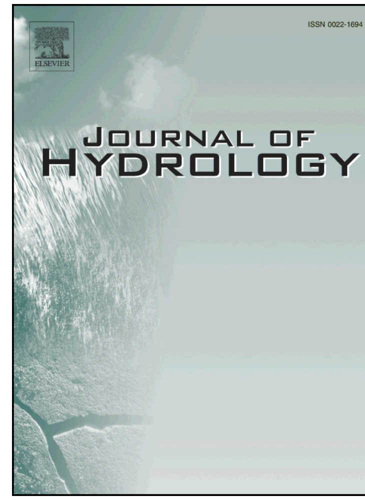

Research papers

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PII: S0022-1694(17)30662-5
DOI: https://doi.org/10.1016/j.jhydrol.2017.09.058
Reference: HYDROL 22278

To appear in: *Journal of Hydrology*

Please cite this article as: Minnig, M., Moeck, C., Radny, D., Schirmer, M., Impact of urbanization on groundwater recharge rates in Dübendorf, Switzerland, *Journal of Hydrology* (2017), doi: https://doi.org/10.1016/j.jhydrol.2017.09.058

Impact of urbanization on groundwater recharge rates in Dübendorf, Switzerland

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Key words

Urban hydrogeology; Groundwater recharge; Urbanization; Surface sealing; Water budget calculation; Switzerland

Abstract

Groundwater, as the world's most important reserve of available fresh water, is known to be affected by urbanization. Managing this resource in a sustainable way is critical for water resource management. Groundwater recharge rates in urban areas remain however still poorly understood and under-researched and knowledge about these rates and their expected changes under increasing urbanization is therefore of primary importance. This study aims to give insight into urban groundwater recharge by performing water budget calculations for four different time periods for an urban study site in northern Switzerland. In order to take into account uncertainty in parameter values a Monte Carlo (MC) approach was carried out.

Our study highlights a strong positive correlation between groundwater recharge rates and the extent of the urban area. In detail, at the study site urban areas expanded from 6% in 1880 to 44% in 2009, leading to an increase in the mean groundwater recharge rate. However, the increase amount in recharge remains uncertain and varies between 29% and 67% depending on the parameter combination originating from the MC approach. Based on our water budget calculations, the transformation of natural landscapes into impervious areas leads to an increase in groundwater recharge rates due to the reduction of evapotranspiration that more than compensates for the increase in runoff. Furthermore, water main leakages contribute to an increase in recharge rates. Overall, we demonstrate that a better understanding of groundwater recharge changes in urban areas is required to move towards a sustainable water management. We hope that this example will encourage the hydrogeological community to pay more attention to urban groundwater recharge.

1. Introduction

More than half of the world's population lives in urban areas, a proportion that is expected to increase in the next decades (McDonald et al., 2014; Schirmer et al., 2013). The contribution from groundwater is vitally important for water supply for most of these urban areas (Moeck et al., 2017; Morris et al., 2007). Indeed groundwater is largely used for food production and drinking water supply in most megacities in the world and has greatly contributed to their development (Drangert and Cronin, 2004; Howard, 2002). As the world population is growing, this resource is probably going to be exploited more intensely (Foster, 2001). Furthermore, land use changes and anthropogenic activities such as surface sealing due to streets and buildings, flood control, forest management and irrigation modify the infiltration and movement of water (Baillieux et al., 2015, Garcia-Fresca, 2007). These modifications often lead to groundwater resources deterioration (Reinstorf et al., 2008; Schirmer et al., 2013; Strauch et al., 2008; Vazquez-Sune et al., 2010).

Urbanization changes sources and flow paths of groundwater recharge compared to the natural state (Grischek et al., 1996; Yang et al., 1999). On a quantitative basis, two urbanization-related processes affect groundwater recharge: i) the increase of impervious surfaces, leading to evapotranspiration reduction and runoff increase (Lerner, 1990; Paulachok, 1991) and ii) the building of water supply and sewer networks, which increase groundwater recharge rates due to leakages (del Campo et al., 2014).

For sustainable water resources management in urban areas knowledge about groundwater recharge rates and expected changes under increasing urbanization is the very basis (Kuroda et al., 2017). However, no consistency about these changes can be found in the literature (Bhaskar et al., 2016).

A few studies indicate that recharge decreases in urban areas as surface sealing prevents infiltration and increases surface water runoff compared to natural landscapes (Grischek et al., 1996; Hardison et al., 2009; Rose and Peters, 2001). In the study of Rose et al. (2001), a notable decline in water levels of urban wells was observed in comparison with non-urban wells for a study area in the vicinity of Atlanta, US. Similarly, Grischek et al. (1996) found a decrease in groundwater recharge of 23% due to surface sealing in Dresden, Germany.

