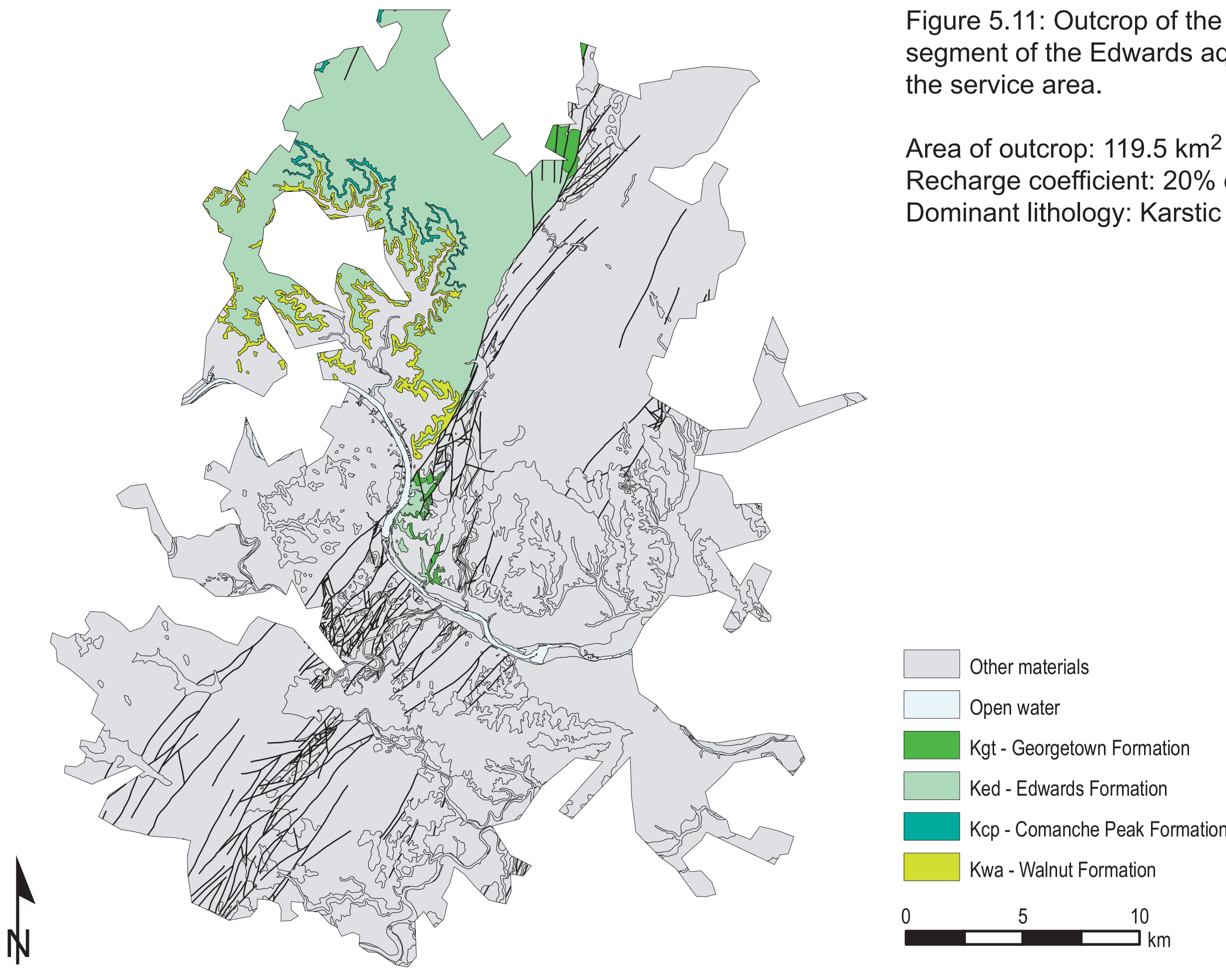


Figure 5.11: Outcrop of the Northern segment of the Edwards aquifer within the service area.

Area of outcrop: 119.5 km^2
Recharge coefficient: 20% of precip.
Dominant lithology: Karstic limestone

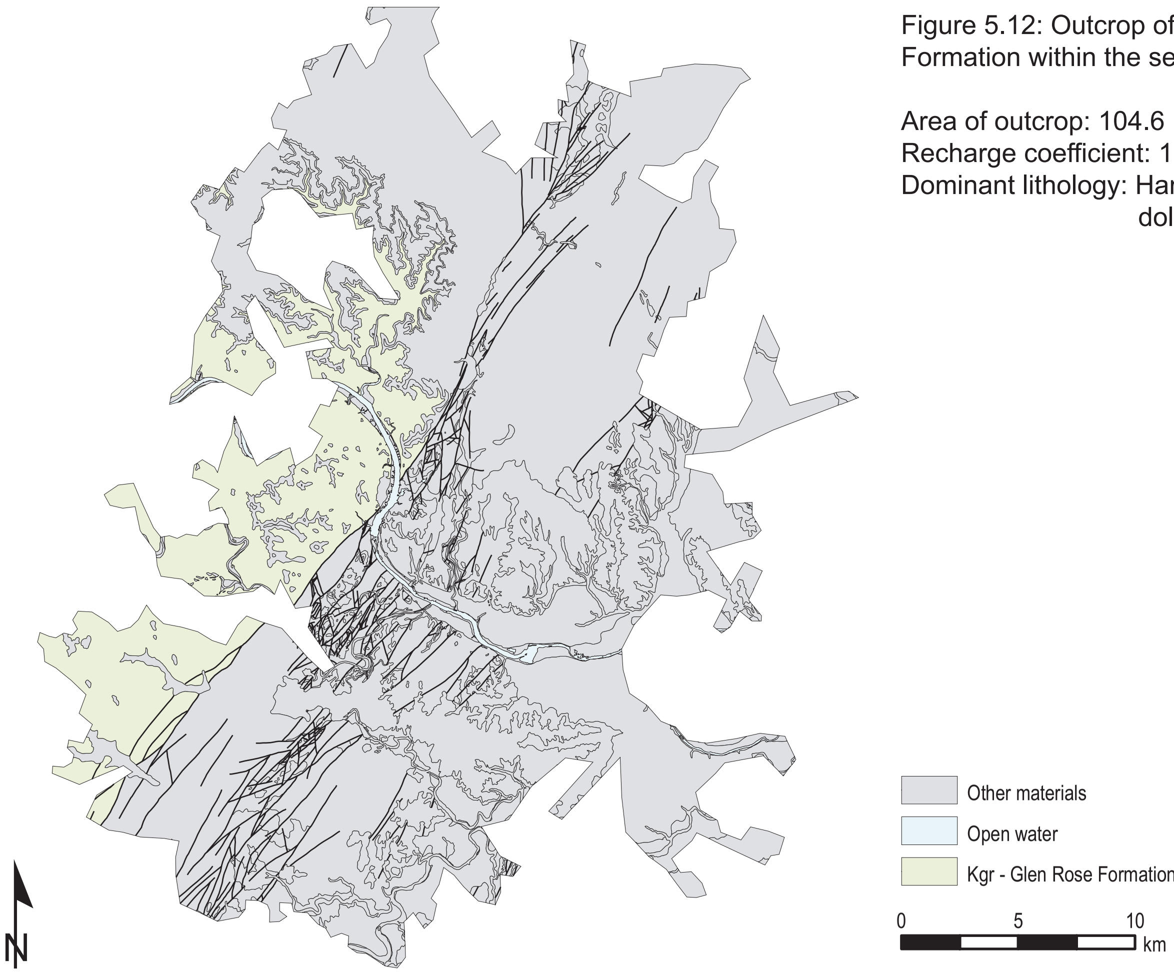


Figure 5.12: Outcrop of the Glen Rose Formation within the service area.

Area of outcrop: 104.6 km^2
Recharge coefficient: 1% of precip.
Dominant lithology: Hard limestone and dolostone

Table 5.1: Direct recharge under preurban conditions.

	PERVIOUS AREA km^2		PPT mm/a	DIRECT RECHARGE (% of ppt)	DIRECT RECHARGE mm/a	DIRECT RECHARGE l/a	DIRECT RECHARGE mm/a
ALLUVIUM *(gravel, sand, silt, clay)*	27.7						
LCR deposits *(sand, silt, clay)*	46.8						
UCR deposits *(gravel, sand, silt)*	37.3						
TRIBUTARY TERRACE dpsts *(lmst gravel, sand, mud)*	33.8						
HIGH TERRACE dpsts *(lmst, sand, gravel, caliche)*	27.1	172.7	813	9 [1]	73.2	12,637,368,398.0	
NAVARRO Gp *(clay)*	3.5						
TAYLOR Fm *(clay)*	45.1	48.6	813	1 [2]	8.1	395,465,544.8	
PILOT KNOB TUFF *(tuff)*	1.3	1.3	813	5 [2]	40.7	51,560,757.1	
AUSTIN Gp *(chalk, lmst to marl)*	167.1	167.1	813	1 [3]	8.1	1,358,883,895.5	
EAGLE FORD Fm *(shale)*	13.3	13.3	813	0 [2]	0.0	0.0	
BUDA Fm *(hard lmst to marl)*	14.7	14.7	813	1 [2]	8.1	119,220,347.5	
DEL RIO Fm *(shale with high clay content)*	17.8	17.8	813	0 [2]	0.0	0.0	
GEORGETOWN Fm *(lmst to marl)*	9.4						
EDWARDS Fm *(dol, lmst, hard lmst, collapsed lmst)*	155.6						
COMANCHE PEAK Fm *(lmst)*	2.5						
KAINER/WALNUT Fm *(hard lmst to marl)*	17.6	185.1					
Barton Springs segment		65.6	813	8.2 [4]	67.1	4,399,391,369.4	
Barton Springs segment		65.6	813	7.0 [5]	56.8	3,727,074,124.2	
Northern segment		119.5	813	20 [6]	162.6	19,427,216,416.4	
GLEN ROSE Fm *(marl, dolst, lmst)*	104.6	104.6	813	1 [2]	8.1	850,584,543.4	
OPEN WATER		5.6			0.0		
total pervious area		730.9					
total city area		730.9					
TOTAL *minus open water*		725.3	813			38,567,374,027.0	53.2

(1) Based on Wickham (1991)
(2) Assumed
(3) Mace (1998)
(4) R = infiltration of ppt + stream loss; based on Slade et al. (1985), Sharp (1990), and Scanlon et al. (2001)
(5) R = infiltration of ppt; based on Slade et al. (1985), Sharp (1990), and Scanlon et al. (2001)
(6) Jones (2003)

Table 5.2: Annual precipitation in Austin and groundwater recharge to the Barton Springs segment of the Edwards aquifer.

	PRECIPITATION (mm/a)	RECHARGE (mm/a)
1980	695.45	62.84
1981	1161.54	103.60
1982	676.402	32.68
	NOOA, online	*after Slade et al., 1985*

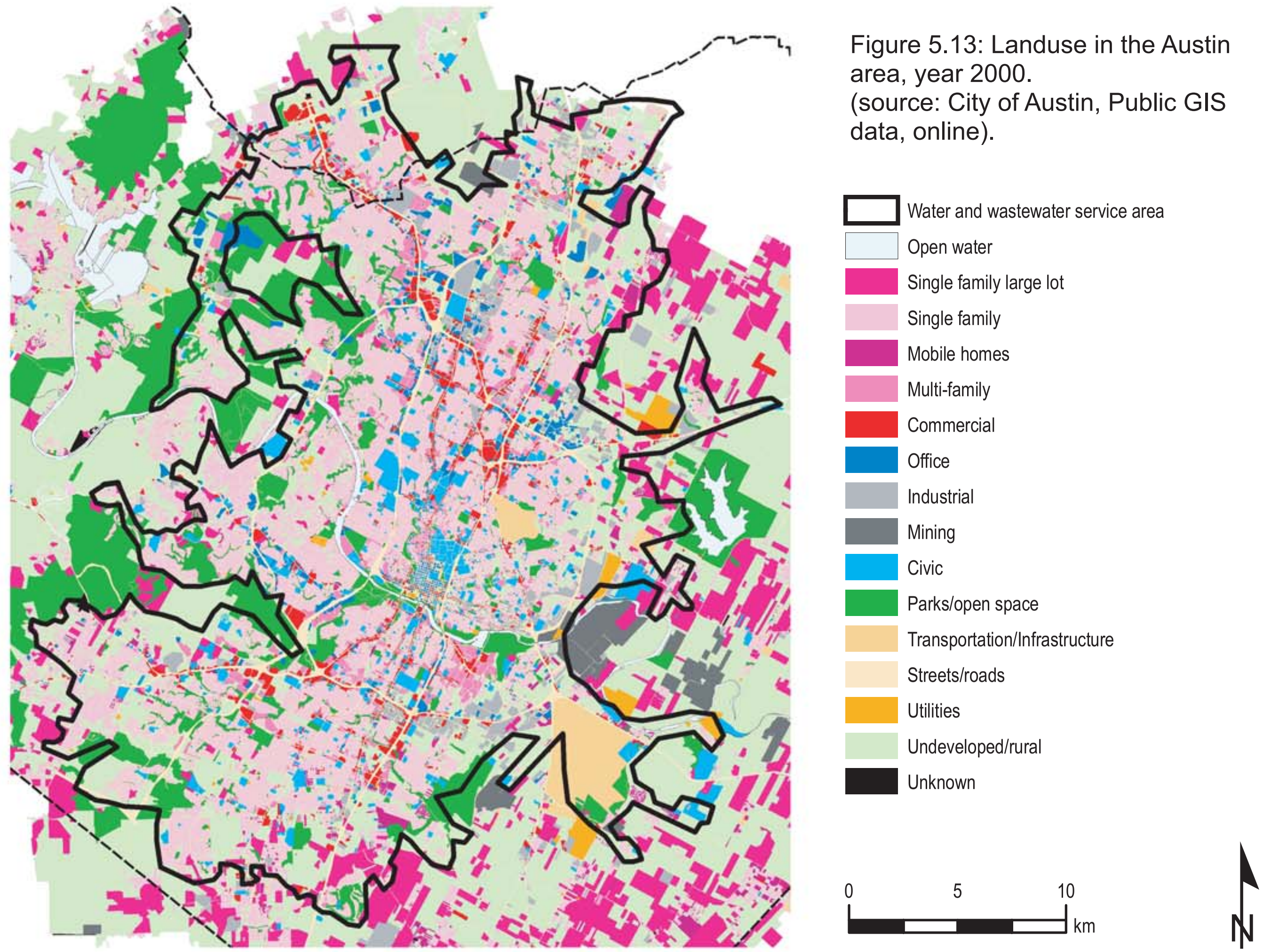


Figure 5.13: Landuse in the Austin area, year 2000.
(source: City of Austin, Public GIS data, online).

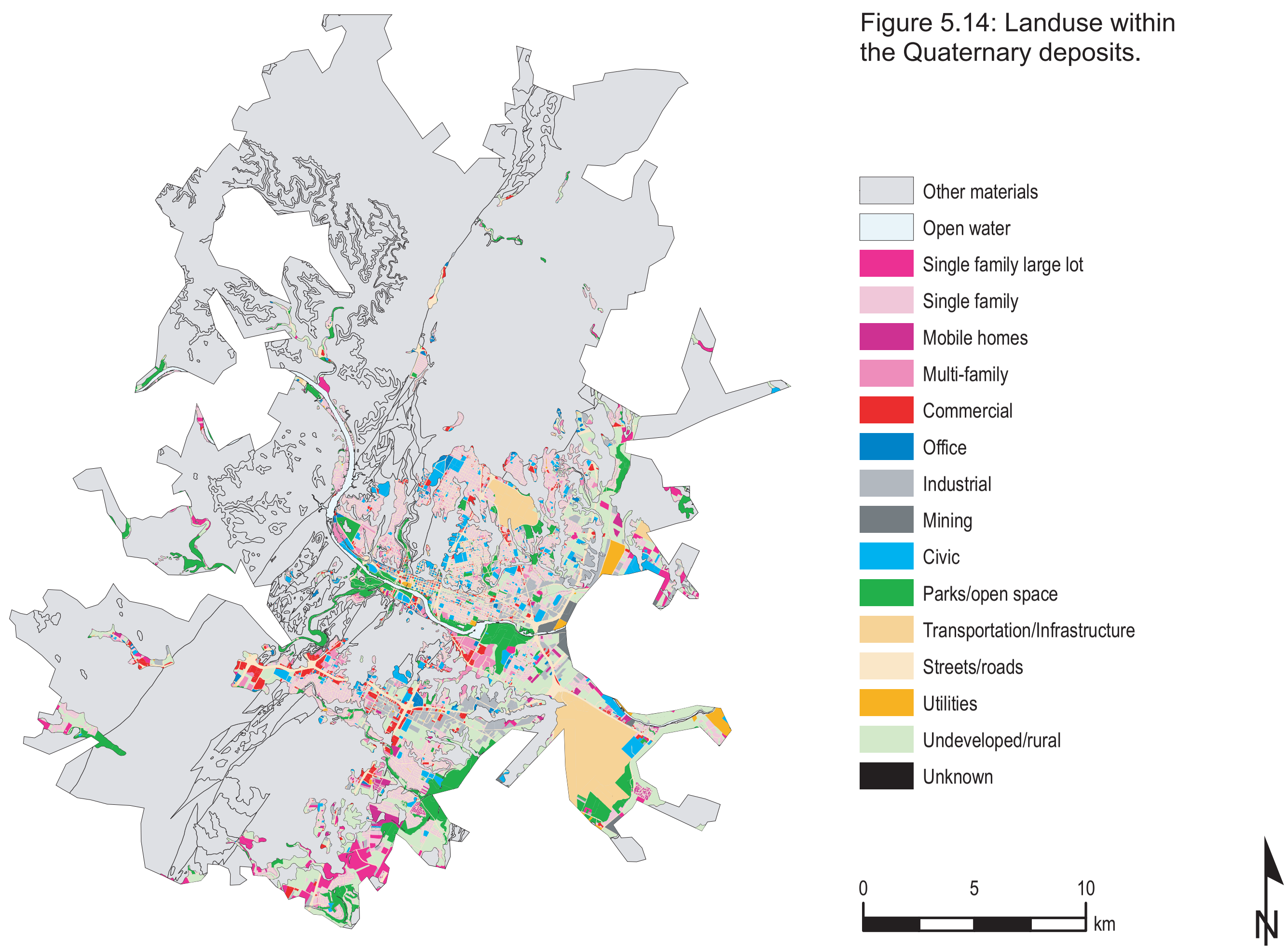


Figure 5.14: Landuse within the Quaternary deposits.

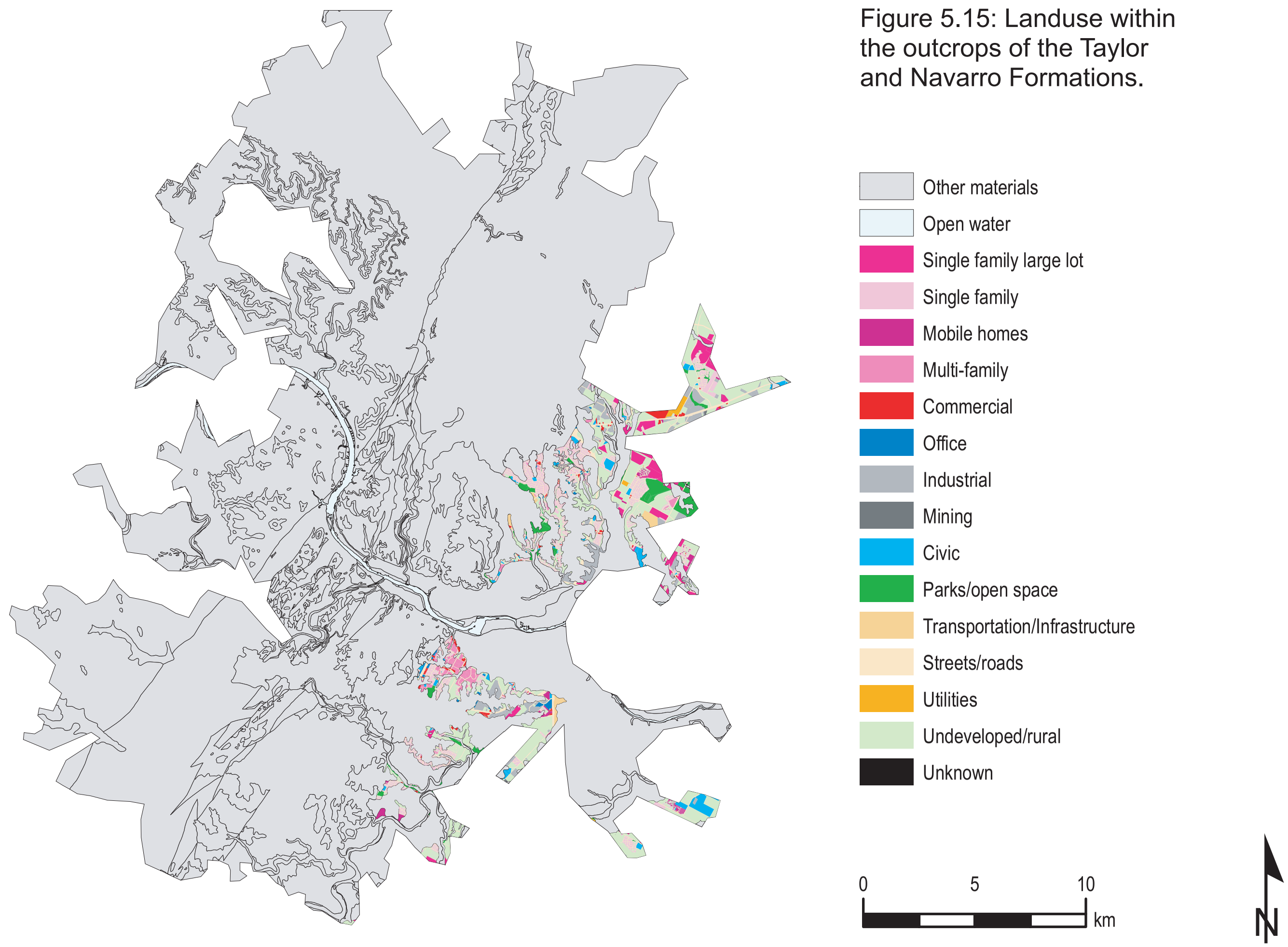
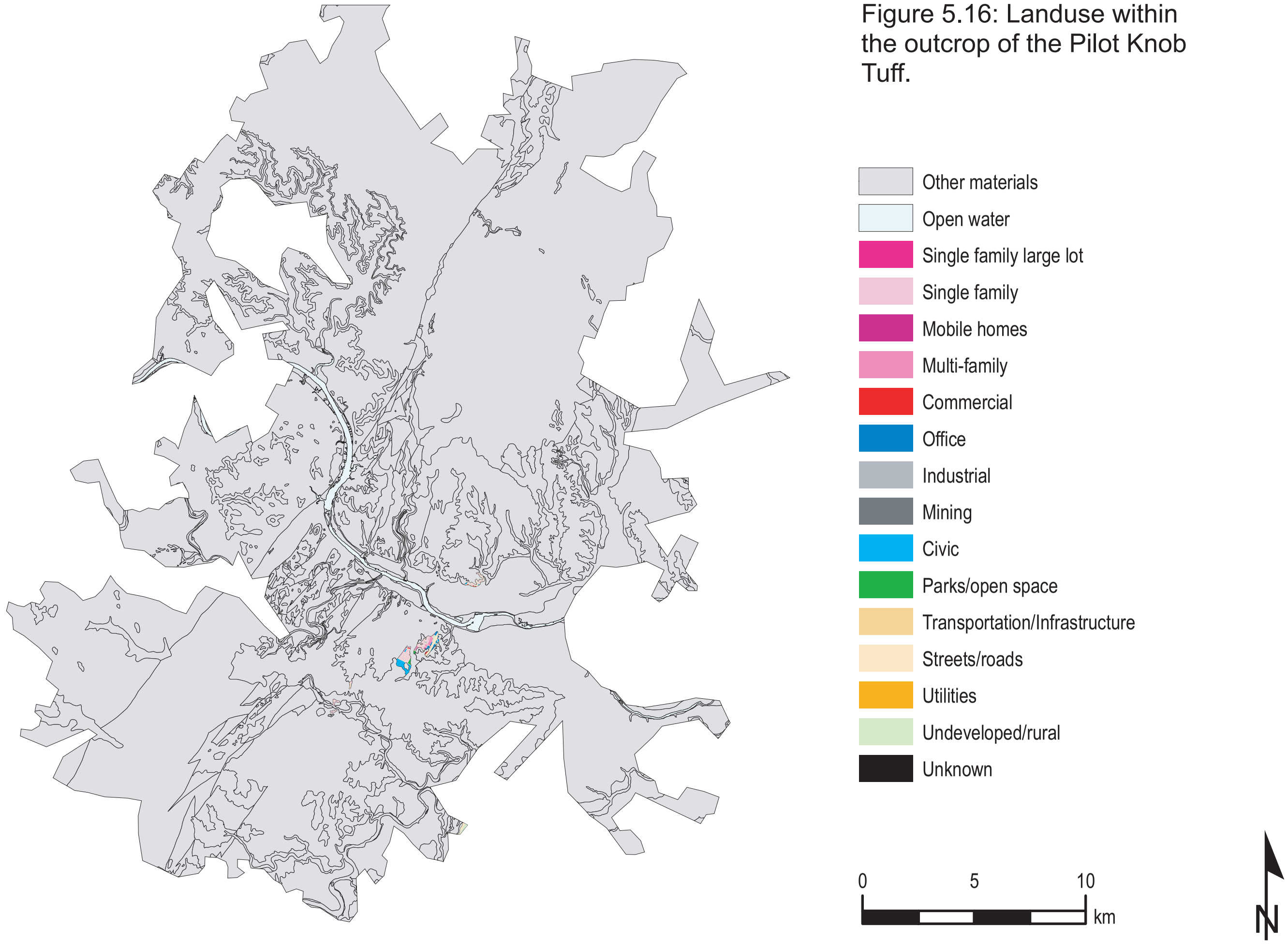