Other studies indicate an increase in groundwater recharge rates due to the reduction in evapotranspiration linked with surface sealing (Appleyard, 1995; Barron et al., 2013; Garcia-Fresca, 2007; Hooker et al., 1999). For instance, Garcia-Fresca (2007) observed for 2000 a groundwater recharge rate almost twice as much as the pre-urban rate in Austin, US, due to the contribution of urban recharge sources such as water main leakages and over-irrigation of e.g. gardens and agriculture areas. Likewise, the study of Barron et al. (2013) indicated that in Perth, Australia, the practice of roof and road runoff infiltration along with the lower evaporative losses caused by the expansion of impervious areas lead to groundwater recharge rates 2 to 3 times higher than in pre-urban conditions.

Overall, all studies mentioned that the net effect of urbanization on groundwater recharge is difficult to predict as every city has different settings and often different climatic conditions.

The objective of this study is to assess the effects of urbanization-driven changes on groundwater recharge rates. These effects were studied for the city of Dübendorf in northern Switzerland. This municipality constitutes a location of choice, as it has undergone a recent and rapid urbanization due to the extension of the neighboring city of Zurich.

A water balance method was applied to calculate groundwater recharge rates for four representative time steps: 1880, 1955, 1980 and 2009. Calculations were performed for each of the 17 watersheds composing the area of Dübendorf. A specific feature of urban areas is the presence of water mains, whose leakages can contribute to groundwater recharge (Leschik et al., 2009b). This feature was explicitly considered in the water balance calculation although exact rates are difficult to estimate. Therefore, uncertainty in parameters such as the percentage of leakage was considered within a Monte Carlo (MC) approach. In the MC approach all possible combinations of uncertain parameters (e.g. runoff and leakages) were taken into account to provide the most reliable water balance calculation within uncertainty ranges.

2. Study area

Dübendorf is located in close vicinity of the city of Zurich in Switzerland (Figure 1). It covers an area of 13.6 km^2 and in 2015 it had a population of 26,700 inhabitants (Statistisches Amt Kanton Zürich, 2016). Dübendorf has a temperate climate with an average temperature of 9.9°C and an average annual rainfall of 1004 mm based on measurements performed between 2006 and 2015 (MeteoSwiss, 2016). Its elevation varies between 430 m and 635 m. The surface is mostly flat in urban areas and even if some relief is present in the forest located in the southern part of the municipality, slopes are not very steep. Most important soil types in the Dübendorf area are brown earth and luvisol. At some local spots gleyic soils can be found (Eidgenössische Forschungsanstalt für Agrarökologie und Landbau FAL, 2013; Kanton Zürich - Amt für Raumentwicklung, 2016b). In these soil types, water can typically easily infiltrate and is readily transmitted through the soil profile (United States - Soil Conservation Service, 1958). Therefore, a homogenous infiltration pattern can be assumed, although locally heterogeneity can lead to different infiltration rates but this can typically not be investigated in detail on catchment scales (Ghasemizade et al., 2015). An unconfined porous aquifer, exploited for drinking water purposes, lies under the northern part of Dübendorf (Kanton Zürich - Amt für Raumentwicklung, 2016a) where the unsaturated zone is typically smaller than 10 m. The municipality is crossed by a network of streams and can be divided into 17 watersheds based on its topography and the hydrology of its surface waters (Swisstopo, 2008) (Figure 2).

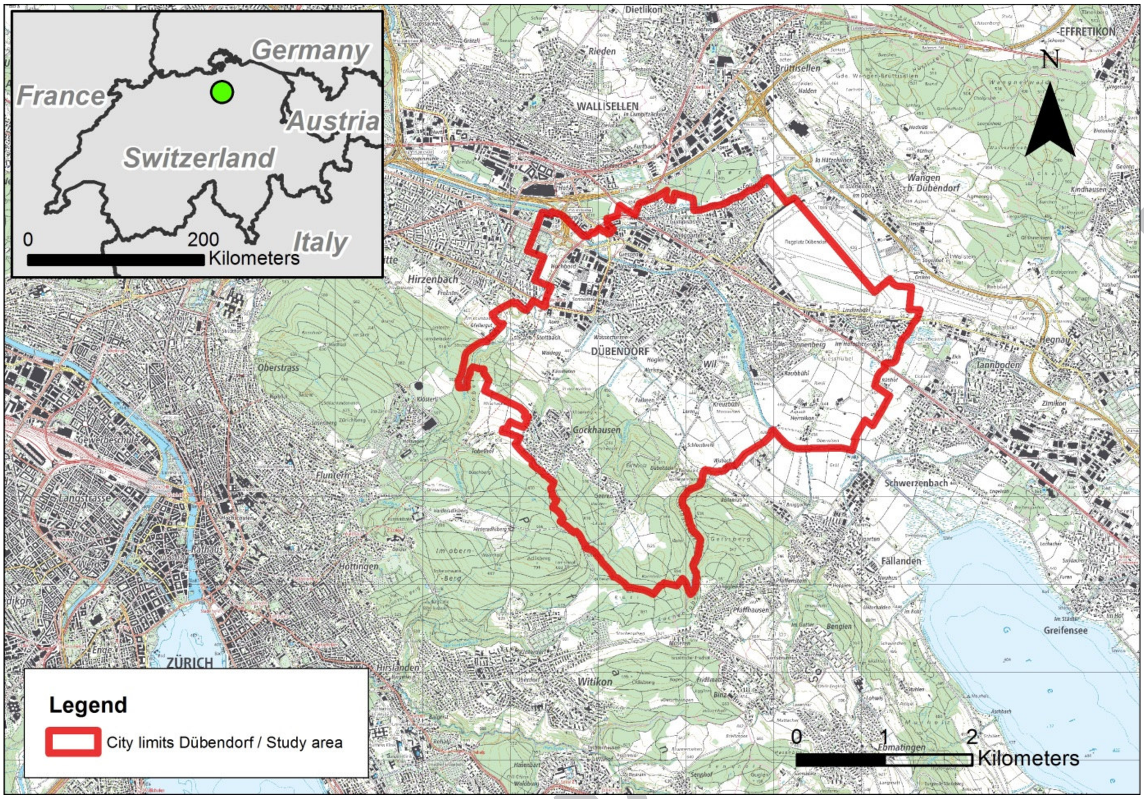

Figure 1 - The municipality of Dübendorf; top left: location of the study area within Switzerland. Base map reproduced with permission of swisstopo / JA100119.