Figure 5.15: Landuse within the outcrops of the Taylor and Navarro Formations.

Figure 5.16: Landuse within the outcrop of the Pilot Knob Tuff.



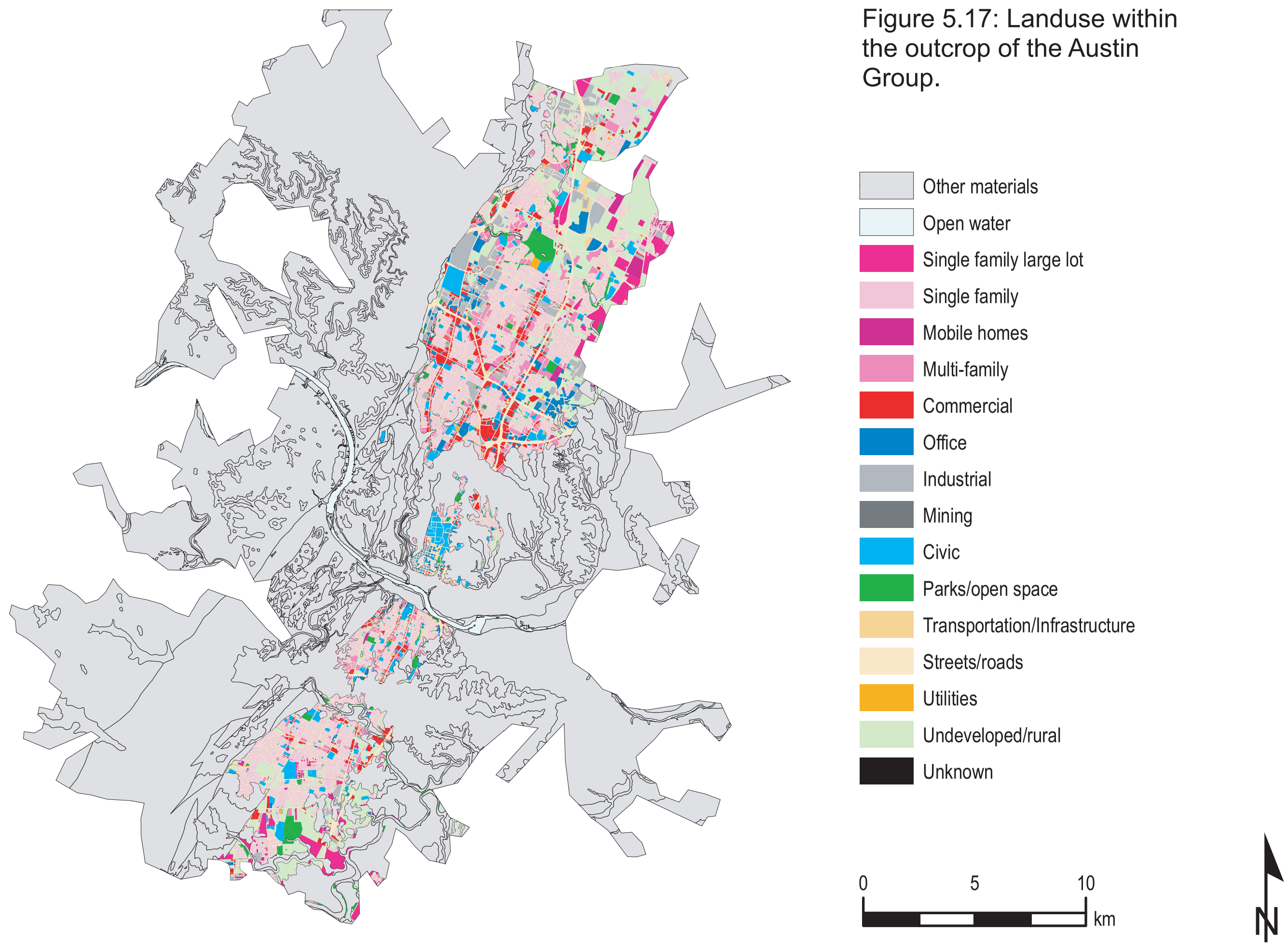


Figure 5.17: Landuse within the outcrop of the Austin Group.

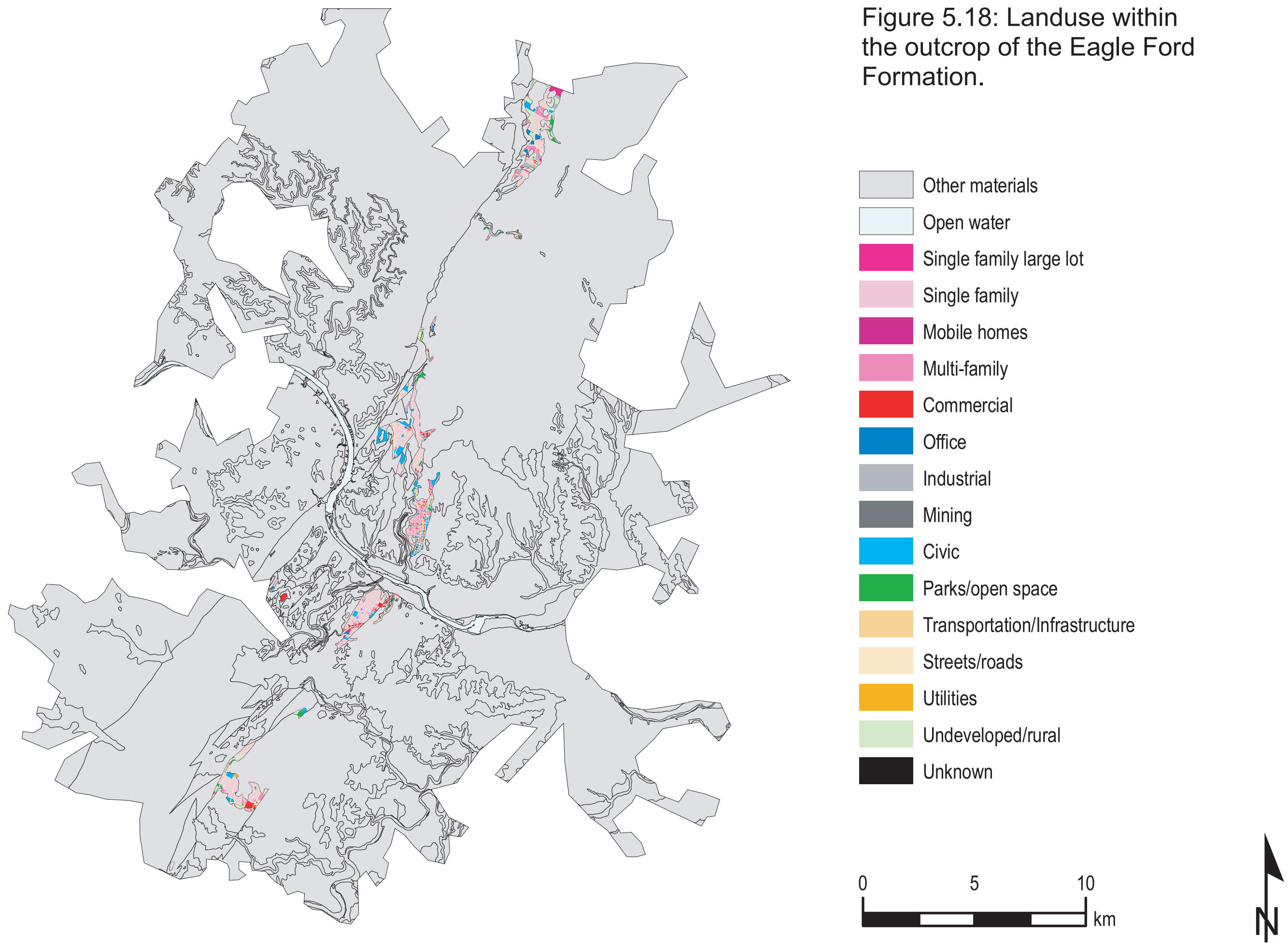


Figure 5.18: Landuse within the outcrop of the Eagle Ford Formation.

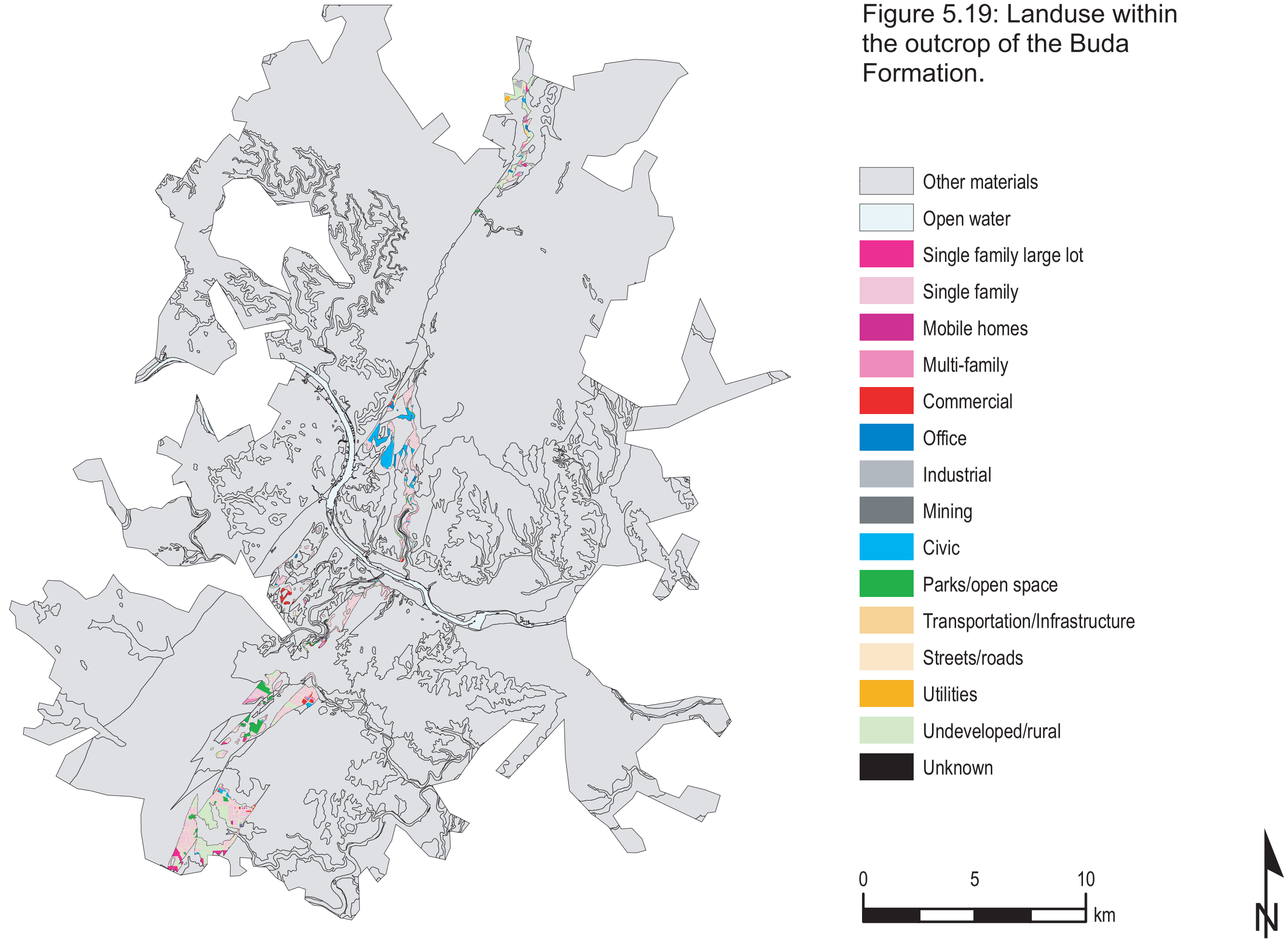


Figure 5.19: Landuse within the outcrop of the Buda Formation.

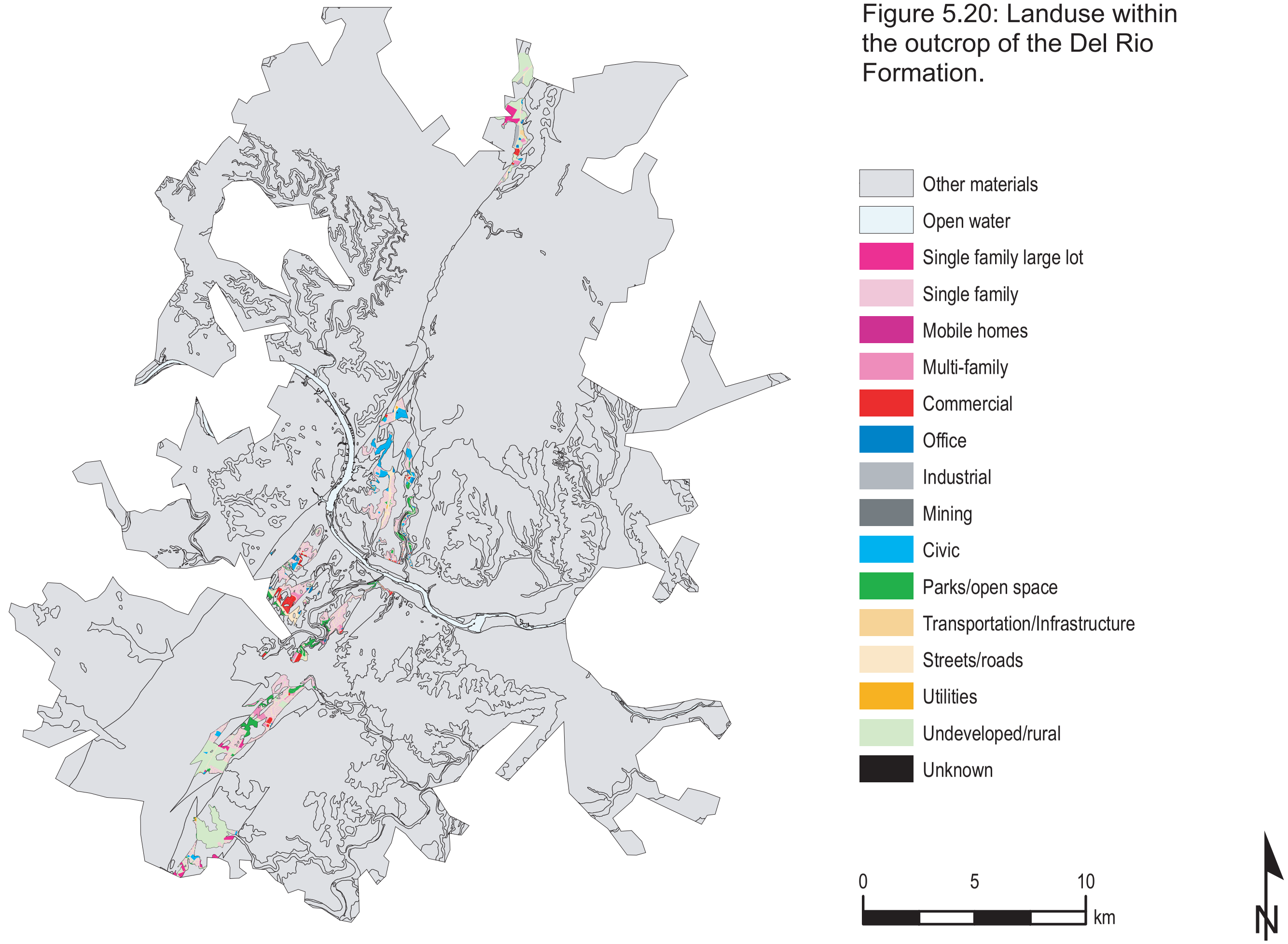


Figure 5.20: Landuse within the outcrop of the Del Rio Formation.

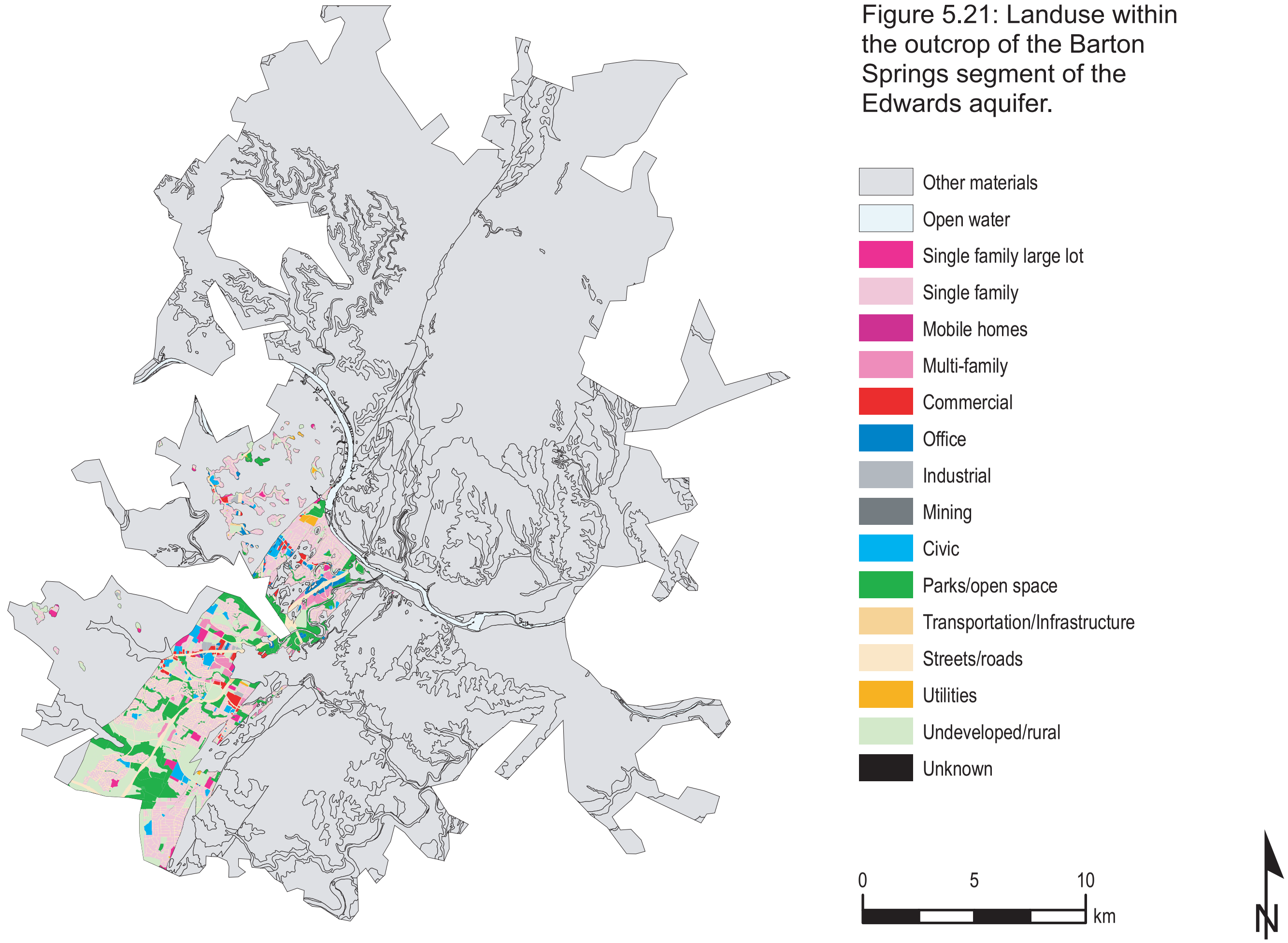


Figure 5.21: Landuse within the outcrop of the Barton Springs segment of the Edwards aquifer.