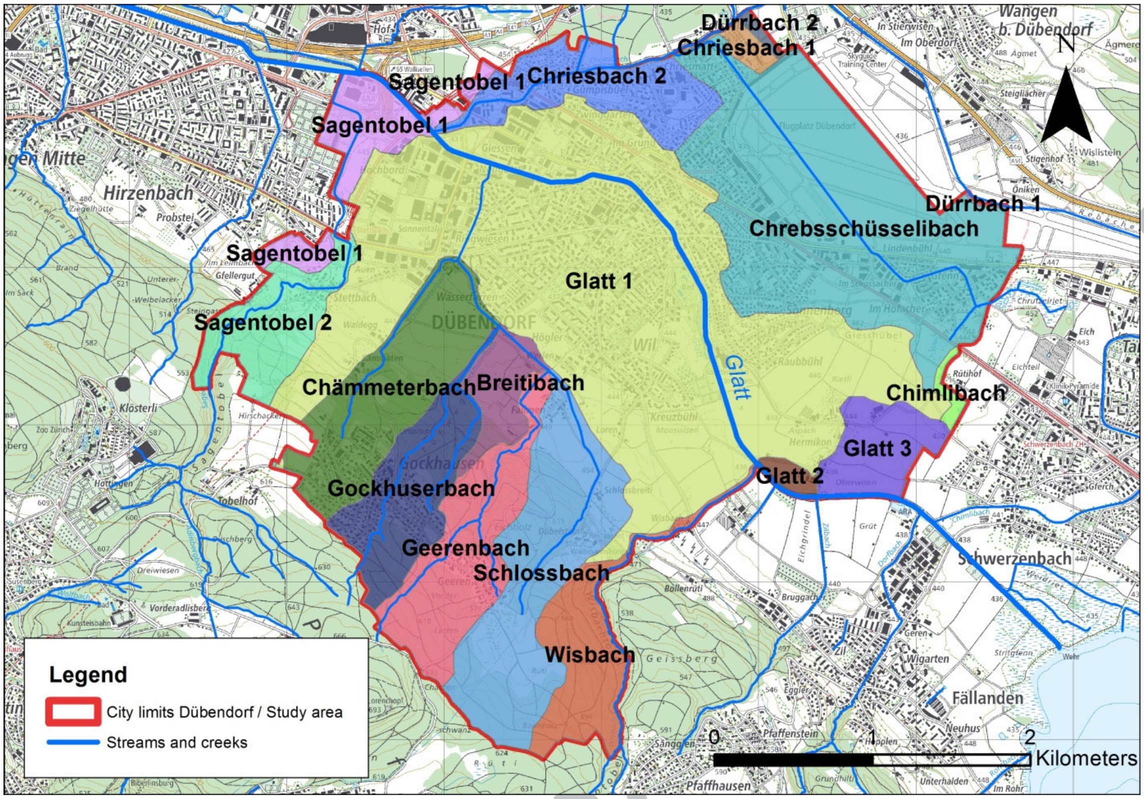

Figure 2 - Division of Dübendorf into 17 watersheds. Base map reproduced with permission of swisstopo / JA100119.

The city structure changed during the last decades from mainly agriculture to an urban setting and can be characterized by four representative time steps: 1880, 1955, 1980 and 2009 (Figure 3). In 1880, agricultural areas were the dominant feature in Dübendorf, covering 70% of its territory (GIS analyses based on the data from Bundesamt für Statistik (BFS) GEOSTAT (2013)). Urban areas accounted only for 6% of the land cover. A rapid urbanization has taken place in the municipality in the second half of the 20th century, due to the extension of the neighbouring city of Zurich. Indeed, between 1955 and 1980, urban areas increased from 23% to 38% of the territory. Its expansion continued since then, but at a slower pace. In 2009, urban areas represented the principal land use class (44%), the remaining land use classes being agricultural areas (34%), forests (20%) and unproductive areas (2%, mainly engineered surface waters with no leakage to groundwater).

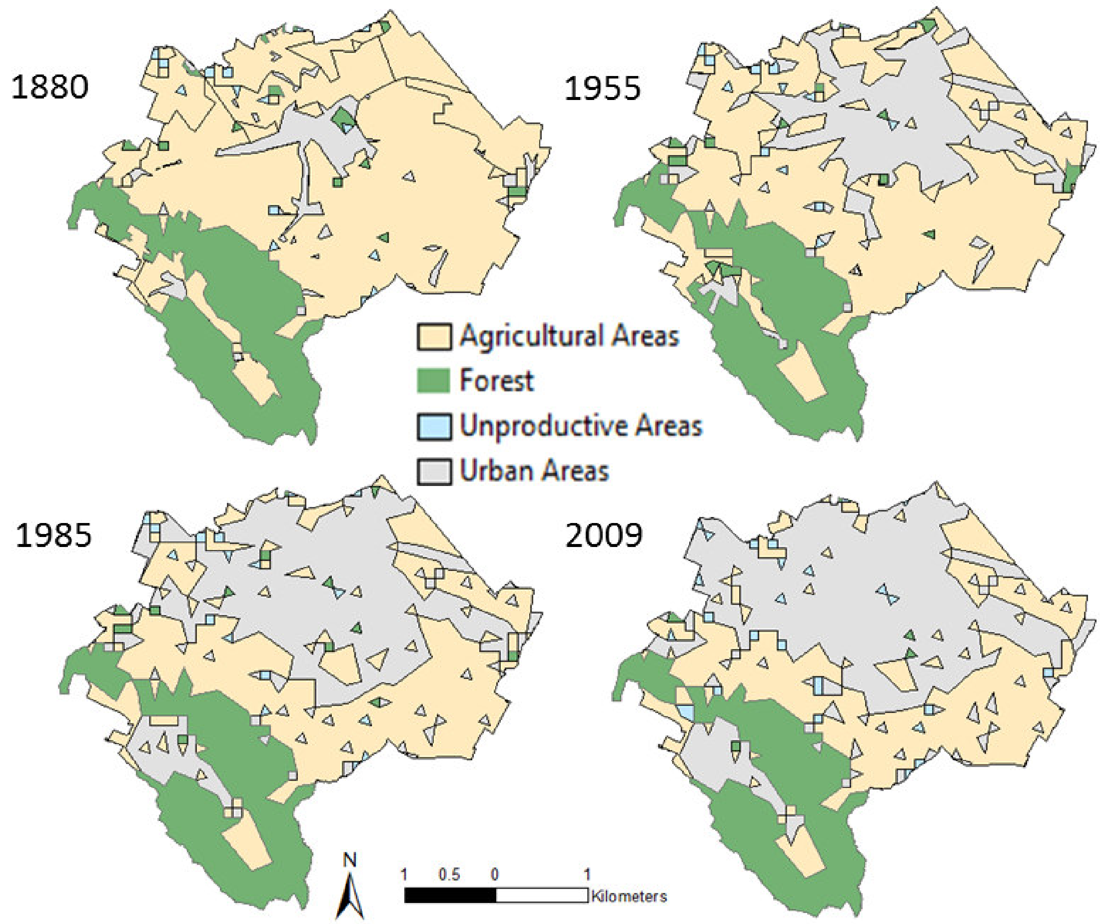

Figure 3 - Land use in Dübendorf in 1880, 1955, 1985 and 2009 (Bundesamt für Statistik (BFS) GEOSTAT, 2013).

Since the building of the first drinking water network in 1894, the water supply increased significantly, reaching its highest point around 1970. Dübendorf water consumption since then decreased slightly even though the population continued to increase, due to a more efficient use (WVD, 2016a). In 2015, drinking water in Dübendorf was composed of ~50% groundwater and of ~50% water imported from Lake Zurich (WVD, 2016b). The first sewer canal in Dübendorf was built in 1920 (Widmer, 2016). The evolution of waste water generation is similar to the one of drinking water, as waste water production depends predominantly on the amount of water consumed.

3. Methods

In the following sections, the applied methods are described. More detailed information can be found in the provided literature and in Minnig (2017).

3.1. Water budget calculation

Reliable groundwater recharge estimation is a difficult task because it is a function of the climatic forcing functions, soil type and land use (Herrmann et al., 2015). Furthermore, groundwater recharge is highly variable in time and space (Moeck et al., 2016a; Newcomer et al., 2014). Although there is a large variety of groundwater recharge estimation methods, their results can differ considerably for individual sites due to the spatio-temporal scales and complexities the methods represent (von Freyberg et al., 2015). Direct field measurements are difficult to obtain (Scanlon et al., 2002) and indirect approaches have therefore to be considered. Furthermore, uncertainties associated with these methods are difficult to assess, as there are no standards to compare with (Huet et al., 2016).

In this study, groundwater recharge rates were estimated by performing water budget calculations for the aforementioned four time periods (1880, 1955, 1980 and 2009). For the calculation the following assumptions were made:

- Storage change in the vadose zone is zero on the annual time scale;
- Groundwater recharge is instantaneous and a homogenous soil is assumed;
- Groundwater catchments and surface water watersheds are comparable in size;
- Subsurface runoff (interflow) is not considered, due to lack of data and of estimation methods. Furthermore, it is expected that subsurface runoff has only a minor effect on the calculated annual groundwater recharge rates due to the flat gradient at the study site.

A specific feature of urban areas is the presence of water mains, whose leakages can significantly contribute to groundwater recharge (Barrett et al., 1999; Leschik et al., 2011; Musolff et al., 2009; Rutsch et al., 2008). Depending on the respective depths of the water mains (water supply system and sewer conduits) and the aquifer, water can either infiltrate or exfiltrate the mains. In general, the water supply system is pressurized and its pressure is in almost all the cases higher than the hydraulic head of the aquifer layer. As a consequence it is very unlikely to have leakages from the groundwater in these pipes. In Dübendorf in general, the sewer system is above the water table, therefore no infiltration of water into the system is expected. Thus, in this study, it is assumed that water always exfiltrates the mains. With these assumptions, groundwater recharge was calculated using Equation (1):

$$GWR = P - ET_a - R_{off} + L \qquad (1)$$

where *GWR* is groundwater recharge (mm/a), *P* is precipitation (mm/a), ET_a is actual evapotranspiration (mm/a), R_{off} is surface runoff (mm/a) and *L* is leakages from waste water and drinking water mains (mm/a). Note that a potential impact from surface waters (either losing or gaining conditions) is negligible and therefore not considered in this study. This assumption is based on the fact that the most relevant surface water, the river Glatt (Figure 2), was channelized end of the 19th century with further constructions along the river banks in the 20th century. Nowadays the Glatt is a degraded straightened channel, has banks with boulders or cement, a clogged riverbed and removed riparian vegetation. Hence it is assumed that groundwater and the surface water are disconnected and the impact of surface water-groundwater interactions on the water balance is negligible. Moreover, as Dübendorf has a temperate climate, only a negligible fraction of the agricultural land is irrigated. Preliminary consideration of irrigation fluxes in the budget calculation showed that the contribution of irrigation is not relevant (Minnig, 2017).

To obtain more detailed results about groundwater recharge rates, calculations were performed for each of the 17 watersheds composing Dübendorf. In order to investigate the change in groundwater recharge induced by urbanization without being influenced by the varying climatic conditions, these ones were set constant for all the time steps considered (2009 was used as reference year).