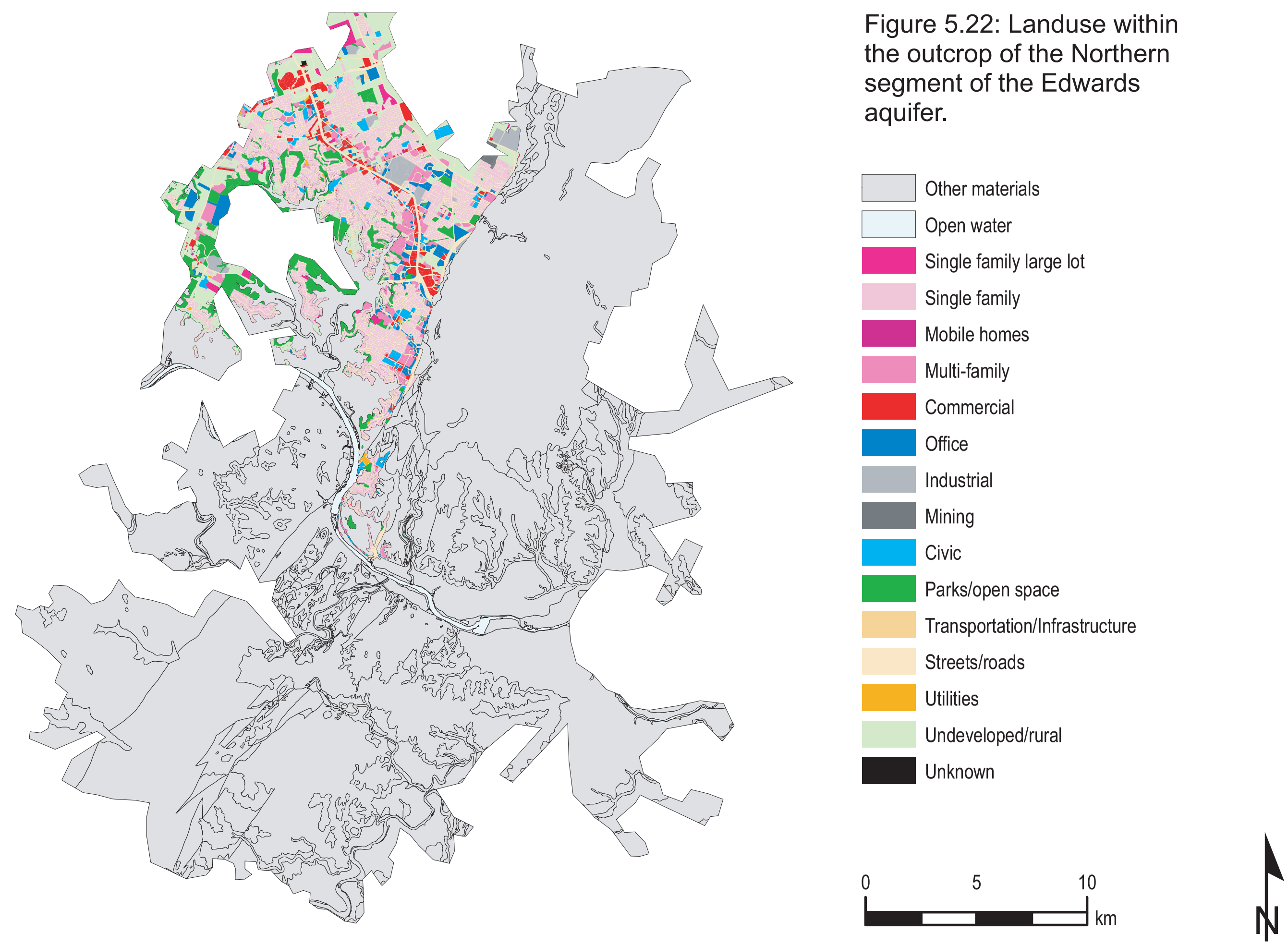


Figure 5.22: Landuse within the outcrop of the Northern segment of the Edwards aquifer.

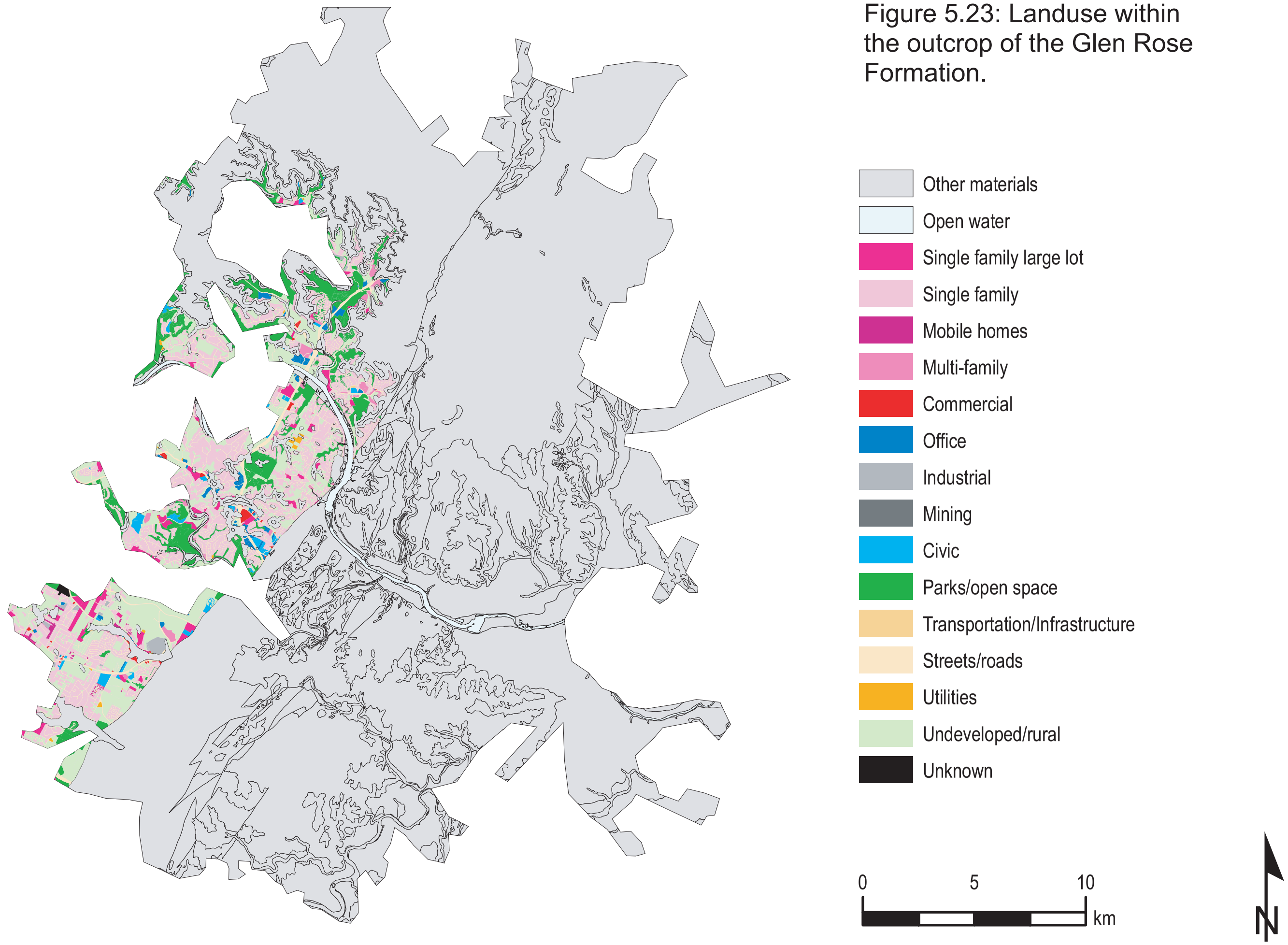


Figure 5.23: Landuse within the outcrop of the Glen Rose Formation.

Table 5.3: Land use and impervious cover over the different geologic formations in Austin, for the year 2000.

LAND USE (2000) Code	Description	Impervious cover % City of Austin[(1)]	Impervious cover % assumed	Quaternary deposits km²	Taylor & Navarro km²	Austin Group km²	Eagle Ford Formation km²	Pilot Knob Tuff km²	Buda Fm km²	Del Rio Formation km²	Edwards Barton Sp km²	Edwards Northern km²	Glen Rose Formation km²	
50	Large lot single family		20	6.4	3.3	6.6	0.2	0.0	0.4	0.6	1.6	1.7	3.8	
100	Single family	32-39	35	32.6	7.6	45.5	5.0	0.4	5.6	5.3	22.2	39.0	36.0	
113	Mobile homes	30	30	2.6	0.9	2.2	0.0	0.0	0.0	0.0	0.0	0.1	0.4	
200	Multi-family	70	70	5.7	1.6	8.9	1.4	0.1	0.5	0.6	2.1	7.2	1.7	
300	Commercial	70	70	6.8	0.9	10.1	0.7	0.0	0.3	0.8	1.4	5.4	0.6	
400	Office	80	80	2.6	0.3	5.9	0.4	0.1	0.3	0.3	1.2	5.3	1.8	
500	Industrial	70	70	10.6	4.4	11.5	0.3	0.0	0.3	0.3	0.5	4.0	0.9	
560	Resource extraction (mining)		5	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	
600	Civic	35	35	9.7	2.0	9.8	1.1	0.2	1.3	1.1	2.4	3.3	2.0	
700	Open space/parks	10	10	18.6	2.6	5.5	0.5	0.0	0.8	1.1	11.5	13.3	16.4	
800	Transportation & infrastructure (Offices, Airports)		80	15.8	0.7	1.2	0.1	0.0	0.1	0.1	0.1	0.4	0.1	
860	Streets & roads		90	24.7	5.8	27.2	2.7	0.3	1.9	2.8	9.8	17.2	9.7	
870	Utility facilities and buildings	70	70	2.1	0.3	0.4	0.0	0.0	0.1	0.0	0.4	0.4	0.4	
900	Undeveloped/rural	3	3	32.2	18.2	30.0	1.0	0.1	3.1	4.5	12.4	21.5	30.6	
940	Open water	3	3	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
999	Unknown		45	0.0	0.0	2.3	0.0	0.0	0.0	0.1	0.0	0.3	0.2	
	Total area (based on land use coverage)			172.7	48.6	164.8	13.3	1.3	14.6	17.7	65.6	119.2	104.6	
	Total area (based on geology coverage)			172.7	48.6	167.1	13.3	1.3	14.7	17.8	65.6	119.5	104.6	
	Total impervious cover (km²)			74.4	16.1	75.6	6.7	0.7	5.4	6.8	23.4	49.3	29.6	287.9
	Total impervious cover (%)			43.1	33.2	45.2	50.6	51.8	37.0	38.0	35.7	41.2	28.3	
	Total pervious cover (km²)			98.3	32.5	91.5	6.6	0.6	9.3	11.0	42.2	70.2	75.0	437.3
	pervious/impervious ratio			1.3							1.8	1.4		

total area 725.2

(1) - Percent Impervious Cover/Landuse assumptions for Phase II CRWR GIS Model, City of Austin, Environmental Resources Management Divison, personal communication.

Table 5.4: Direct recharge under urban conditions (year 2000).

		PERVIOUS AREA km²	PPT mm/a	DIRECT RECHARGE (% of ppt)	DIRECT RECHARGE mm/a	DIRECT RECHARGE l/a	DIRECT RECHARGE mm/a
ALLUVIUM	*(gravel, sand, silt, clay)*						
LCR deposits	*(sand, silt, clay)*						
UCR deposits	*(gravel, sand, silt)*						
TRIBUTARY TERRACE dpsts	*(lmst gravel, sand, mud)*						
HIGH TERRACE dpsts	*(lmst, sand, gravel, caliche)*	98.3	813	9 *(1)*	73.2	7,194,204,528.8	
NAVARRO Gp	*(clay)*						
TAYLOR Fm	*(clay)*	32.5	813	1 *(2)*	8.1	263,879,548.3	
PILOT KNOB TUFF	*(tuff)*	0.6	813	5 *(2)*	40.7	25,477,372.0	
AUSTIN Gp	*(chalk, lmst to marl)*	91.5	813	1 *(3)*	8.1	744,078,142.1	
EAGLE FORD Fm	*(shale)*	6.6	813	0 *(2)*	0.0	0.0	
BUDA Fm	*(hard lmst to marl)*	9.3	813	1 *(2)*	8.1	75,292,514.6	
DEL RIO Fm	*(shale with high clay content)*	11.0	813	0 *(2)*	0.0	0.0	
GEORGETOWN Fm	*(lmst to marl)*						
EDWARDS Fm	*(dol, lmst, hard lmst, collapsed lmst)*						
COMANCHE PEAK Fm	*(lmst)*						
KAINER/WALNUT Fm	*(hard lmst to marl)*	185.1					
	Barton Springs segment	42.2	813	8.2 *(4)*	67.1	2,829,474,073.7	
	Barton Springs segment	42.2	813	7.0 *(5)*	56.8	2,397,072,394.8	
	Northern segment	70.2	813	20 *(6)*	162.6	11,422,587,528.0	
GLEN ROSE Fm	*(marl, dolst, lmst)*	75.0	813	1 *(2)*	8.1	609,979,497.8	
OPEN WATER		5.6			0.0		
	total pervious area	442.9					
	total city area	730.9					
TOTAL	*minus open water*	725.3	813			22,732,571,526.3	31.3

(1) Based on Wickham (1991)
(2) Assumed
(3) Mace (1998)
(4) R = infiltration of ppt + stream loss; based on Slade et al. (1985), Sharp (1990), and Scanlon et al. (2001)
(5) R = infiltration of ppt; based on Slade et al. (1985), Sharp (1990), and Scanlon et al. (2001)
(6) Jones (2003)

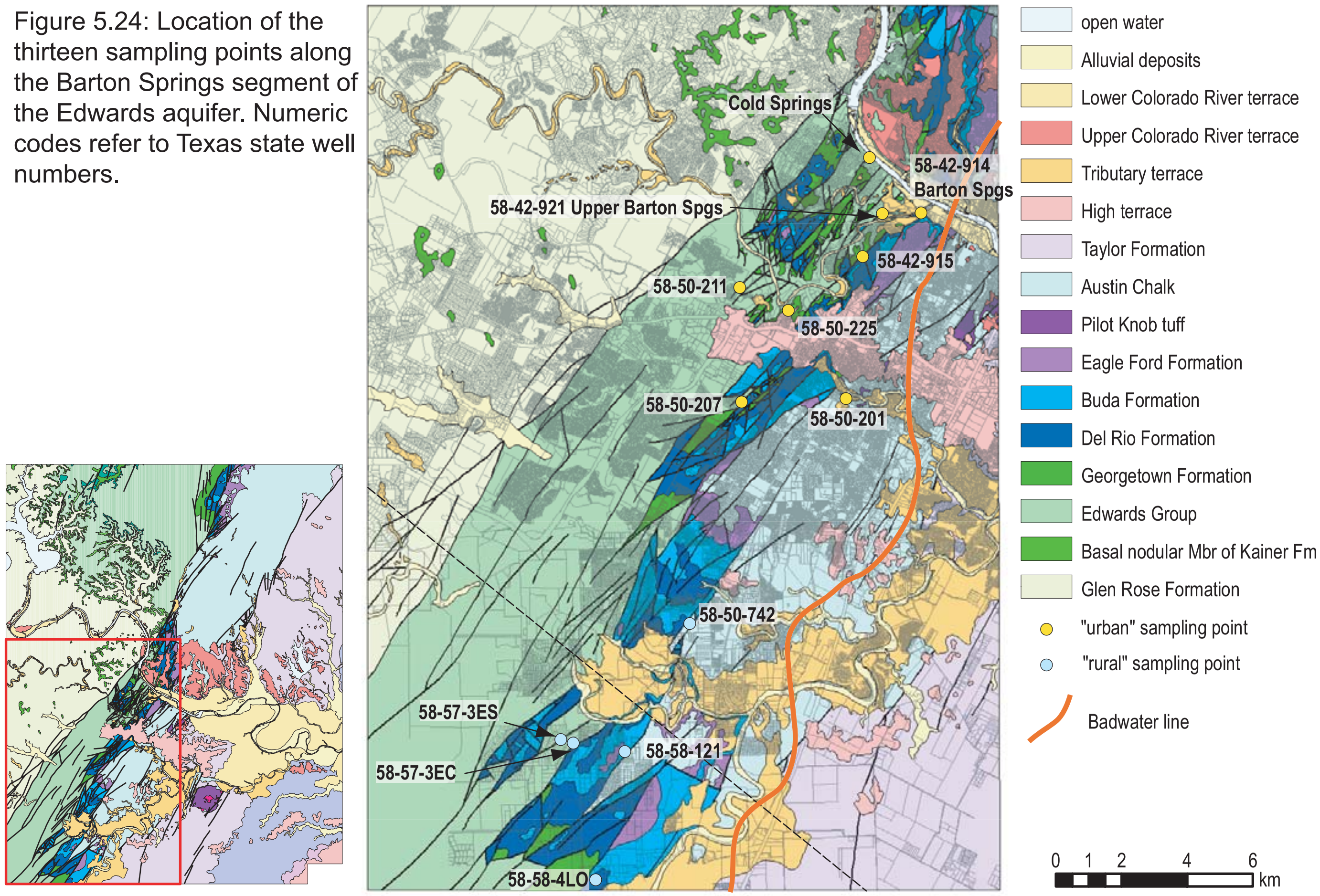


Figure 5.24: Location of the thirteen sampling points along the Barton Springs segment of the Edwards aquifer. Numeric codes refer to Texas state well numbers.

Table 5.5: Results of the chemical analyses of samples of groundwaters from the Barton Springs Segment of the Edwards aquifer, first sampling campaign.

		58-57-3ES	58-58-121	58-50-225	58-50-742	58-42-914 Barton Spgs	58-57-3EC	58-58-4LO	58-50-201	Cold Springs	58-42-921 Upper Barton Spgs	58-42-915	58-50-211	58-50-207
		rural	rural	urban	rural	urban	rural	rural	urban	urban	urban	urban	urban	urban
$^{87}Sr/^{86}Sr$		0.707604	0.707755	0.707915	0.707930	0.707964	0.707967	0.707980	0.707984	0.708020	0.708149	0.708198	0.708245	0.708941
Sr	ppb	2093	1875	213.9	519.0	927.5	250.0	222.6	4186	259.3	379.8	2089	179.8	116.0
Sr/Ca	$x10^3$	23.65	29.97	2.92	5.53	9.77	2.78	3.13	39.60	2.80	3.78	11.55	2.19	1.20
Mg/Ca	molar	0.395	0.554	0.439	0.392	0.341	0.364	0.388	0.323	0.399	0.377	0.515	0.518	0.454
Na	ppm	7.041	6.147	13.65	8.666	14.19	7.084	8.304	21.49	18.33	11.03	28.94	9.686	5.650
Mg	ppm	21.22	21.01	19.51	22.31	19.62	19.85	16.72	20.68	22.43	22.98	56.46	25.83	26.64
Al	ppb	45.88	BDL	3.481	BDL	10.40	BDL	8.854	2.184	2.33	13.90	126.2	63.03	13.28
Si	ppm	5.212	4.555	4.503	5.490	5.058	5.320	3.941	5.462	4.524	5.776	6.033	5.520	6.710
P	ppb	2.018	6.664	10.83	2.796	3.246	3.214	4.716	4.640	5.629	12.40	5.879	11.27	5.189
K	ppm	1.286	1.331	1.590	1.248	1.356	1.122	1.322	6.543	1.450	1.425	3.839	1.050	1.612
Ca	ppm	88.49	62.57	73.31	93.93	94.90	90.04	71.02	105.7	92.70	100.4	180.8	82.27	96.78
Fe	ppb	53.12	51.66	72.56	89.38	52.72	50.58	53.96	111.4	57.87	56.85	441.1	84.94	54.10
Mn	ppb	0.197	BDL	0.365	0.774	1.973	BDL	0.978	2.706	0.352	1.298	9.055	2.037	0.412
Rb	ppb	1.427	1.141	0.839	1.115	1.244	0.960	0.830	2.014	1.010	1.225	7.703	0.752	1.091
Cd	ppb	BDL	0.038	0.023	BDL	BDL	BDL	BDL	0.232	0.038	BDL	0.581	BDL	BDL
Ba	ppb	33.31	30.15	36.51	36.04	43.65	31.89	26.01	94.02	52.43	109.6	33.38	93.33	117.7
Pb	ppb	0.557	21.37	0.768	0.348	BDL	0.212	0.231	1.185	0.197	0.025	9.235	2.405	0.326
Th	ppb	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	0.405	BDL	0.011	BDL	BDL
U	ppb	0.418	0.492	0.458	0.495	0.464	0.518	0.408	0.441	0.693	0.536	0.647	0.399	0.343
alkalinity	meq/l	1.086	0.944	0.855	1.147	1.054	1.101	0.821	1.115	0.967	1.158	1.104	1.070	1.355
$\delta^{18}O$	$^o/_{oo}$	-4.10	-4.01	-3.60	-4.11	failed	-4.11	-3.85	-3.75	-3.70	-4.08	-3.93	-4.11	-4.75
$\delta^{18}O$ replicates	$^o/_{oo}$	-4.06	-3.99	-3.74	-5.22	-3.94	-4.06	-3.79	-3.59	-3.62	-3.90	-3.96	-4.07	-4.74
avg $\delta^{18}O$	$^o/_{oo}$	-4.08	-4.00	-3.67	error?	-3.94	-4.09	-3.82	-3.67	-3.66	-3.99	-3.94	-4.09	-4.74
δD	$^o/_{oo}$	-21.1	-22.7	-23.8	-23.8	-22.5	-21.4	-19.2	-17.7	-13.8	failed	failed	-23.9	failed

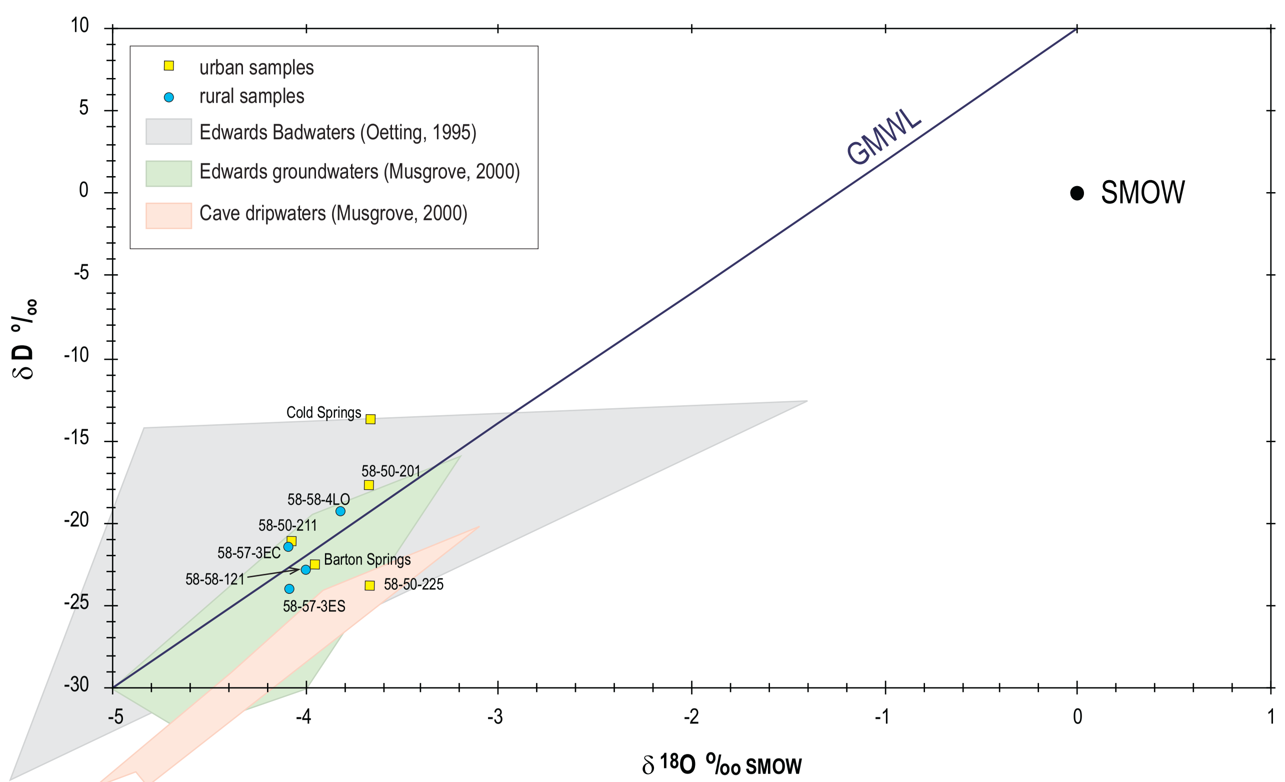


Figure 5.25: Oxygen and hydrogen isotopic composition of the samples compared to the Global Meteoric Water Line (GMWL) and the Standard Mean Ocean Water (SMOW) isotopic composition. Colored areas represent values from previous studies in central Texas. A badwater sample from Oetting's (1995) study, which lays close to SMOW has been omitted.

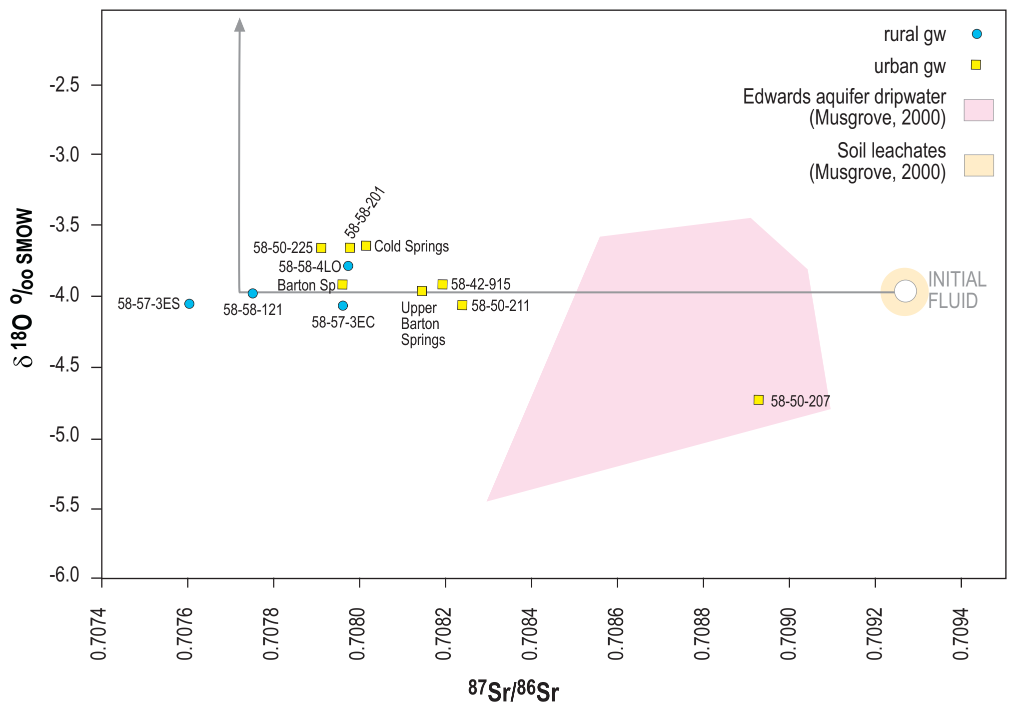


Figure 5.26: $^{87}Sr/^{86}Sr$ versus $\delta^{18}O$. The arrow illustrates the evolution of both parameters in a fluid as water-limestone interaction increases, from an initial fluid composition based on soil leachates (Musgrove, 2000; Banner et al., 1989). Colored areas represent values from previous studies in central Texas, showing a progressive geochemical evolution from soil leachates to cave dripwaters to Edwards goundwater.

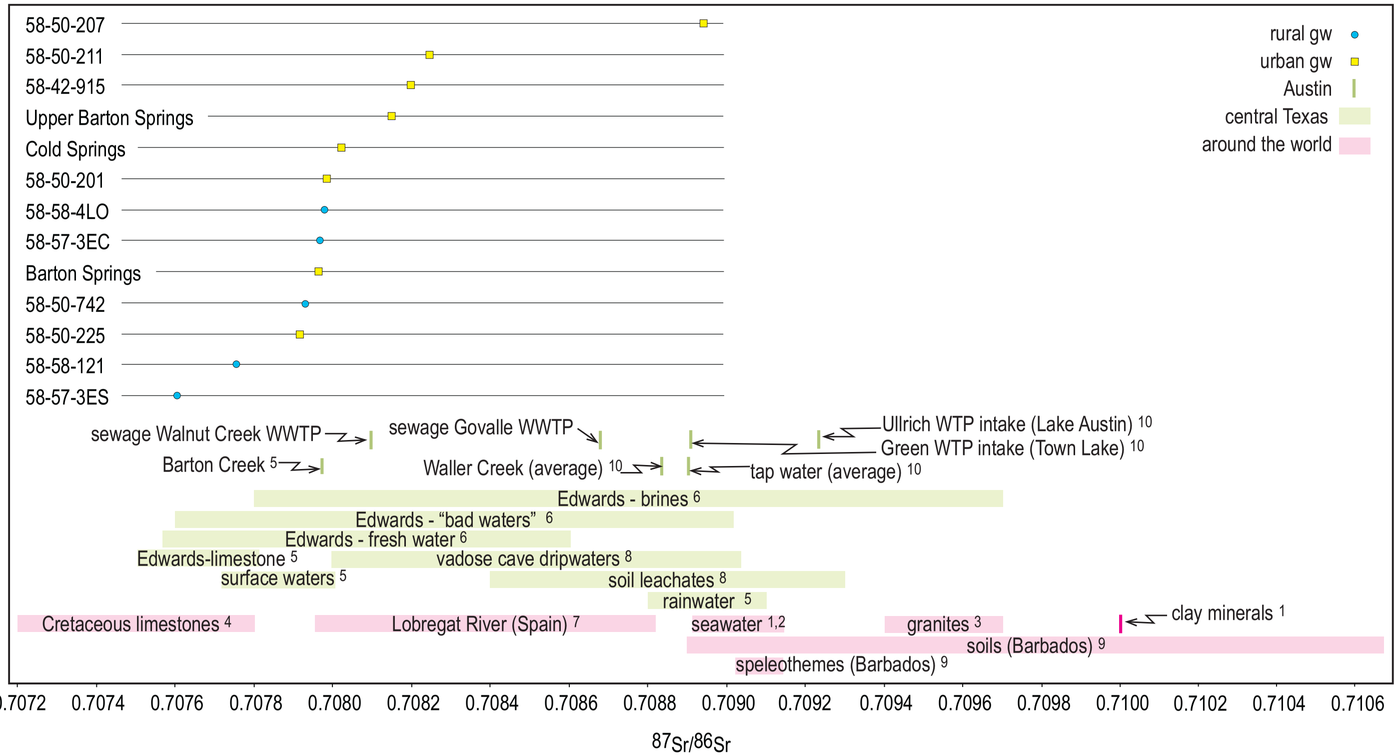


Figure 5.27: $^{87}Sr/^{86}Sr$ values in groundwater from the Barton Springs Segment of the Edwards Aquifer, compared to works by 1) Banner, 2004; 2) Capo and de Paolo, 1990; 3) Neumann and Dreiss, 1995; 4) Burke et al., 1982; 5) Oetting 1995; 6) Oetting et al., 1996; 7) Soler et al., 2002; 8) Musgrove, 2000; 9) Banner et al., 1996; and 10) Christian, in preparation.

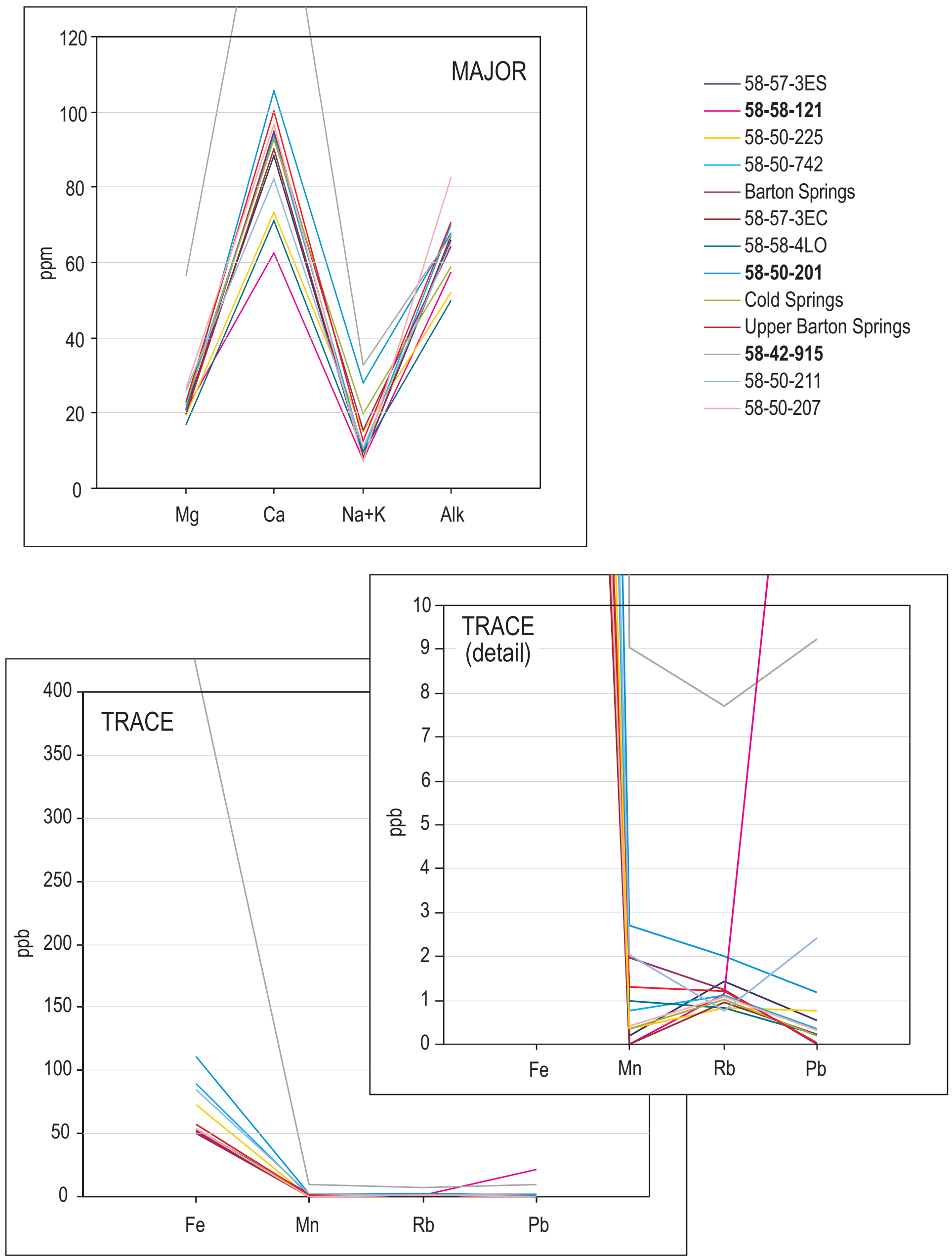


Figure 5.28: Schoeller diagrams for some major and trace constituents of groundwaters from the Barton Springs segment of the Edwards aquifer (no charge balance has been calculated).

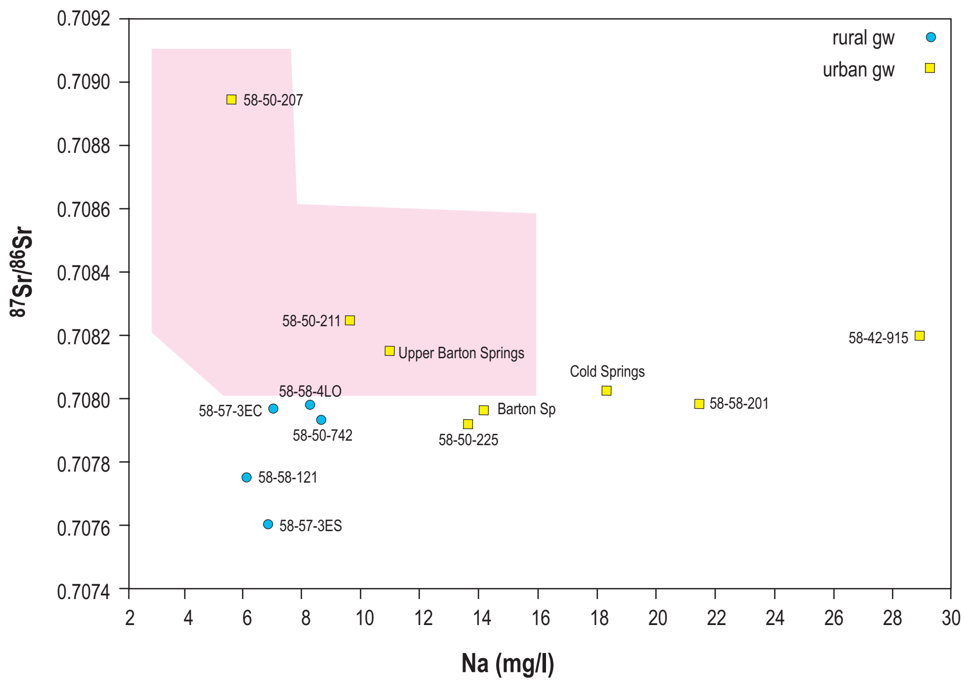


Figure 5.29: Sodium versus $^{87}Sr/^{86}Sr$. The shaded area represents values from dripwaters in Natural Bridge Caverns and Inner Space Cavern (both in the Edwards aquifer) reported by Musgrove (2000).

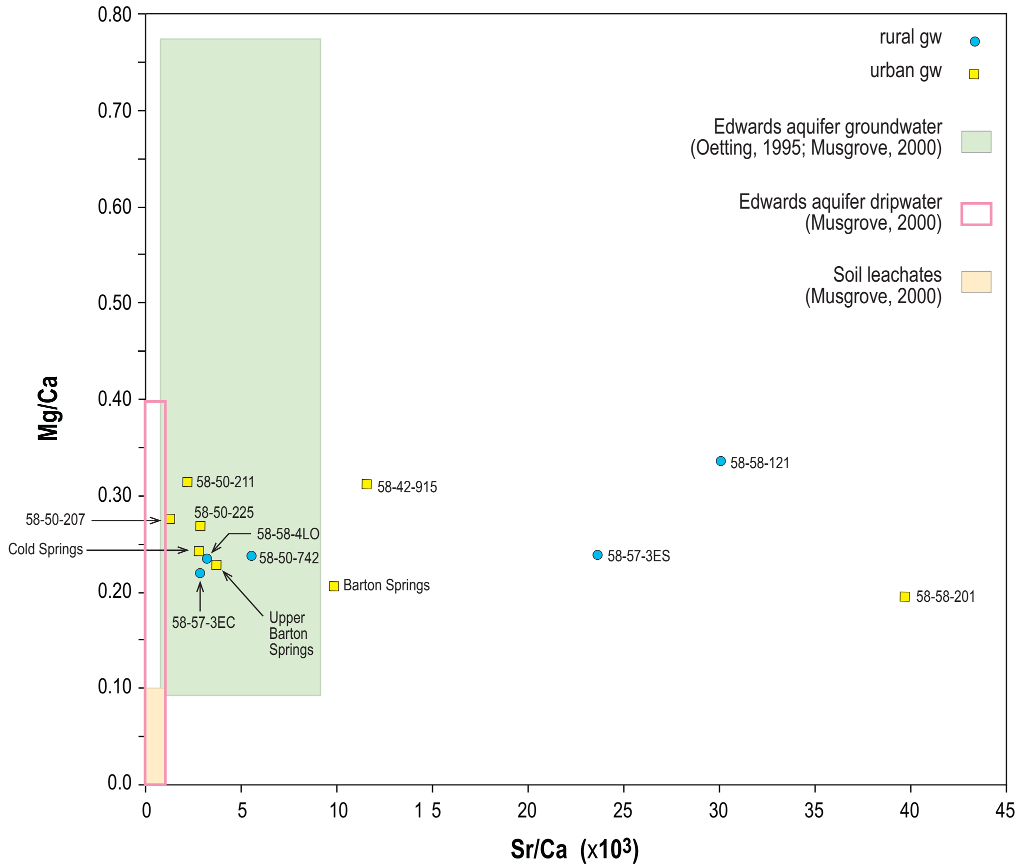


Figure 5.30: Sr/Ca versus Mg/Ca of groundwater from the Barton Springs segment of the Edwards aquifer. Colored areas represent values from previous studies in central Texas.

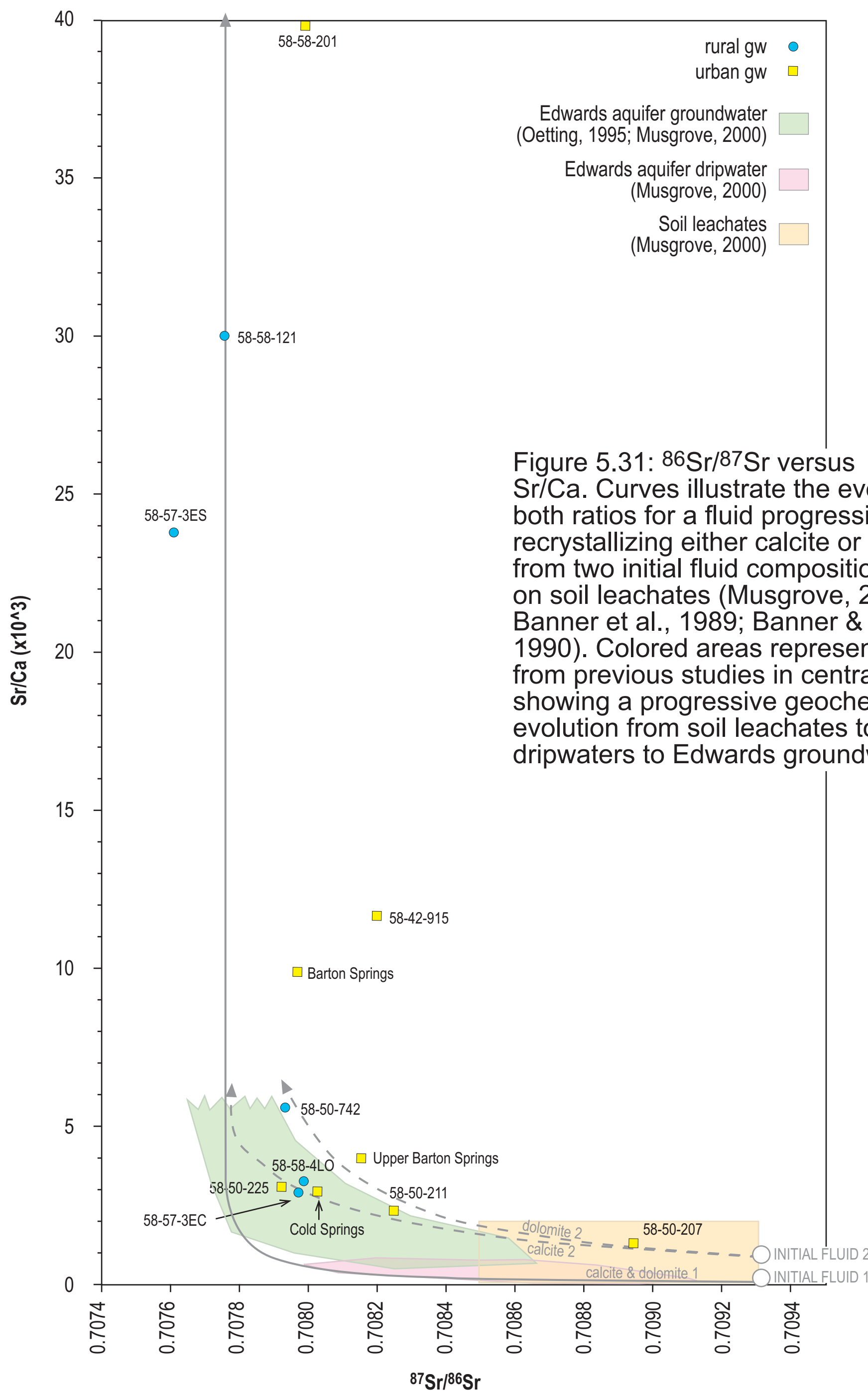


Figure 5.31: $^{86}Sr/^{87}Sr$ versus Sr/Ca. Curves illustrate the evolution of both ratios for a fluid progressively recrystallizing either calcite or dolomite from two initial fluid compositions based on soil leachates (Musgrove, 2000; Banner et al., 1989; Banner & Hanson, 1990). Colored areas represent values from previous studies in central Texas, showing a progressive geochemical evolution from soil leachates to cave dripwaters to Edwards groundwater.