Note that few groundwater level measurements were also available in Dübendorf for a limited number of years. Using this data for further analysis and validation of the water budget calculations was not possible because of numerous urban influences that affected the water table measurements in Dübendorf. Construction of surface and underground infrastructure, lowering of riverbeds and aquifer exploitation for drinking water purposes resulted in artificial groundwater level fluctuations. Natural fluctuations caused by recharge changes could therefore not be identified. The major challenges for the estimation of recharge rate changes in urban areas are discussed in section 4.4 in more detail.

3.2. Land use categories

For the present study, land use within the 17 watersheds was divided in four main domains: agricultural areas, forests, urban areas and unproductive areas (mainly engineered surface waters with no leakage to groundwater). Provided land use datasets for 1985 and 2009 (Bundesamt für Statistik (BFS) GEOSTAT, 2013) were used for the delineation of the aforementioned areas. For previous time periods, land use categories were digitalized in ArcGIS™ (ESRI®), according to historical maps provided by Swisstopo (Swisstopo, 2017). As urban areas can present very heterogeneous degrees of surface sealing and consequently very different runoff rates, a further sub-division into building areas, gardens and streets was made (Figure 4). The percentage of each subclass was assessed once based on the land use dataset of 2009 and this subdivision was assumed constant throughout the time. The calculations of ET_a and R_{off} for urban areas were therefore made by weighting the values obtained for each of the three subclasses by their respective areas. Similarly, agricultural areas were further divided into cultivated lands and grasslands.

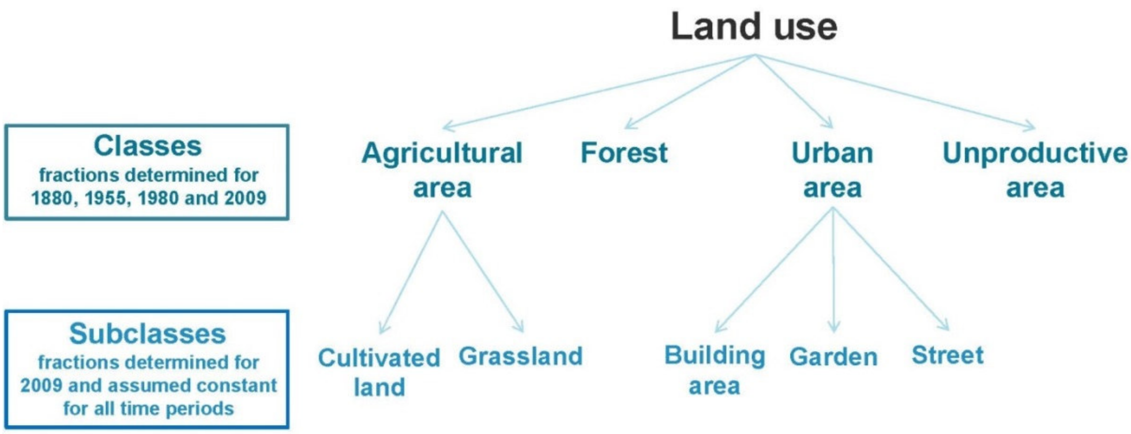

Figure 4 – Land use division into classes and subclasses (Bundesamt für Statistik (BFS) GEOSTAT, 2013).

3.3. Climatic data

The climatic data from the year 2009 (Table 1), measured at the meteorological station of Zurich-Kloten which is in a distance of 8 km from the study area, was used for the water balance calculation. This data was provided by MeteoSwiss (Swiss Federal Office of Meteorology and Climatology).

Table 1 - Average annual climatic condition in Dübendorf in 2009 (MeteoSwiss, 2016).

Climatic data	Value
Precipitation [mm/a]	947
Temperature [°C]	9.9
Solar irradiation [J/cm^2]	1222
Relative Humidity [%]	77.8

3.4. Actual evapotranspiration

Potential evaporation (ET_{pot}) was calculated based on the Turc formula. Subsequently, actual evapotranspiration (ET_a) can be determined for land surfaces using the Bagrov method (Equation 2) (DVWK, 1996). Indeed, this method allows ET_a assessment for different land use types, including urban areas, by calculating annual ET_a as a coefficient of ET_{pot}.