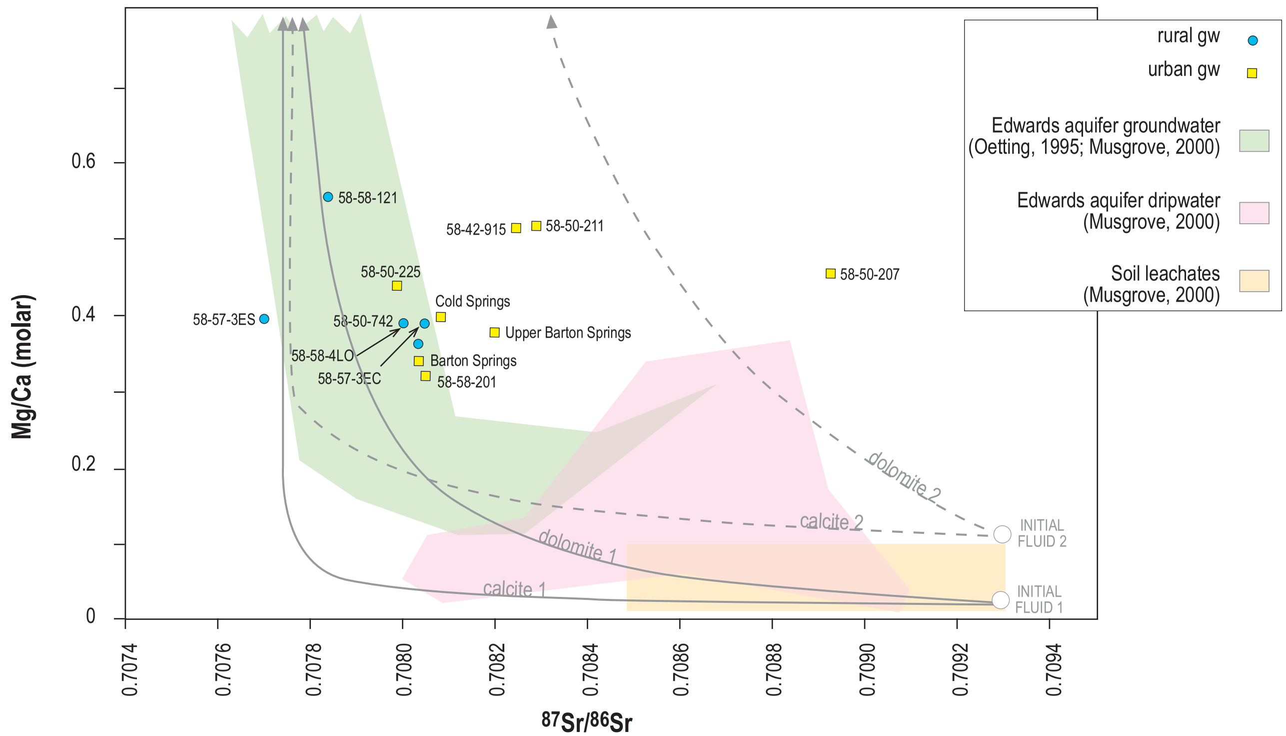


Figure 5.32: $^{87}Sr/^{86}Sr$ versus Mg/Ca. Curves illustrate the evolution of both ratios for a fluid progressively recrystallizing either calcite or dolomite from two initial fluid compositions based on soil leachates (Musgrove, 2000; Banner et al., 1989; Banner & Hanson, 1990). Colored areas represent values from previous studies in central Texas, showing a progressive geochemical evolution from soil leachates to cave dripwaters to Edwards goundwater.

Table 5.6: Water Balance and water statistics for Austin, year 2000.

					mm/a		
	Population		656,562				(1)
	Area		704	km2			(2)
	Population density		933	p/km2			
P	Mean annual precipitation				813		(3)
DRP	Direct recharge (preurban)				53		
DRU	Direct recharge (urban)				31		
W	Served water	population served	738,229				(4)
		area served	710	km2			(4)
		average	541,000	m3/d	278		(2)
		peak	856,000	m3/d	440		(2)
		max. capacity	984,000	m3/d	506		(2)
WW	Treated wastewater	population served	685,783				(4)
		area served	601	km2			(4)
		average	318,000	m3/d	193		(2)
		max. capacity	492,000	m3/d	299		(2)
EUW	Excess urban water	avg W - max WW			-21		
		avg W - avg WW			85		
		max W - max WW			207		
	Gross unbilled water	12%	64,920	m3/d	33		(4,5)
WL	Mains leakage rate	7.7%	41,657	m3/d	21	(similar to Thornton, 2002)	
WWL	Sewer leakage rate	5%	16,737	m3/d	10		
I	Irrigation	not area weighted			54	avg	
					175	max	
		area weighted by 725/437			90	avg	
					291	max	
PWR	Plant water requirement	not area weighted			81	low	(6)
					364	intermediate	
					910	high	
		area weighted by 437/725			49	low	
					219	intermediate	
					548	high	
IPWR		from irrigation only			5	low	
					22	intermediate	
					54	high	
IR	Irrigation return flow				49	low	
					32	intermediate	
					-1	high	
R	Total recharge	ET not accounted			116		
		after substracting PWR			111	low	
					94	intermediate	
					62	high	

(1) US Census Bureau, online
(2) City of Austin, online
(3) NOOA, online
(4) Dan Pedersen, CoA W&WW, personal communication
(5) Austin American Statesman, 1998
(6) Texas Evapotranspiration Network, online

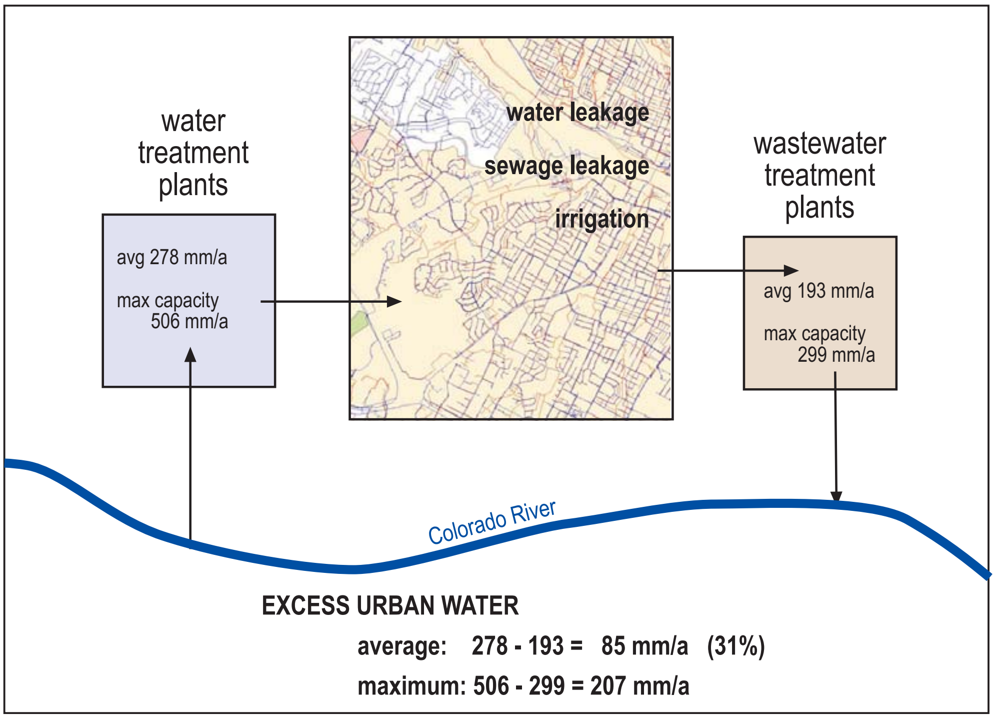


Figure 5.33: Excess urban water determined as the difference between the water treated at the water treatment plants, and the sewage treated at the wastewater treatment plants on the year 2000.

avg indicates the average amount of water treated and distributed and *max capacity* the maximum treatment capacity. The use of the average or maximum treatment capacity rates results on average and a maximum excess urban water values.

Based on data from the City of Austin Water and Wastewater Utility (online).

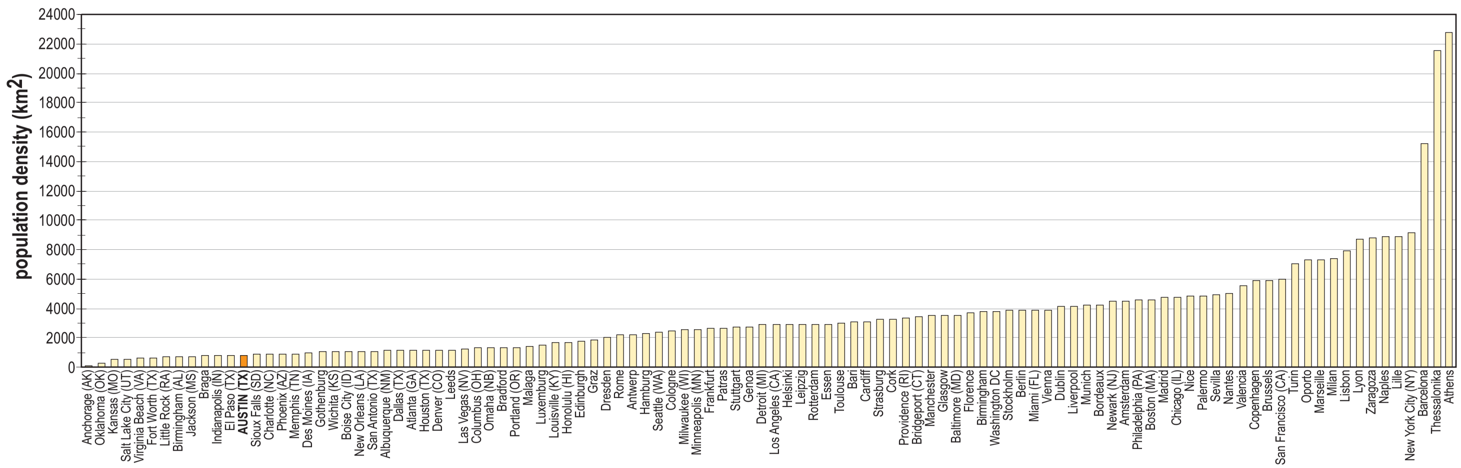


Figure 5.34: Population density in 57 European cities (European commision Urban Audit, online) and highly populated cities in most states of the USA (Wendell Cox Consultancy, online).

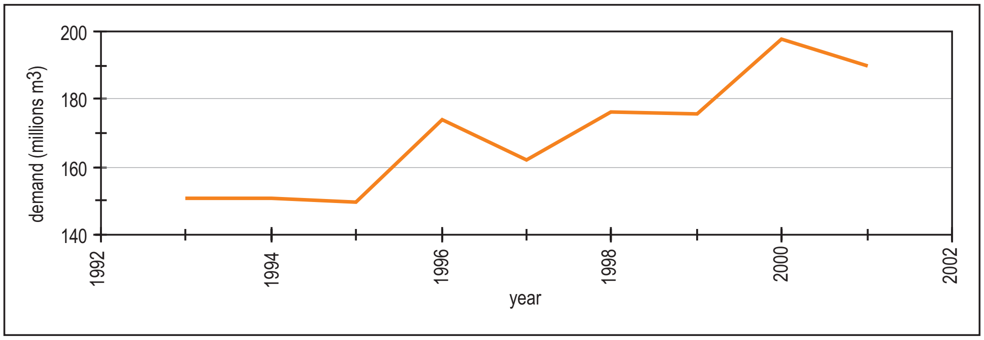


Figure 5.35: Historic evolution of the water demand in the City of Austin (City of Austin Water and Wastewater Utility, online).

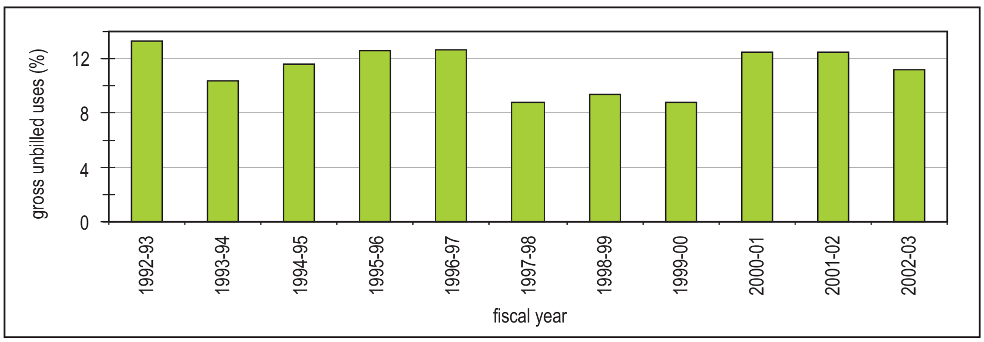


Figure 5.36: Historic evolution of the gross unbilled treated water in the City of Austin (Dan Pedersen, City of Austin Water and Wastewater Utility, personal communication).

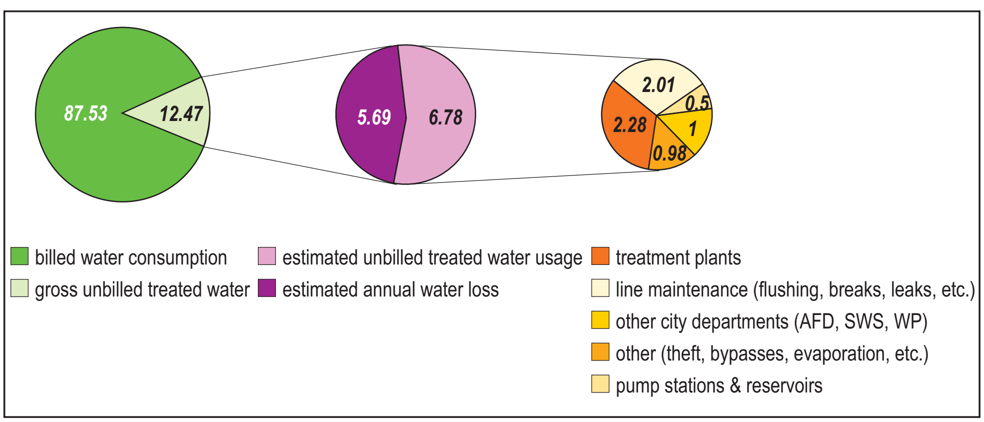


Figure 5.37: Water uses in the City of Austin for the fiscal year 2001-02, including unbilled uses and water losses (Dan Pedersen, personal communication).

Table 5.7: Evapotranspiration and plant water requirement for Austin, year 2003.

	E (1)	ETo (2)	PLANT WATER REQUIREMENT (PWR)			
			0.9	0.6	0.2	Kc
			1	0.6	0.4	AS
	mm/mo	mm/mo	mm/mo	mm/mo	mm/mo	
Jan	63.5	50.0	45.0	18.0	4.0	
Feb	66.0	33.3	29.9	12.0	2.7	
Mar	91.4	70.4	63.3	25.3	5.6	
Apr	108.0	103.1	92.8	37.1	8.2	
May	133.4	115.3	103.8	41.5	9.2	
Jun	177.8	130.3	117.3	46.9	10.4	
Jul	222.3	133.9	120.5	48.2	10.7	
Aug	223.5	149.6	134.6	53.9	12.0	
Sep	177.8	97.3	87.6	35.0	7.8	
Oct	146.1	78.0	70.2	28.1	6.2	
Nov	101.6	49.8	44.8	17.9	4.0	
Dec	73.7	51.8	46.6	18.7	4.1	
	mm/a	mm/a	mm/a	mm/a	mm/a	
	1511.3	1010.9	909.8	363.9	80.9	
			HIGH	INTMD	LOW	

TURFF OR CROP COEFFICIENT (2)		
	Tc or Kc	range
trees, groundcover	0.5	0.2 - 0.9
shrub, perennials	0.5	0.2 - 0.7
cool season turfgrass, annuals	0.8	0.6 - 0.8
warm season turfgrass	0.6	0.3 - 0.6

ALLOWABLE STRESS (2)	
	AS
No Stress	1
Low Stress	0.8
Normal Stress	0.6
High Stress	0.5
Very High Stress	0.4

(1) Average monthly gross lake evaporation (NCDC, online)

(2) Monthly reference evapotranspiration (TexasET, online)

Table 5.8: Analytical results from the second sampling campaign.

			Eliza Springs	Eliza Springs REPEAT	Barton Springs - Pool	Upper Barton Spring	Old Mill Springs	Green Intake - Town Lake	Green Intake - Town Lake DUPLICATE	Govalle WWTP - raw influent sewage	Walnut Creek WWTP - raw influent sewage	Walnut Creek WWTP - before chlorination	Walnut Creek WWTP - during chlorination	Walnut Creek WWTP - final effluent sewage	Ullrich Intake - Lake Austin
SAMPLING		DATE	8/28/2002		8/28/2002	8/28/2002	8/28/2002	9/1/2002		9/6/2002	9/6/2002	9/6/2002	9/6/2002	9/6/2002	9/6/2002
		nº samples	whole suite		δ^{15}N & THM	whole suite	whole suite	whole suite		whole suite	whole suite	♦	♦	♦	δ^{15}N & THM ♦
FIELD PARAMETERS	T	ºC	22.32					n/a		30.9	29.1				
	pH		6.66			Hauwert & Pope (CoA)		n/a		7.52	7.3				
	conductivity	µS/cm	631					n/a		735	751				
	turbidity	TU	2.8					n/a							
ALKALINITY		DATE	8/30/2002		8/30/2002	8/30/2002	8/30/2002	9/2/2002		9/6/2002	9/6/2002	9/6/2002	9/6/2002	9/6/2002	9/6/2002
	0.1N HCl	mℓ	1.29		1.23	1.32	1.23	0.66		0.99	0.84	0.25	0.23	0.18	0.59
	from 25 ml	mg/ℓ	314.76		299.88	321.84	301.10	160.31		241.32	203.74	61.49	56.61	42.70	143.47
	0.1N HCl	mℓ											0.08		
	from 10 ml	mg/ℓ											49.41		
ANIONS		DATE	2/1/2003	2/1/2003		2/1/2003	2/1/2003	2/1/2003		2/1/2003	2/1/2003				
	Cl	mg/ℓ	24.00	23.99		19.91	46.14	19.26		66.94	67.83				17.73
	NO_2	mg/ℓ	0.03	0.03		0.00	0.00	0.00		0.00	0.00				< 0.005
	Br	mg/ℓ	0.11	0.13		0.00	0.28	0.00		0.23	0.64				0.08
	NO_3	mg/ℓ	4.97	4.93		9.77	5.37	0.80		0.35	1.46				0.19
	PO_4	mg/ℓ	0.00	0.00		0.00	0.00	0.00		10.60	13.17				< 0.020
	SO_4	mg/ℓ	27.92	27.13		28.73	42.51	16.36		19.85	15.11				12.97
		DATE	2/3/2003	2/3/2003		2/3/2003	2/3/2003	2/3/2003	2/3/2003	2/3/2003	2/3/2003				
MAJOR CATIONS	Na	ppm	13.30	13.44		10.71	25.31	12.53	12.00	57.22	66.63				11.67
	Mg	ppm	21.34	21.97		25.77	23.16	13.68	13.46	17.73	12.88				12.82
	Si	ppm	5.08	5.65		5.86	4.90	4.78	5.15	6.97	5.97				4.72
	K	ppm	1.23	1.28		1.21	1.57	3.13	3.24	13.30	13.24				3.23
	Ca	ppm	86.59	91.30		100.10	84.50	45.65	46.07	27.06	28.14				39.38
MINOR CATIONS	Li	ppb	10.71	9.13		8.85	23.09	5.58	5.39	7.13	7.45				4.18
	B	ppb	61.37	59.32		45.90	90.35	52.97	64.23	201.86	240.07				N/A
	Al	ppb	< 6.75	< 6.75		< 6.75	< 6.75	7.68	10.43	12.77	15.15				9.59
	P	ppb	< 480	< 480		< 480	< 480	< 480	< 480	2848	5628				N/A
	V	ppb	4.23	3.80		5.29	4.44	3.95	4.13	3.97	4.34				2.94
	Cr	ppb	1.46	0.82		1.75	1.67	< 0.15	< 0.15	0.42	0.37				< 0.15
	Mn	ppb	5.05	5.41		0.60	0.83	3.11	3.08	96.31	55.96				7.42
	Fe	ppb	5.90	6.01		1.67	2.50	8.47	9.60	171.98	146.32				73.17
	Co	ppb	3.66	3.89		4.10	3.84	2.03	2.81	1.67	1.67				< 0.45
	Ni	ppb	1.48	1.80		1.86	1.71	1.51	2.65	5.13	5.14				< 1.35
	Cu	ppb	< 1.5	< 1.5		< 1.5	< 1.5	< 1.5	< 1.5	1.59	6.45				2.49
	Zn	ppb	< 3.6	< 3.6		< 3.6	< 3.6	< 3.6	< 3.6	18.08	22.93				5.39
	As	ppb	< 0.75	< 0.75		< 0.75	< 0.75	1.81	2.30	2.81	2.10				1.73
	Se	ppb	< 1.95	< 1.95		< 1.95	< 1.95	< 1.95	< 1.95	< 1.95	2.05				< 1.95
	Rb	ppb	1.01	0.99		0.82	1.33	1.11	1.21	8.39	9.33				1.12
	Sr	ppb	773	750		441	846	262	271	151	135				227
	Mo	ppb	0.79	0.68		0.45	0.68	0.87	0.94	29.93	14.74				0.87
	Cs	ppb	0.08	0.08		< 0.06	0.14	< 0.06	< 0.06	0.14	0.11				N/A
	Ba	ppb	50.16	49.98		128.12	57.50	46.97	47.25	7.75	2.94				45.03
	La	ppb	< 0.06	< 0.06		< 0.06	< 0.06	< 0.06	< 0.06	< 0.06	< 0.06				N/A
	Pb	ppb	0.13	< 0.12		0.12	0.17	< 0.12	0.20	0.22	0.48				0.12
	Bi	ppb	0.08	0.07		0.08	0.08	0.07	0.07	0.09	0.13				N/A
	Th	ppb	0.74	0.65		0.70	0.69	0.68	0.65	0.66	0.67				N/A
	U	ppb	N/A	N/A		N/A	N/A	N/A	N/A	N/A	N/A				0.48
	Cd	ppb	0.13	0.11		0.11	0.11	0.10	0.11	0.18	0.15				N/A
	Ag	ppb	< 0.15	< 0.15		< 0.15	< 0.15	< 0.15	< 0.15	< 0.15	< 0.15				N/A
	Hg	ppb	3.38	3.11		3.24	3.02	2.49	2.44	2.12	2.37				N/A
	Sn	ppb	< 0.24	< 0.24		< 0.24	< 0.24	< 0.24	< 0.24	< 0.24	0.30				N/A
	Sb	ppb	0.12	< 0.12		< 0.12	0.12	0.19	0.19	0.40	0.41				N/A
STRONTIUM ♦♦		DATE			7/29/2004					4/17/2004	4/17/2004				2/1/2004
	$^{87}Sr/^{86}Sr$	ratio			0.70796					0.70867	0.70810				0.70924
TRIHALO-METHANES ♣		DATE	9/19/2002		9/19/2002	9/19/2002	9/19/2002	9/19/2002		9/19/2002	9/19/2002				9/19/2002
	TCM	µg/ℓ	< 1.00		< 1.00	< 1.00	< 1.00	< 1.00		5.60	4.89				< 1.00
	BDCM	µg/ℓ	< 1.00		< 1.00	< 1.00	< 1.00	< 1.00		< 1.00	< 1.00				< 1.00
	DBCM	µg/ℓ	< 1.00		< 1.00	< 1.00	< 1.00	< 1.00		< 1.00	< 1.00				< 1.00
	TBM	µg/ℓ	< 1.00		< 1.00	< 1.00	< 1.00	< 1.00		< 1.00	< 1.00				< 1.00
	TOTAL THM	µg/ℓ	< 1.00		< 1.00	< 1.00	< 1.00	< 1.00		5.60	4.89				< 1.00
δ^{15}N (air) ♥		DATE													
	in NH_3	$^{o}/_{oo}$	-		-	-	-			-	14.00				-
	in NO_3^- ♠	$^{o}/_{oo}$	3.40		ins	2.40	2.00			7.50	7.90				-1.60

			Ullrich WTP - City water (treated)	Ullrich WTP - City water (treated) REPEAT	Green Intake - Town Lake (2)	Green WTP - City Water (treated)	Davis WTP - City Water (treated)	Little Shoal @ 4th	Waller - West Tributary	Waller @ 3rd	Waller @ 9th A	Waller @ 9th B	Waller @ 24th	Waller @ 41st	Waller @ Skyview
SAMPLING		DATE	9/6/2002		9/6/2002	9/6/2002	9/6/2002	7/15/2003	7/15/2003	7/15/2003	7/15/2003	7/15/2003	7/15/2003	7/15/2003	7/15/2003
		nº samples	whole suite		THM ♦	whole suite	whole suite	THM	THM	THM	THM	THM	THM	THM	THM
FIELD PARAMETERS	T	ºC													
	pH		9.7			10.2	9.7								
	conductivity	µS/cm	306			236	213								
	turbidity	TU	0.06			0.06	0.07								
ALKALINITY		DATE	9/6/2002		9/6/2002	9/6/2002	9/6/2002								
	0.1N HCl	mℓ	0.22			0.24	0.21								
	from 25 ml	mg/ℓ	53.68			58.32	50.02								
	0.1N HCl	mℓ			0.26										
	from 10 ml	mg/ℓ			156.16										
ANIONS		DATE	2/1/2003	2/1/2003		2/1/2003	2/1/2003								
	Cl	mg/ℓ	22.49	22.28	18.45	23.42	22.30								
	NO_2	mg/ℓ	0.03	0.03	< 0.005	0.00	0.00								
	Br	mg/ℓ	0.00	0.00	0.09	0.00	0.00								
	NO_3	mg/ℓ	0.68	0.68	0.05	0.89	0.56								
	PO_4	mg/ℓ	0.00	0.00	< 0.020	0.00	0.00								
	SO_4	mg/ℓ	19.06	18.84	14.77	19.10	21.08								
		DATE	2/3/2003			2/3/2003	2/3/2003								
MAJOR CATIONS	Na	ppm	10.94		12.10	12.99	12.89								
	Mg	ppm	7.43		13.79	9.19	7.84								
	Si	ppm	4.13		4.70	4.17	3.97								
	K	ppm	3.30		3.07	3.11	3.24								
	Ca	ppm	12.55		45.09	14.67	14.82								
MINOR CATIONS	Li	ppb	3.72		4.26	5.10	4.71								
	B	ppb	47.11		N/A	47.56	46.05								
	Al	ppb	< 6.75		< 6.75	< 6.75	< 6.75								
	P	ppb	< 480		N/A	< 480	< 480								
	V	ppb	4.08		2.76	4.39	4.21								
	Cr	ppb	0.21		< 0.15	0.24	0.24								
	Mn	ppb	< 0.24		1.70	< 0.24	< 0.24								
	Fe	ppb	< 1.2		10.28	24.19	1.34								
	Co	ppb	0.63		< 0.45	0.62	0.65								
	Ni	ppb	< 1.35		< 1.35	< 1.35	< 1.35								
	Cu	ppb	< 1.5		1.84	< 1.5	9.36								
	Zn	ppb	< 3.6		22.42	< 3.6	< 3.6								
	As	ppb	< 0.75		1.64	< 0.75	< 0.75								
	Se	ppb	< 1.95		< 1.95	< 1.95	< 1.95								
	Rb	ppb	1.11		1.12	1.09	1.12								
	Sr	ppb	101		275	114	106								
	Mo	ppb	1.04		0.77	0.98	1.02								
	Cs	ppb	< 0.06		N/A	< 0.06	< 0.06								
	Ba	ppb	11.60		46.37	7.90	9.64								
	La	ppb	< 0.06		N/A	< 0.06	< 0.06								
	Pb	ppb	< 0.12		0.19	0.15	0.30								
	Bi	ppb	0.07		N/A	0.07	0.08								
	Th	ppb	0.71		N/A	0.74	0.69								
	U	ppb	N/A		0.50	N/A	N/A								
	Cd	ppb	0.11		N/A	0.11	0.11								
	Ag	ppb	< 0.15		N/A	< 0.15	< 0.15								
	Hg	ppb	4.07		N/A	3.93	3.70								
	Sn	ppb	< 0.24		N/A	< 0.24	< 0.24								
	Sb	ppb	0.20		N/A	0.18	0.20								
STRONTIUM ♦♦		DATE			2/1/2004										
	$^{87}Sr/^{86}Sr$	ratio			0.70891										
TRIHALO-METHANES ♣		DATE	9/19/2002		9/19/2002	9/19/2002	9/19/2002	7/21/2003	7/21/2003	7/21/2003	7/21/2003	7/21/2003	7/21/2003	7/21/2003	7/21/2003
	TCM	µg/ℓ	25.80		< 1.00	28.70	42.68	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00
	BDCM	µg/ℓ	7.63		< 1.00	11.74	10.94	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00
	DBCM	µg/ℓ	0.69		< 1.00	2.21	1.23	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00
	TBM	µg/ℓ	< 1.00		< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00
	TOTAL THM	µg/ℓ	34.12		< 1.00	42.65	54.85	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00	< 1.00
δ^{15}N (air) ♥		DATE													
	in NH_3	$^{o}/_{oo}$	-		-	-	1.50								
	in NO_3^- ♠	$^{o}/_{oo}$	1.20		-1.30	0.70	-								

FOOTNOTES:

- ♦ Lance Christian collected the rest of the suite (except δ^{15}N and THM)
- ♦♦ Data shared with Lance Christian
- ♣ Detection limit: 1µg/ℓ
- ♥ Precision: $1^{0}/_{00}$
- ♠ Nitrate + nitrite

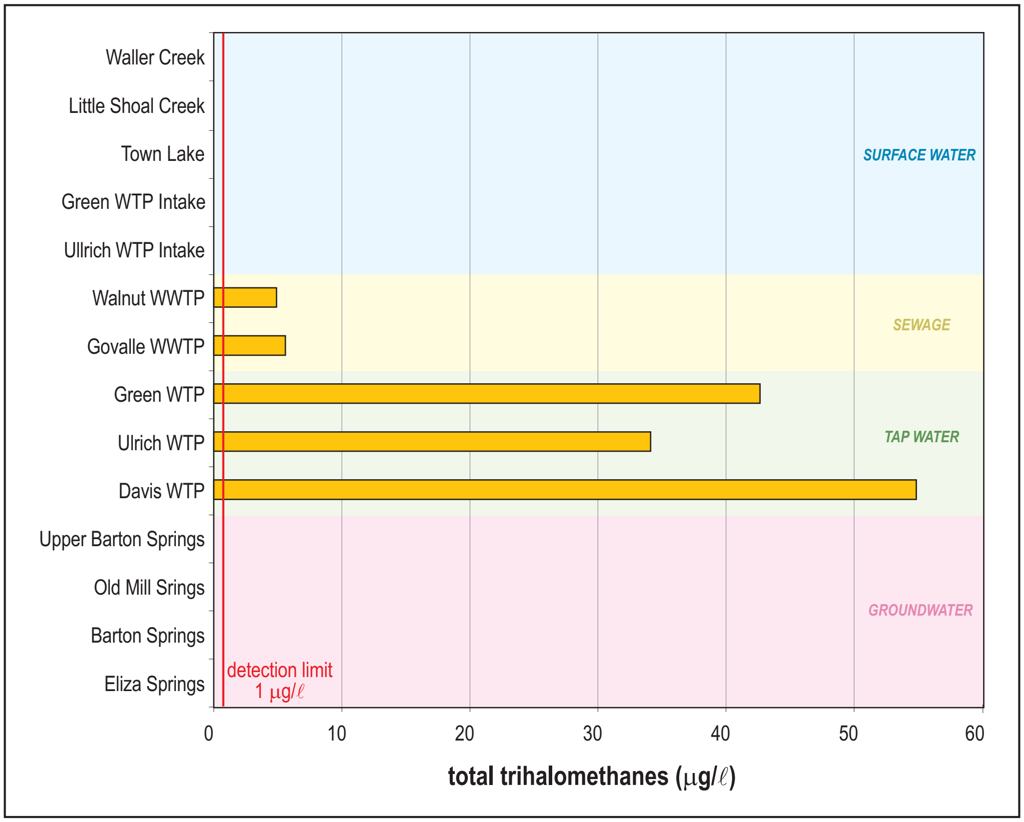


Figure 5.38: Total trihalomethanes in surface water, sewage from two wastewater treatment plants, tap water from three water treatment plants, and groundwater from the Barton Springs system.

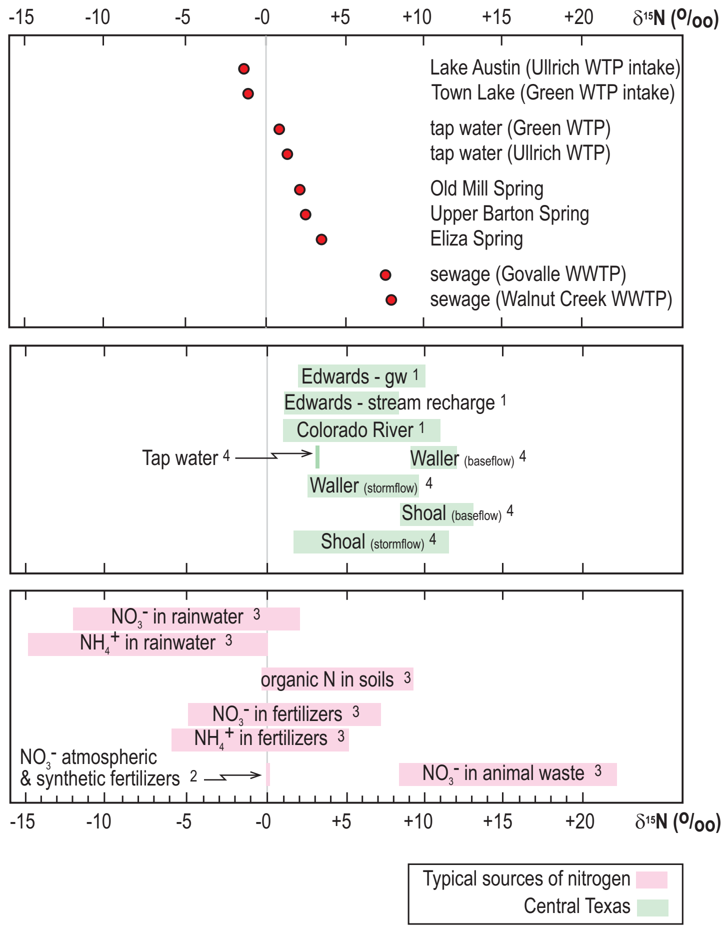


Figure 5.39: $\delta^{15}N$ of dissolved nitrate in waters from the City of Austin compared to studies by 1) Kreitler and Browning, 1983; 2) Mariotti, 1984; 3) Heaton, 1986; and 4) Ging et al., 1996.

Conclusions

Urban development affects the rate and location of direct recharge and introduces new sources of indirect and artificial recharge. A review of the literature demonstrates groundwater recharge is almost always increased in urban areas. Only one of the twenty three cities reported shows a slight decrease in groundwater recharge.

The principal sources of recharge in Austin are: irrigation return flows, precipitation, leakage from water mains, leakage from sewers, and designed infiltration structures, including on-site sewage treatment systems. A water balance for the city indicates that the amount of water potentially available for recharge has nearly doubled under urban conditions, and the urban sources of recharge (irrigation and leakage) provide larger amounts of water than the natural sources (precipitation). Assuming a mains leakage around 7%, most of the recharge takes place by infiltration of irrigation return flow, followed by direct infiltration of rainfall, mains leakage, and sewer leakage.

A hydrogeological analysis of the local geology of a city provides a ready tool to infer the spatial distribution of recharge rates. When this is combined with the spatial distribution of the water and wastewater networks and impervious cover, the locations of preferential urban recharge can be assessed. In the city of Austin, the

Edwards aquifer and the Quaternary deposits are the most important hydrogeologic units and thus the *loci* of most urban recharge. The pervious/impervious ratio is smaller in the Edwards, which increases the relevance of this aquifer with respect to recharge compared to the Quaternary deposits.

Environmental isotopes combined with traditional hydrochemical parameters and trihalomethanes are promising tracers of the urban sources of recharge. Although trihalomethanes appear to be an inadequate tracer of recharge to the groundwater from mains leakage in Austin, further exploration of this method is recommended. It is especially desirable to lower the detection limit of the analytical method. The data obtained from $\delta^{15}N$ in this study is inconclusive. However, it may yet be worthwhile to explore this parameter as a tracer of the sources of nitrate and an indicator of urban influence on groundwater. Besides its potential as a tracer of sewage, its potential as an indicator of irrigation return flow should also be tested, as it is a potential tracer of nitrogen compounds found in fertilizer.

Different land uses in the Barton Springs segment of the Edwards aquifer favor different infiltration mechanisms. The $^{87}Sr/^{86}Sr$ values from groundwater of wells in less urbanized areas are lower (i.e., closer to those of limestones) than values for samples from wells in densely urbanized areas. This reflects longer residence times and greater degree of water-rock interaction for waters in the *rural* areas. It could also reflect inputs from surface waters having a higher $^{87}Sr/^{86}Sr$ value in urbanized areas. Increasing impervious cover favors the concentration of runoff and its infiltration through discrete flow pathways. Therefore groundwaters in urban wells are not as evolved as the ones infiltrated in the *rural* areas. Some samples, generally in the urban zone, show signs of fluid mixing between fresh groundwaters of meteoric origin and saline groundwaters from the deeper parts of the aquifer. Water from well 58-42-915 is likely mixed with saline water from the Glen Rose formation, but the high lead content indicates this well could be receiving pollution from some other sources. Leaky drinking water pipes could also contribute to the higher ratios found in *urban* samples. However, the role of leaky utilities on these groundwater samples is yet to be studied and further research is needed in order to improve the application of this geochemical parameter as an urban tracer.

Although the impacts of urbanization on groundwater are beginning to receive more attention, the net effects are not always easy to assess. Numerous parameters, processes, and feedbacks are still poorly understood, and site-specific conditions may be paramount. Significant efforts are still needed in order to implement methods for studying recharge in the urban environment. Some suggestions are given in the next section. The involvement of geoscientists, especially hydrologists and hydrogeologists, is critical for the development of livable urban areas, as well as for the sustainable management of the waters necessary to maintain urban systems.

Future Work

The present study is the initial effort to estimate urban effects on groundwater recharge in Austin. Significant further work is necessary to validate the conceptual model proposed in this thesis, including the following actions:

An exhaustive inventory of hydrologic features in the City of Austin, identifying points of discharge (e.g., springs) and recharge (e.g., sinkholes and designed infiltration structures). Such study should cover both karstic and clastic aquifers.

An evaluation of alteration of the permeability field of the shallow urban underground by utility trenches and other buried urban structures. This can be accomplished by experimental tests on a scaled model (including groundwater flow direction, and tracer tests) and by a selection of shallow geophysical tests, such as electrical resistivity.

An assessment of the effects of the water and wastewater network characteristics on the spatial distribution of leakage. Examples of parameters to study include the material and size of the pipes, age and maintenance of the network, and the effect of different pressure zones on leakage rates (Farley and Trow, 2003).

An in-depth study of the hydrogeochemistry of groundwaters and urban endmember waters in the study area. This should include completing the dataset and performing charge balances. Further exploration of the selected tracers is also desirable (TTHM, and the isotope composition of dissolved strontium and nitrogen in dissolved nitrate), and perhaps new ones. For instance, the isotope composition of boron, an element present in domestic detergents, is a potential tracer of sewage. The sampling scheme should incorporate examples of all the recharge and discharge end-members allowing the construction of a mass-balance in order to quantify the relative contributions of the different recharge sources. This can be carried out on a city-wide and low resolution scale, or on a higher resolution study conducted on a selected representative study area within the city.

A literature review and initial field-based assessment of the potential effects of urban development on the local climate. Some of the potential topics include the urban heat island and variations on the wind regime, which highly influence evapotranspiration rates.

Finally, the potential uses of increased urban recharge should be explored. Such assessment should lead to define strategies to incorporate urban recharge into the comprehensive management of the local water resources.

Appendix 1

This appendix compiles the images used in the assessment of the areal growth of the City of Austin (Figures A1.1-9). Because of the disparity in scales and quality of the different maps, three reference points were used to correlate the images: the mouths of both Shoal and Waller Creeks into Town Lake, and the Capitol. The product of this exercise (Figure 4.10) provides a useful visual aid to qualitatively assess the urban growth of the city and should not be used for quantitative purposes.

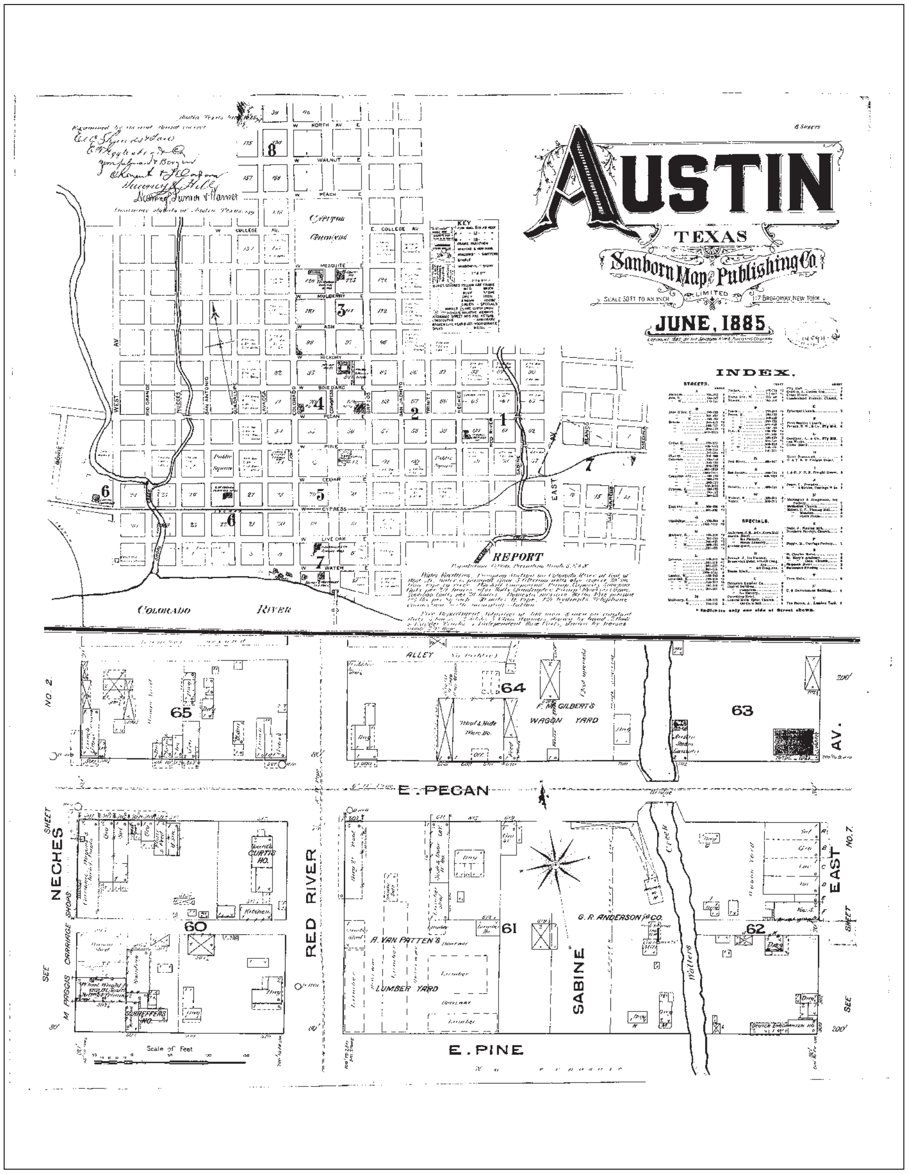


Figure A1.1: Map of Austin in 1885 (source: Sanborn Digital Maps, online).

Figure A1.2: Map of Austin in 1891 (source: Texas General Land Office, online).