$$ET_a = 1.1 \cdot ET_{pot} \cdot y \qquad (2)$$

where y is the ratio between ET_a and ET_{pot}. This ratio is determined according to soil type, land use, rate of capillary rise (*CR*) (depending on the vegetation; values taken from DVWK 1996), and the number of days with *CR* (values taken from DVWK 1996). Note, when the Turc formula is used in the Bagrov method the evapotranspiration should be corrected by a factor of 1.1. For water surfaces, the Dalton law was used (Equation 3) (DVWK, 1996). This formula links ET_a with wind speed, temperature of the water surface and relative humidity.

$$ET_{a,water} = f(v) \cdot e_s(T_{w0}) \cdot (1 - \frac{RH}{100}) \qquad (3)$$

Where $e_s(T_{wo})$ is the saturation vapor pressure at water surface temperature [hPa], *RH* is the average daily relative humidity [%] and $f(v)$ is the wind function, given in Equation (4):

$$f(v) = a + b \cdot v^c \qquad (4)$$

where v is the average daily wind velocity [m/s] and *a, b, c* are determination coefficients. Different coefficients have been determined according to the meteorological and physical conditions of the place. In the present case, coefficients given in the WMO-guideline were used (DVWK, 1996), determined for the Rhine (German part), which is in closes vicinity of the study area. Note that all ET_a calculations were carried out on daily time scales for each land use type and subsequently aggregated to obtain annual values.

3.5. Runoff

Runoff (R_{off}) was assessed using the Soil Conservation Service Curve Number (SCS-CN) method (Equation 5, 6 and 7) (United States - Soil Conservation Service, 1958). With this method, a Curve Number (CN) is attributed to each land surface (Table 2), quantifying its runoff potential according to its soil type, land use and hydrological condition. The CN ranges from 30 to 100 and the runoff potential increases with increasing CN. The quantity of water running off the surface can then be determined based on the CN and on the daily precipitation (Eshtawi et al., 2016). R_{off} calculations were performed on a daily basis for each land use type using the meteorological data from 2009 and results were subsequently aggregated to obtain annual values. Following the U.S. Soil Conservation Service (1958), soil types in Dübendorf are fitting the hydrological soil group B.

$$R_{off} = \begin{cases} 0 \; for \; P \leq I_a \\ \frac{(P-I_a)^2}{(P-I_a+S)} \; for \; P > I_a \end{cases} \quad (5)$$

$$S = \frac{1000}{CN} - 10 \quad (6)$$

$$I_a = 0.05 \cdot S \quad (7)$$

where R_{off} is Runoff [in], P is daily rainfall [in], S is potential maximal retention [in], CN is the dimensionless curve number indicating the runoff potential of each surface and I_a is the initial abstraction [in].

Table 2 - CN values chosen for the different land use classes and subclasses (United States - Soil Conservation Service, 1958).

Land use class	Subclass	CN [-]
Agricultural areas	Cultivated land	70
	Grassland	69
Forests	Forests	60
Urban areas	Building areas	92
	Garden	61
	Streets	98

3.6. Leakages

Water leakages can contribute significantly to urban groundwater recharge (Musolff et al., 2010b; Schirmer et al., 2011). However, despite intensive research made on this subject, the magnitude of main exfiltration is still uncertain, especially from sewer conduits and only rough estimations can be made (Rutsch et al., 2008). In the present study, it was assumed that pipes are always losing water (leakage) to groundwater due to the hydraulic head between the pipes and the groundwater table elevation. For water supply systems indeed, leakages from the groundwater to the water mains are very unlikely due to the fact that the pipes are pressurized and that their hydraulic heads are therefore in almost all cases higher than the groundwater table elevation. Leakages were assumed to be proportional to the quantity of water passing through water mains. Their rates were provided directly by the drinking water provider as well as by the waste water facility (personal communication). In Dübendorf, leakages account for 7% for water supply systems and 10% for sewer conduits (Widmer, 2016; WVD, 2016b). These values seem consistent with the ones found in literature. For instance, Wakida and Lerner (2005) have identified an exfiltration rate of 13% (both waste water and drinking water leakages) in Nottingham, UK, while Rutsch et al. (2006) estimated waste water leakages to be between 5 and 20% of the waste water dry weather flow using tracer experiment in different European cities. Also Musolff et al. (2010a) found similar values of 9.9-13.0% loss of waste water dry weather flow to groundwater in Leipzig, Germany. Furthermore, Ellis et al. (2003) determined a rate of 5-10% of dry-weather flow based on laboratory experiments. In the present study, leakages are evenly distributed over the whole urban area and their rates were determined for each time period.