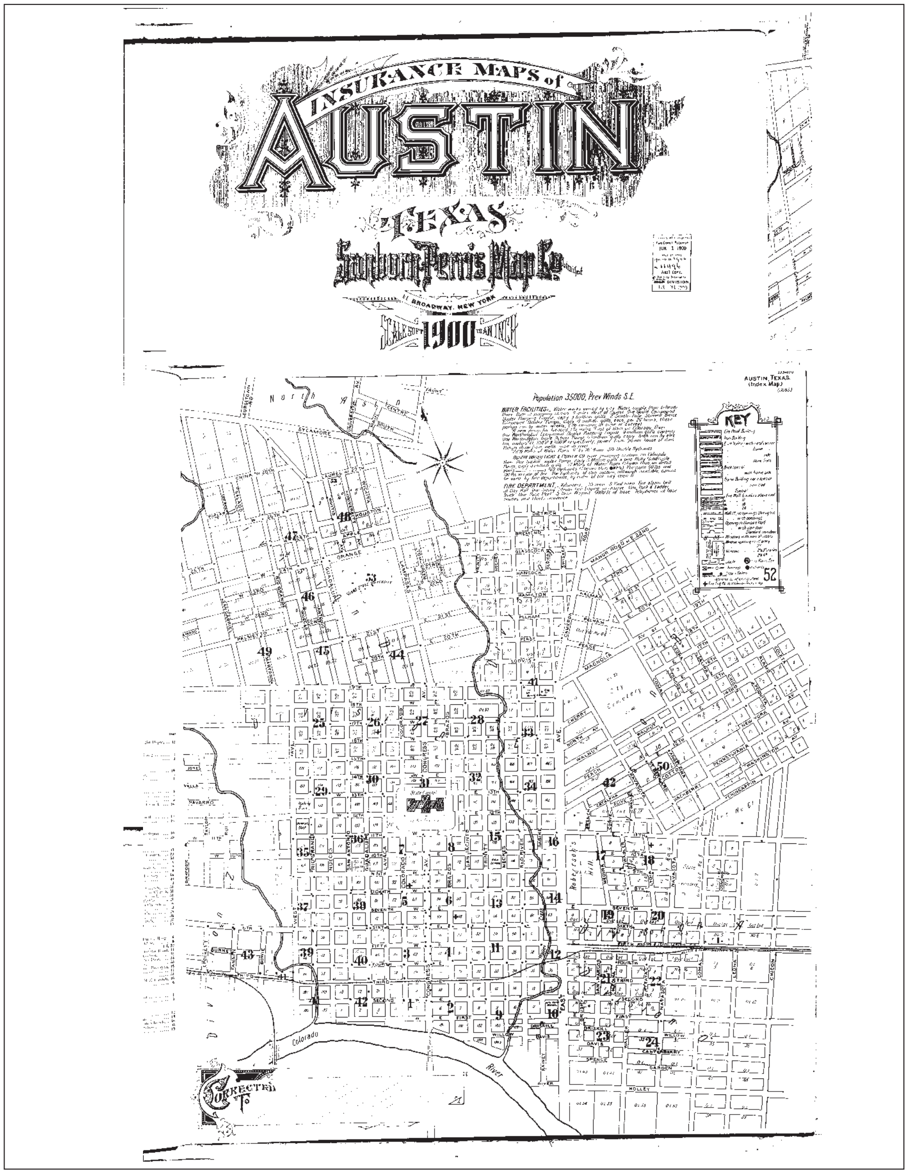


Figure A1.3: Map of Austin in 1900 (source: Sanborn Digital Maps, online).

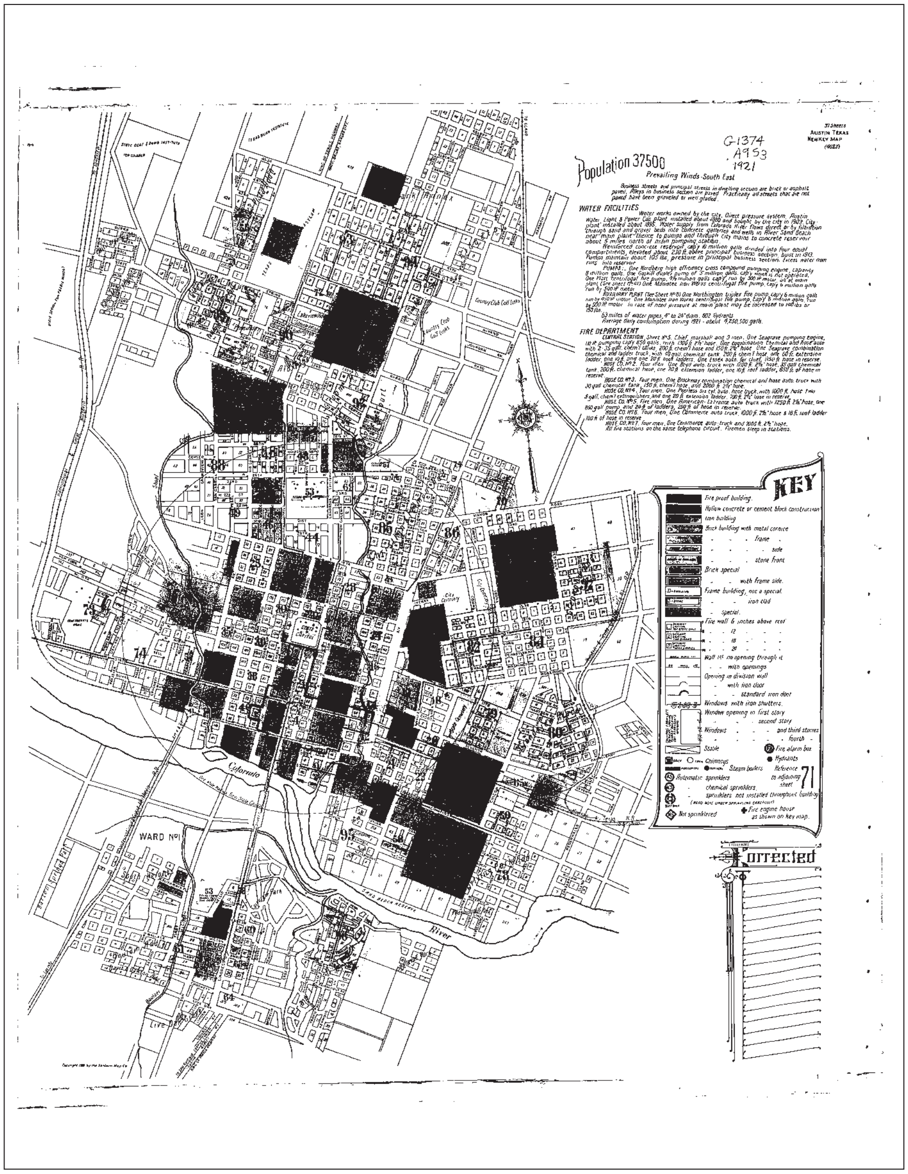


Figure A1.4: Map of Austin in 1922 (source: Sanborn Digital Maps, online).

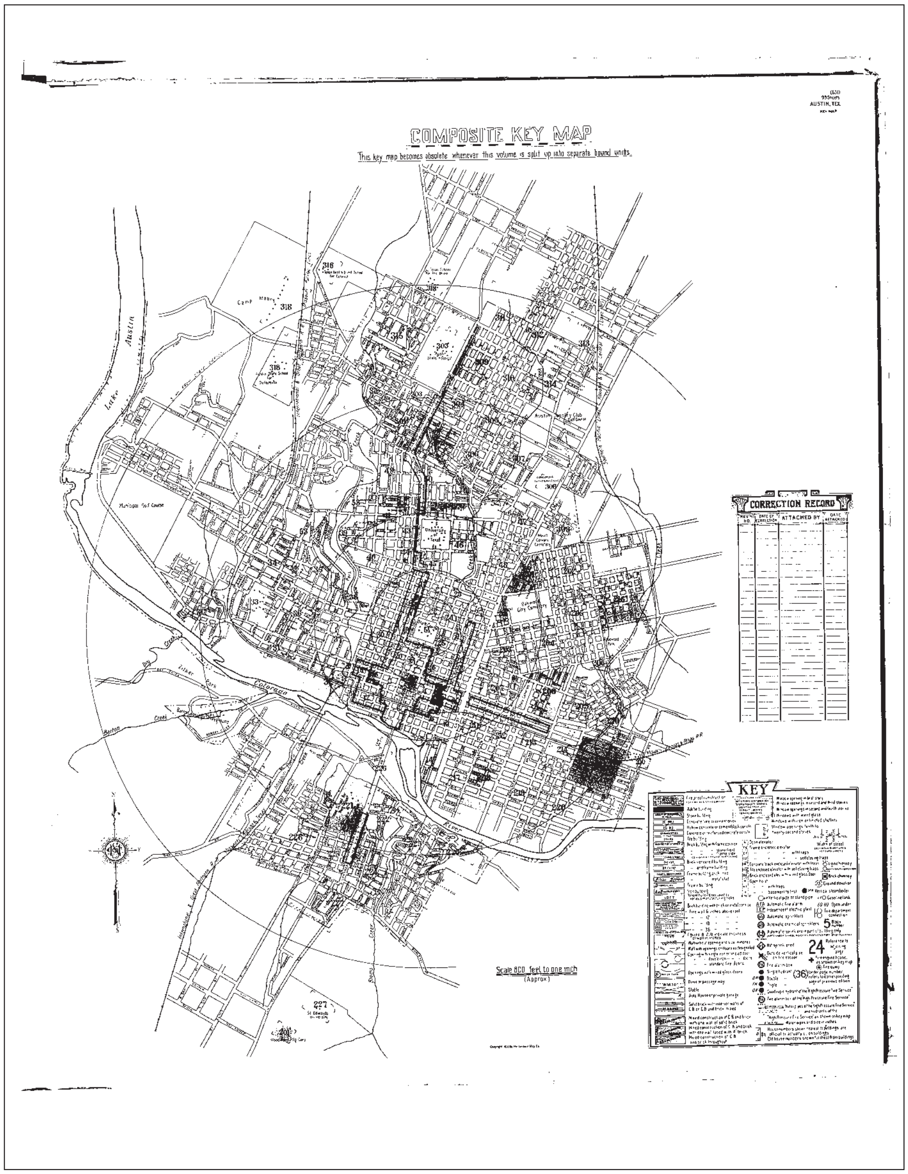


Figure A1.5: Map of Austin in 1935 (source: Sanborn Digital Maps, online).

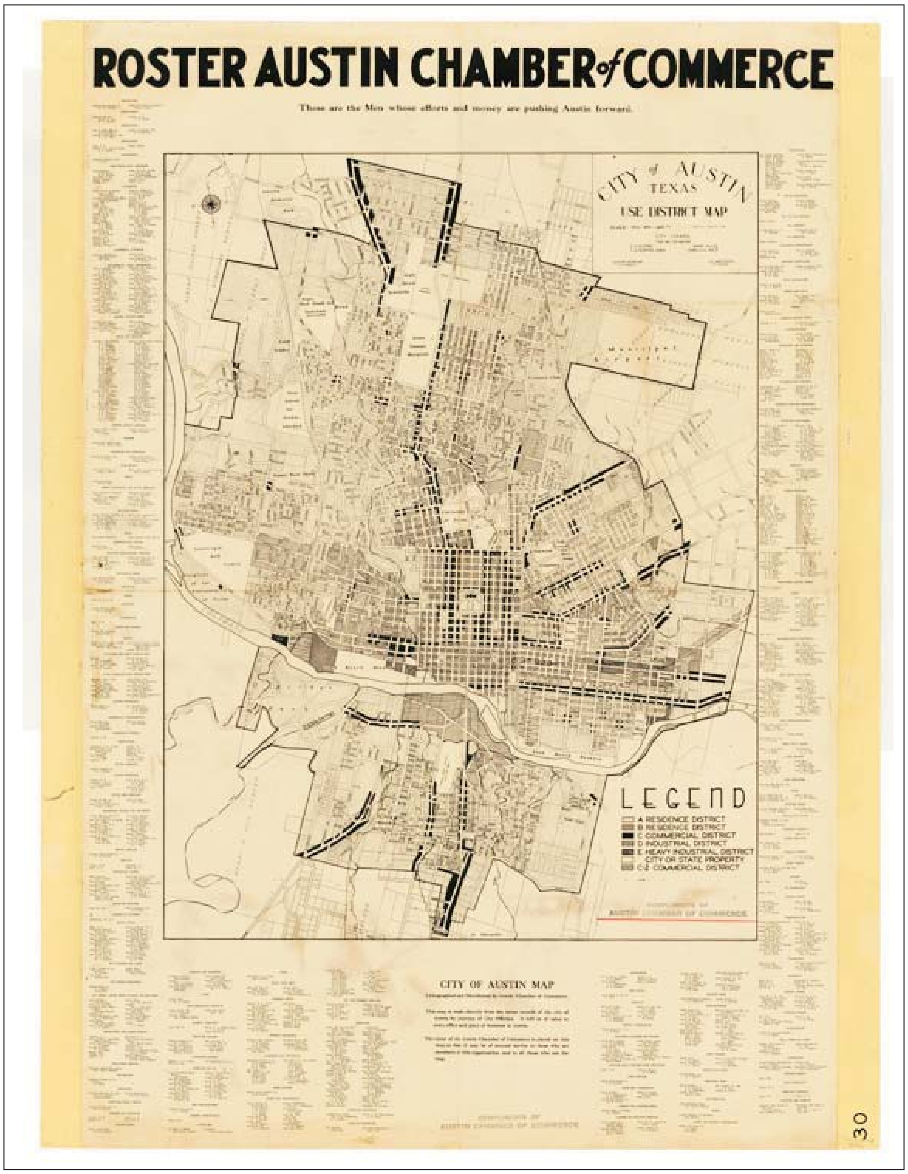


Figure A1.6: Map of Austin in 1939 (source: Texas General Land Office, online).

Figure A1.7: Map of Austin in 1950 (source: Texas General Land Office, online).

Figure A1.8: Map of Austin in 1964 (source: Texas General Land Office, online).

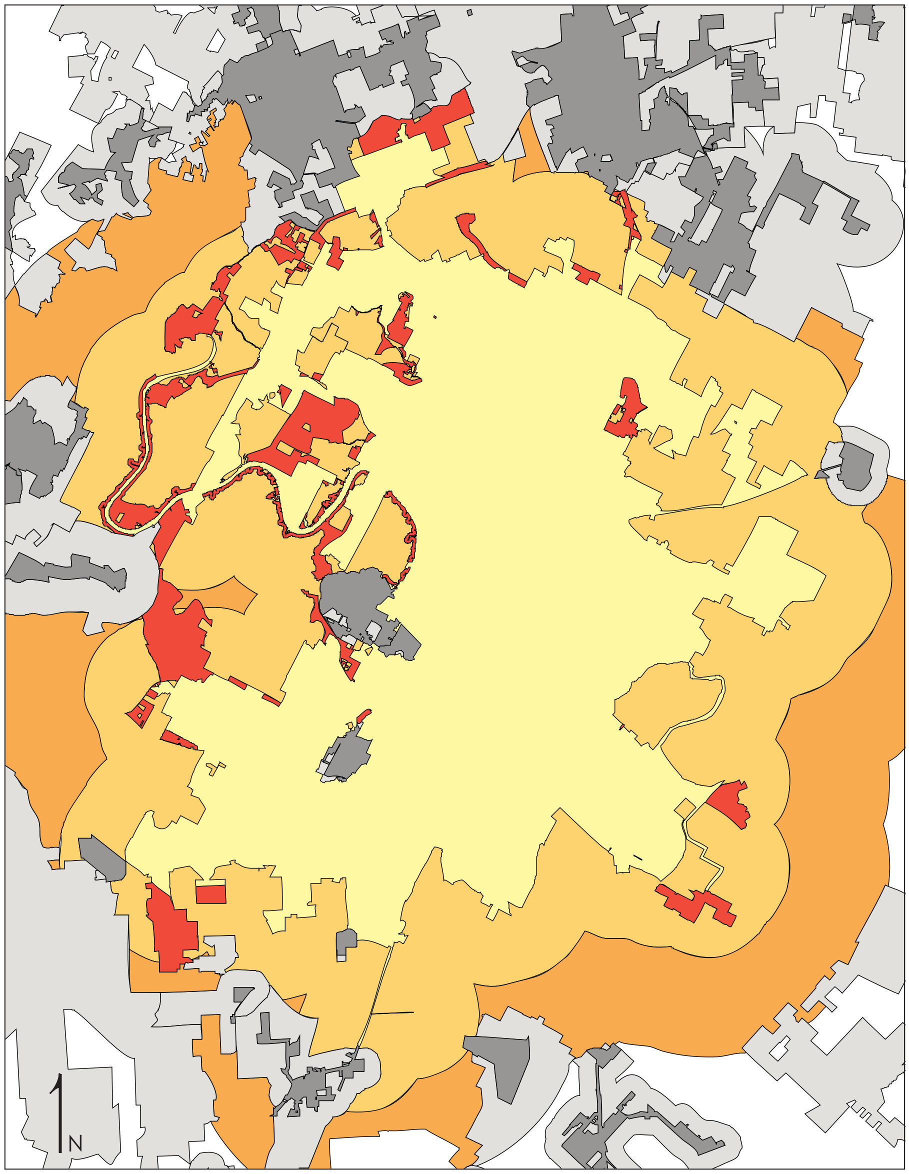


Figure A1.9: Digital map of Austin in 1985 (source: City of Austin, public GIS datasets, online).

Appendix 2

Table A2.1 compiles all the analytical results of $^{87}Sr/^{86}Sr$ of waters collected in both sampling campaigns of this study. The internal precision (machine error) of each measurement is expressed as two standard deviations of the mean value ($2\sigma_{m}$). This means 95% of the time the true value will be within $\pm 2\sigma$ of the measured value

The uncertainty or external precision of the measurements is determined as two standard deviations of the population of values ($2\sigma_{pop}$), for all measurements of the NBS 987 standard (Table A2.2, and Figure A2.1). The long term external precision, based on hundreds of determinations on the standard, is ±0.000016. This means 95% of the time the true value will be within ±0.000016 of the measured value. The external precision of this dataset is ±0.000017, which is very close to the long term value.

Tables A2.3 and A2.4 contain details about the cation determinations for the first sampling campaign.

Table A2.1: Compilation of $^{86}Sr/^{87}Sr$ values measured for both sampling campaigns.

		SAMPLES	$^{87}Sr/^{86}Sr$	Internal precision (2σ m) ±
1	4/4/2001	58-54-914	0.707964	0.000008
	4/4/2001	58-50-207	0.708978	0.000027
2	4/4/2001	58-57-3EC	0.707967	0.000010
3	3/23/2001	58-50-211	0.708245	0.000011
4	3/23/2001	58-57-3ES	0.707604	0.000011
5	3/23/2001	58-50-742	0.707930	0.000011
6	3/23/2001	58-50-201	0.707984	0.000008
7	3/23/2001	Cold Springs	0.708020	0.000008
8	3/23/2001	58-58-121	0.707755	0.000007
9	3/23/2001	58-58-225	0.707915	0.000008
10	3/23/2001	59-42-915	0.708199	0.000007
11	3/23/2001	59-58-4LO	0.707980	0.000010
	3/23/2001	53-50-207	0.708928	0.000027
12	3/23/2001	58-42-921	0.708149	0.000009
13		spliced 53-50-207	0.708953	0.000019
14	7/29/2004	Barton Springs - pool	0.707957	0.000007
15	4/17/2004	Govalle WWTP - raw influent sewage	0.708671	0.000006
16	4/17/2004	Walnut Creek WWTP - raw influent sewage	0.708102	0.000008
17	2/1/2004	Ullrich Intake - Lake Austin (RAW)	0.709240	0.000007
18	2/1/2004	Green Intake - Town Lake (2) - (RAW)	0.708911	0.000008

first sampling campaign

second sampling campaign

Table A2.2: $^{86}Sr/^{87}Sr$ values of standard sample NBS 987 measured for both sampling campaigns.

	NBS 987	$^{87}Sr/^{86}Sr$	Internal precision (2σ m) ±
1	4/4/2001	0.710283	0.000008
2	3/23/2001	0.710256	0.000018
3	3/23/2001	0.710264	0.000009
	mean 2001	0.710268	
	External precision (2σ pop) ±	0.000023	
4	7/29/2003	0.710274	0.000007
5	7/29/2003	0.710273	0.000009
6	2/1/2004	0.710263	0.000008
7	2/1/2004	0.710262	0.000010
8	4/17/2004	0.710255	0.000009
9	4/17/2004	0.710261	0.000008
	mean 2004	0.710265	
	External precision (2σ pop) ±	0.000013	
	overall mean	0.710266	
	External precision (2σ pop) ±	0.000017	

first sampling campaign

second sampling campaign

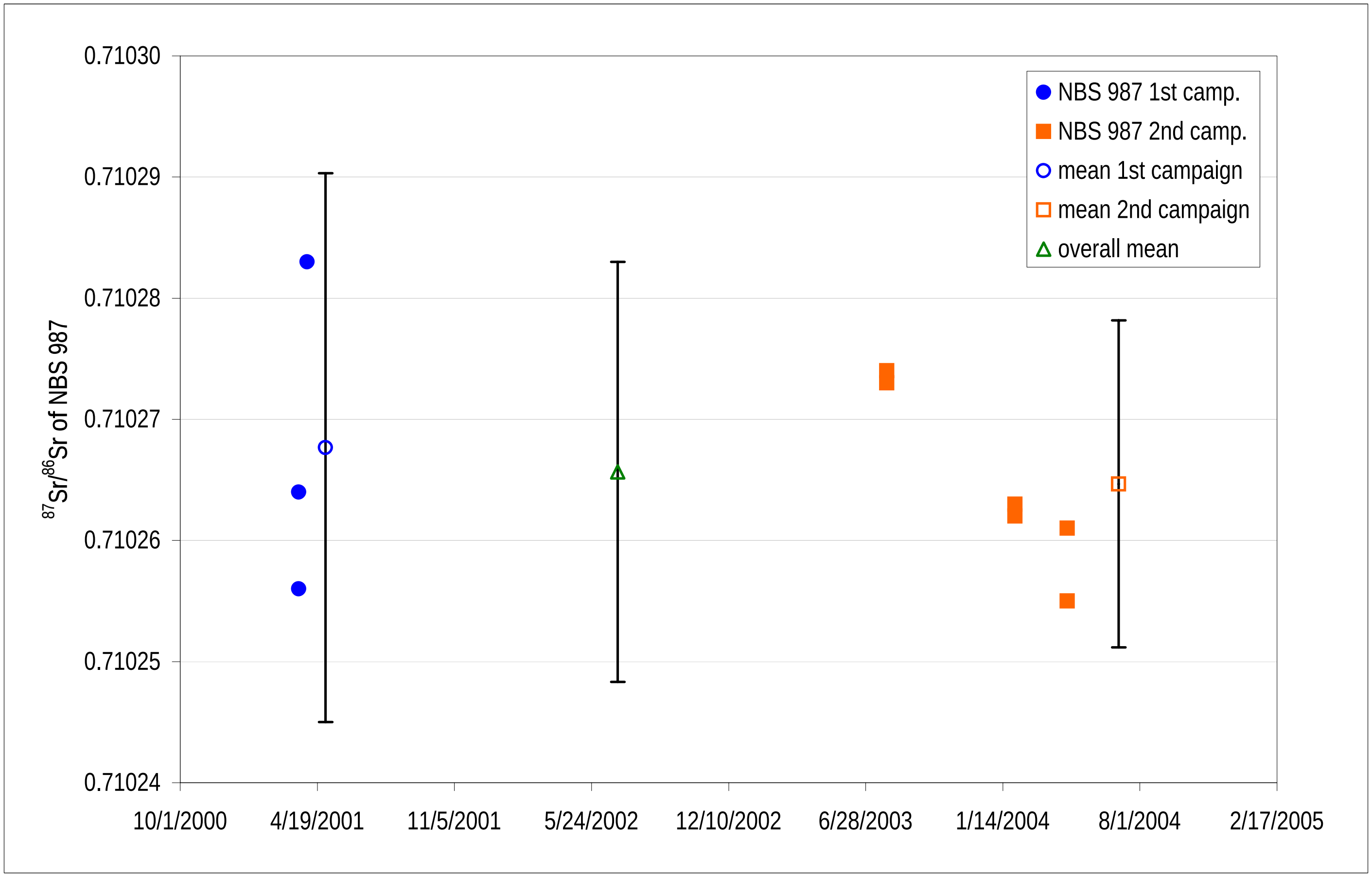


Figure A2.1: Uncertainty of $^{86}Sr/^{87}Sr$ values measured for this study, based on reproducibility of standard sample NBS 987 (dates for mean values were arbitrarily assigned and have no meaning).

Table A2.3: Raw intensity data for major cations and trace metals on samples from the first sampling campaing, prior to blank subtraction. Detection limits are for samples diluted 20 times (instrument detection limits are approximately 10 times lower). The analytical uncertainty is represented by the standard deviations associated with the concentration values reported. For concentrations that are greater than 10 times the detection limit the precisions are 1-3% the RSD (relative standard deviation). Detection limits are defined as 3 times the standard deviation of the blank.

4/11/2001	Na	Mg	Al	Si	P	K	Ca	Sc	Fe	Mn	Rb	Sr	Rh	Cd	Ba	Pb	Th	U
Blank[3]	1398348	1769	4815	77073	6496	688803	4589	955280	50712	1787	3610	297	1784231	7	362	177	213	75
SD of Blank[3]	12420	41	296	578	211	5764	62	132971	386	39	64	27	8389	2	60	10	5	9
%RSD of Blank[3]	0.89	2.29	6.14	0.75	3.25	0.84	1.36	13.92	0.76	2.20	1.78	9.09	0.47	35.02	16.51	5.39	2.22	12.40
Water-1 Std[3]	21307490	2395515	374140	234928	57474	7558164	1440000	970938	378278	2368034	514735	681406	1774150	77247	629304	449220	1365365	1385309
SD of Water-1 Std[3]	310526	25195	3939	1813	880	66327	13393	89161	4103	29695	4567	6918	14300	645	9402	4990	10781	11914
%RSD of Water-1 Std[3]	1.46	1.05	1.05	0.77	1.53	0.88	0.93	9.18	1.09	1.25	0.89	1.02	0.81	0.84	1.49	1.11	0.79	0.86
COLD SPRINGS	15891431	2130939	6562	170836	6770	1994586	2643264	1061653	54777	2310	4635	355040	1860008	13	66366	358	1340	2018
SD of COLD SPRINGS	471340	82798	117	2417	88	45565	89966	39934	417	98	74	13280	66708	3	3251	16	120	102
%RSD of COLD SPRINGS	2.97	3.89	1.79	1.42	1.30	2.28	3.40	3.76	0.76	4.25	1.59	3.74	3.59	20.41	4.90	4.38	8.96	5.04
58-42-914	12566644	1858371	12527	181517	6646	1905474	2701440	927874	54338	4829	4865	1268496	1771370	4	55250	145	112	1376
SD of 58-42-914	426630	89706	432	4327	157	36722	120857	137351	285	137	92	63643	76722	1	2213	10	12	91
%RSD of 58-42-914	3.40	4.83	3.45	2.38	2.37	1.93	4.47	14.80	0.52	2.84	1.88	5.02	4.33	28.85	4.01	6.89	10.82	6.63
58-42-915	24132570	5330104	97897	201121	6773	4130172	5133888	944883	82233	15828	11515	2854613	1801263	98	42297	8574	243	1888
SD of 58-42-915	664903	181399	9331	4753	40	69473	147149	109359	881	346	242	78336	47336	11	988	332	19	43
%RSD of 58-42-915	2.76	3.40	9.53	2.36	0.59	1.68	2.87	11.57	1.07	2.18	2.10	2.74	2.63	10.81	2.34	3.87	7.92	2.27
58-42-921	9998058	2164011	15077	195430	7094	1959336	2848032	984211	54503	3734	4823	518790	1813082	4	137995	205	41	1576
SD of 58-42-921	262896	69323	1031	3686	116	30037	80681	103921	431	105	146	17405	53729	2	5580	13	10	50
%RSD of 58-42-921	2.63	3.20	6.84	1.89	1.63	1.53	2.83	10.56	0.79	2.80	3.02	3.36	2.96	51.45	4.04	6.18	23.27	3.18
Blank[4]	1328040	1823	4871	77902	6470	681940	4696	950661	50372	1709	3553	312	1783188	6	327	183	214	81
SD of Blank[4]	4872	83	605	558	131	6424	98	131159	431	45	31	29	18880	3	16	17	6	7
%RSD of Blank[4]	0.37	4.57	12.42	0.72	2.02	0.94	2.09	13.80	0.86	2.62	0.88	9.18	1.06	49.14	4.94	9.13	2.86	8.71
Water-1 Std[4]	20881476	2354451	368279	231346	56716	7452414	1425888	930309	373221	2318035	509085	678394	1759204	76463	625909	444268	1357924	1375782
SD of Water-1 Std[4]	247450	28284	5536	2825	1011	111075	28717	110348	3897	45210	6021	7325	19945	416	5870	3034	16587	9031
%RSD of Water-1 Std[4]	1.19	1.20	1.50	1.22	1.78	1.49	2.01	11.86	1.04	1.95	1.18	1.08	1.13	0.54	0.94	0.68	1.22	0.66
58-50-201	18074104	1943531	6697	188269	6660	6535914	2998944	987921	58216	5886	5561	5704589	1791530	43	118237	1252	178	1310
SD of 58-50-201	338753	20173	145	1894	164	78237	33734	81104	600	162	92	31364	6995	9	1537	37	33	47
%RSD of 58-50-201	1.87	1.04	2.17	1.01	2.47	1.20	1.13	8.21	1.03	2.75	1.66	0.55	0.39	21.66	1.30	2.94	18.42	3.59
58-50-207	5706145	2503275	14925	213379	6667	2116410	2750112	978478	54026	2314	4579	158365	1766156	7	147948	474	54	1036
SD of 58-50-207	63476	17447	609	1863	109	27684	23161	87036	503	37	68	1748	13029	1	741	10	7	23
%RSD of 58-50-207	1.11	0.70	4.08	0.87	1.63	1.31	0.84	8.90	0.93	1.61	1.48	1.10	0.74	19.63	0.50	2.12	12.35	2.25
58-50-211	8827820	2427760	51409	189032	6950	1610220	2340576	1015092	56174	4836	4200	245155	1774846	5	117313	2349	48	1194
SD of 58-50-211	111537	25040	3196	1499	155	23949	24323	8972	721	124	131	2335	18935	1	2560	50	6	37
%RSD of 58-50-211	1.26	1.03	6.22	0.79	2.22	1.49	1.04	0.88	1.28	2.56	3.13	0.95	1.07	29.54	2.18	2.13	13.09	3.07
58-50-225	11882573	1834118	7960	168173	6908	2093004	2088288	1013068	55214	2220	4259	291382	1774846	10	46087	870	31	1358
SD of 58-50-225	153563	10079	234	1256	159	45281	21100	10557	528	65	174	3719	15467	3	697	21	7	46
%RSD of 58-50-225	1.29	0.55	2.94	0.75	2.31	2.16	1.01	1.04	0.96	2.94	4.09	1.28	0.87	28.35	1.51	2.46	21.90	3.42
Blank[4a]	1287938	1784	5507	76596	6353	661055	4648	946899	49922	1640	3366	338	1777279	7	327	176	211	74
SD of Blank[4a]	8992	53	1125	306	142	2555	79	132349	455	46	93	15	15181	1	15	14	9	4
%RSD of Blank[4a]	0.70	2.95	20.42	0.40	2.23	0.39	1.71	13.98	0.91	2.81	2.75	4.56	0.85	19.44	4.67	7.96	4.07	5.67
Water-1 Std[4a]	20632794	2356656	367531	230141	56288	7478358	1433952	966634	374720	2344842	511598	676272	1763375	76548	624975	441953	1361740	1384001
SD of Water-1 Std[4a]	244928	21970	1849	1942	174	104532	24319	87342	3898	35766	4933	7153	15477	883	5933	5427	12154	7186
%RSD of Water-1 Std[4a]	1.19	0.93	0.50	0.84	0.31	1.40	1.70	9.04	1.04	1.53	0.96	1.06	0.88	1.15	0.95	1.23	0.89	0.52
58-50-742	7979437	2092906	5308	187119	6505	1774908	2670912	1032155	56202	2864	4453	703582	1818991	6	45193	486	159	1458
SD of 58-50-742	116482	16633	107	2176	148	16665	36888	12539	498	70	89	6029	19686	3	497	25	38	44
%RSD of 58-50-742	1.46	0.80	2.02	1.16	2.28	0.94	1.38	1.22	0.89	2.46	1.99	0.86	1.08	49.69	1.10	5.13	23.57	3.03
58-57-3EC	6746704	1859749	4137	183256	6531	1657596	2558016	1025792	53348	1366	4263	338502	1810996	5	39919	363	55	1521
SD of 58-57-3EC	101687	8856	168	1112	109	20496	35001	12958	598	42	155	3195	13740	1	466	23	5	65
%RSD of 58-57-3EC	1.51	0.48	4.06	0.61	1.66	1.24	1.37	1.26	1.12	3.05	3.64	0.94	0.76	25.35	1.17	6.31	9.54	4.26
58-57-3ES	6704519	1985974	38653	180661	6478	1800006	2511648	1029263	53460	1979	4711	2826130	1807868	7	41556	668	38	1240
SD of 58-57-3ES	60101	9467	889	1231	104	24312	21489	9225	671	58	77	30839	17047	2	625	26	5	48
%RSD of 58-57-3ES	0.90	0.48	2.30	0.68	1.61	1.35	0.86	0.90	1.26	2.93	1.64	1.09	0.94	25.07	1.50	3.88	12.32	3.84
58-58-121	6005345	1963650	4072	167024	6713	1836666	1775808	744762	53286	1409	4388	2527387	1631287	13	37538	19164	35	1445
SD of 58-58-121	70046	36614	94	1011	107	11864	23627	10207	533	57	53	47354	20066	3	787	388	5	51
%RSD of 58-58-121	1.17	1.87	2.31	0.61	1.59	0.65	1.33	1.37	1.00	4.05	1.20	1.87	1.23	19.69	2.10	2.02	13.49	3.56
58-58-410	7661489	1561274	11648	154259	6623	1824258	2013120	884661	53379	3205	4041	299874	1768589	5	32332	375	31	1209
SD of 58-58-410	87586	7421	305	1265	81	13317	14650	107237	586	84	61	2846	10615	2	650	17	9	60
%RSD of 58-58-410	1.14	0.48	2.62	0.82	1.22	0.73	0.73	12.12	1.10	2.62	1.51	0.95	0.60	40.94	2.01	4.64	28.34	4.99
Blank[5]	1250180	1761	5195	74741	6396	641659	4552	769250	49491	1693	3171	338	1766851	8	328	170	217	79
SD of Blank[5]	23677	94	794	787	87	5728	71	135216	533	61	72	15	11437	3	22	12	15	7
%RSD of Blank[5]	1.89	5.34	15.29	1.05	1.36	0.89	1.56	17.58	1.08	3.60	2.26	4.58	0.65	36.49	6.67	7.14	7.14	8.61
Water-1 Std[5]	20507021	2337639	363764	226758	55755	7395450	1424448	844819	369012	2325264	503286	668013	1759204	74744	612119	433711	1349910	1366255
SD of Water-1 Std[5]	255465	10060	1996	862	497	119568	16526	89986	3153	24025	5408	8962	25347	1002	7231	6274	12923	11138
%RSD of Water-1 Std[5]	1.25	0.43	0.55	0.38	0.89	1.62	1.16	10.65	0.86	1.03	1.08	1.34	1.44	1.34	1.18	1.45	0.96	0.82

Table A2.4: Concentrations of major cations and trace metals on samples from the first sampling campaing, prior to blank subtraction. Detection limits are for samples diluted 20 times (instrument detection limits are approximately 10 times lower). The analytical uncertainty is represented by the standard deviations associated with the concentration values reported. For concentrations that are greater than 10 times the detection limit the precisions are 1-3% the RSD (relative standard deviation). Detection limits are defined as 3 times the standard deviation of the blank.

4/11/2001	Na	Mg	Al	Si	P	K	Ca	Fe	Mn	Rb	Sr	Cd	Ba	Pb	Th	U
	ppm	ppm	ppb	ppm	ppb	ppm	ppm	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb	ppb
COLD SPRINGS	18.33	22.43	2.33	4.52	5.63	1.45	92.70	57.87	0.35	1.01	259.30	0.04	52.43	0.20	0.41	0.69
SD of COLD SPRINGS	0.59	0.87	0.16	0.12	1.76	0.05	3.15	5.85	0.06	0.07	9.69	0.02	2.58	0.02	0.04	0.04
%RSD of COLD SPRINGS	3.21	3.86	6.80	2.55	31.23	3.46	3.39	10.10	17.55	7.07	3.74	42.87	4.92	8.72	10.65	5.23
58-42-914	14.19	19.62	10.40	5.06	3.25	1.36	94.90	52.72	1.97	1.24	927.50	0.00	43.65	0.00	0.00	0.46
SD of 58-42-914	0.54	0.94	0.58	0.21	3.13	0.04	4.24	4.01	0.09	0.09	46.47	0.01	1.75	0.01	0.00	0.03
%RSD of 58-42-914	3.77	4.80	5.60	4.13	96.29	2.98	4.46	7.61	4.37	7.12	5.01	60.31	4.02	27.82	11.92	7.03
58-42-915	28.94	56.46	126.20	6.03	5.88	3.84	180.80	441.10	9.06	7.70	2089.00	0.58	33.38	9.24	0.01	0.65
SD of 58-42-915	0.84	1.91	12.63	0.23	0.79	0.08	5.16	12.14	0.22	0.23	57.18	0.07	0.78	0.36	0.01	0.02
%RSD of 58-42-915	2.89	3.38	10.00	3.81	13.39	1.99	2.85	2.75	2.41	3.02	2.74	11.57	2.34	3.93	64.78	2.36
58-42-921	11.03	22.98	13.90	5.78	12.40	1.43	100.40	56.85	1.30	1.23	379.80	0.00	109.60	0.03	0.00	0.54
SD of 58-42-921	0.33	0.73	1.40	0.18	2.31	0.03	2.83	6.02	0.07	0.14	12.72	0.01	4.43	0.01	0.00	0.02
%RSD of 58-42-921	2.99	3.18	10.09	3.10	18.66	2.32	2.82	10.59	5.13	11.48	3.35	87.31	4.04	54.86	5.59	3.34
58-50-201	21.49	20.68	2.18	5.46	4.64	6.54	105.70	111.40	2.71	2.01	4186.00	0.23	94.02	1.19	0.00	0.44
SD of 58-50-201	0.44	0.22	0.20	0.09	3.33	0.09	1.18	8.41	0.10	0.09	23.11	0.06	1.23	0.04	0.01	0.02
%RSD of 58-50-201	2.04	1.04	9.26	1.73	71.83	1.33	1.12	7.55	3.86	4.57	0.55	25.45	1.31	3.44	95.47	3.82
58-50-207	5.65	26.64	13.28	6.71	5.19	1.61	96.78	54.10	0.41	1.09	116.00	0.00	117.70	0.33	0.00	0.34
SD of 58-50-207	0.08	0.19	0.82	0.09	2.18	0.03	0.81	7.03	0.02	0.06	1.29	0.01	0.59	0.01	0.00	0.01
%RSD of 58-50-207	1.47	0.70	6.18	1.38	42.06	1.93	0.84	13.00	5.82	5.83	1.11	243.60	0.50	3.45	4.19	2.43
58-50-211	9.69	25.83	63.03	5.52	11.27	1.05	82.27	84.94	2.04	0.75	179.80	0.00	93.33	2.41	0.00	0.40
SD of 58-50-211	0.15	0.27	4.37	0.08	3.13	0.03	0.85	10.07	0.08	0.13	1.72	0.01	2.04	0.06	0.00	0.01
%RSD of 58-50-211	1.50	1.03	6.93	1.35	27.72	2.56	1.03	11.85	3.92	16.66	0.96	78.89	2.19	2.31	3.80	3.28
58-50-225	13.65	19.51	3.48	4.50	10.83	1.59	73.31	72.56	0.37	0.84	213.90	0.02	36.51	0.77	0.00	0.46
SD of 58-50-225	0.20	0.11	0.32	0.06	3.23	0.05	0.74	7.42	0.04	0.17	2.74	0.02	0.56	0.02	0.00	0.02
%RSD of 58-50-225	1.47	0.55	9.31	1.40	29.80	3.18	1.01	10.22	11.40	20.17	1.28	79.49	1.53	3.06	3.80	3.62
58-50-742	8.67	22.31	0.00	5.49	2.80	1.25	93.93	89.38	0.77	1.12	519.00	0.00	36.04	0.35	0.00	0.50
SD of 58-50-742	0.15	0.18	0.15	0.11	2.99	0.02	1.31	7.05	0.05	0.09	4.51	0.02	0.41	0.03	0.01	0.02
%RSD of 58-50-742	1.74	0.80	105.20	1.98	107.10	1.53	1.39	7.89	5.82	7.84	0.87	552.10	1.13	8.03	70.78	3.20
58-57-3EC	7.08	19.85	0.00	5.32	3.21	1.12	90.04	50.58	0.00	0.96	250.00	0.00	31.89	0.21	0.00	0.52
SD of 58-57-3EC	0.13	0.10	0.24	0.06	2.19	0.02	1.24	8.44	0.03	0.15	2.39	0.01	0.38	0.03	0.00	0.02
%RSD of 58-57-3EC	1.87	0.49	13.99	1.05	68.24	2.07	1.38	16.69	14.00	15.65	0.96	48.01	1.18	12.14	3.36	4.49
58-57-3ES	7.04	21.22	45.88	5.21	2.02	1.29	88.49	53.12	0.20	1.43	2093.00	0.00	33.31	0.56	0.00	0.42
SD of 58-57-3ES	0.08	0.10	1.22	0.06	2.11	0.03	0.77	9.48	0.04	0.08	23.16	0.01	0.51	0.03	0.00	0.02
%RSD of 58-57-3ES	1.11	0.48	2.66	1.20	104.50	2.15	0.87	17.85	18.97	5.42	1.11	16040.00	1.53	5.26	2.69	4.09
58-58-121	6.15	21.01	0.00	4.56	6.66	1.33	62.57	51.66	0.00	1.14	1875.00	0.04	30.15	21.37	0.00	0.49
SD of 58-58-121	0.09	0.39	0.13	0.05	2.16	0.01	0.83	7.55	0.04	0.05	34.80	0.02	0.63	0.43	0.00	0.02
%RSD of 58-58-121	1.47	1.85	7.73	1.07	32.44	0.97	1.32	14.61	21.07	4.61	1.86	45.44	2.09	2.01	2.65	3.75
58-58-410	8.30	16.72	8.85	3.94	4.72	1.32	71.02	53.96	0.98	0.83	222.60	0.00	26.01	0.23	0.00	0.41
SD of 58-58-410	0.11	0.08	0.43	0.06	1.65	0.02	0.52	8.30	0.05	0.06	2.11	0.01	0.53	0.02	0.00	0.02
%RSD of 58-58-410	1.38	0.47	4.80	1.61	34.89	1.16	0.74	15.39	5.51	7.26	0.95	62.08	2.03	8.57	4.77	5.33

Appendix 3

Although only total trihalomethanes are considered in this study, different species within the trihalomethane family were analyzed as summarized on Figure A3.1.

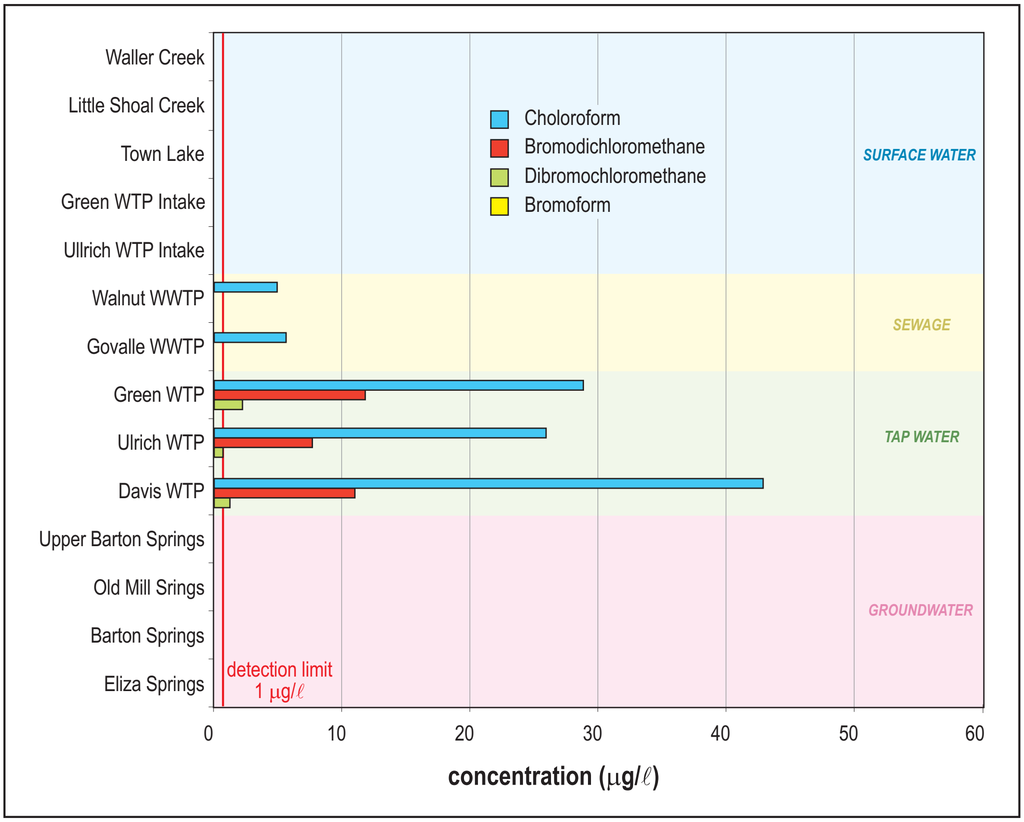


Figure A3.1: Concentrations of the different trihalomethane species in surface water, sewage from two wastewater treatment plants, tap water from three water treatment plants, and groundwater from the Barton Springs system.

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Collecting samples from
an urban well (right)
and from Cold Springs
(below)

Photos by Nico Hauwert
2/25/2001

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