3.7. Uncertainty and sensitivity analysis

In calculating groundwater recharge, uncertainties are omnipresent and difficult to quantify. Uncertainties arise from measurement errors, simplifying assumptions in the chosen method and unavailability of detailed data (Healy, 2010; Jiménez-Martínez et al., 2010). By calculating the water budget to estimate recharge rates, the result accuracy depends directly on the uncertainties of each of the components used in the calculation. For example, as ET_a, which represents a significant part of the water budget, is bound to large uncertainties, these uncertainties also affect groundwater recharge rates (Huet et al., 2016).

In order to take into account uncertainty in the parameter values such as leakages or ET_a we explored the parameter space using a Monte Carlo (MC) approach. Our results are therefore not only based on one specific value, rather we obtain recharge rates including different parameter combinations. For uncertain parameters (Table 3) we generated 100,000 random parameter values within a realistic parameter range and calculated the corresponding flux. The parameter range for drinking water leakages (*DWL*) as well as waste water leakages (*WWL*) is based on the study of Rutsch et al. (2006). They estimated leakages up to 20%. The *CN* number range shows a value between ±10 from the reported value for the considered land use class (Mishra and Singh, 2003). The range for the rate of *CR* as well as the number of days with capillary rise are within the provided range based on DVWK (1996).

The rate of capillary rise (*CR*) and the number of days with capillary rise affect the ET_a rates. Variation in the *CN* number will lead to variation in R_{off} rates and variability in the *DWL* as well as *WWL* will affect the leakage rate for the four different time periods considered. In our applied MC approach different combinations of the parameters were possible which led to a GWR range for the four different time periods considered. We considered the entire parameter range as the most simplistic but also robust description for different hydrological processes and consequently recharge fluxes. We used the MC uncertainty analysis for the watershed Glatt 1, the largest watershed in Dübendorf as an example for the remaining watersheds. The results and drawbacks from these calculations for the watershed Glatt 1 are transferable to all other watersheds from this study.

Table 3 - Generated parameter range for land use classes with different subclasses for rate of capillary rise (CR), number of days with capillary rise (DVWK 1996), CN number (Mishra and Singh, 2003), drinking water leakages (DWL) as well as waste water leakages (WWL) (Rutsch et al., 2006).

Land use classes	Subclasses	CR [mm/d]	N of days with capillary rise [d]	CN Number [-]	DWL [%]	WWL [%]
Agricultural Area	Cultivated land	0.3 - 3.5	40 - 80	60 - 80	2 - 20	2- 20
	Grassland	0.3 - 3.5	100 - 120	59 - 79		
Forest	Forest	0.3 - 3.5	100 - 120	50 - 70		
Urban Area	Building area	0.3 - 3.5	0	82 - 98		
	Garden	0.3 - 3.5	100 - 120	51 - 71		
	Street	0.3 - 3.5	0	82 - 98		

4. Results and Discussion

In the following section ET_a, R_{off} and leakages calculated for each land use type are presented and discussed (Table 4). Then the groundwater recharge rate of each watershed is provided and compared with the extent of its urban area, knowing that the degree of sealing inside urban areas is assumed to be constant (Figure 5). Additionally, the water budget for the whole municipality of Dübendorf is shown for the four time steps of consideration. Subsequently, obtained results will be discussed including the uncertainty analysis of the estimated fluxes. Finally, a comparison of our findings with the results from other studies is performed.

4.1. ET_a, R_{off} and leakages assessment

As expected, ET_a is much higher in forests, in agricultural areas and gardens than in impermeable urban areas (Table 4). The impermeability of the surface prevents evapotranspiration and increases R_{off}. Furthermore, our result highlights that it is crucial to account for the presence of gardens in urban areas, as they have an ET_a rate much higher than impervious surfaces. Neglecting these features can lead to biased groundwater recharge rate estimations.

On the contrary to ET_a, R_{off} is high in urban areas as expected, especially for streets, and it is low in forests and agricultural areas. Indeed, as soils in these natural areas have good water infiltration capacity, precipitation must exceed a relatively high threshold before runoff is triggered, whereas for urban areas, this threshold is much lower as surface imperviousness prevents the water to infiltrate into the soil (Tashie et al., 2016). For instance, in the year 2009 184 rainy days occurred in Dübendorf but only 20 (11%) have led to runoff in forests (CN=60), 42 (23%) for cultivated lands (CN=70), 96 (52%) for building areas (CN=92) and 149 (81%) for streets (CN=98). Especially during storm events, surface runoff is dramatically increased in case of impervious cover (Tashie et al., 2016; The Federal Interagency Stream Restoration Working Group, 1998). In accordance to our results, the study by Schueler (1994) shows that urbanization can increase annual volume of storm water runoff by 2 to 16 times compared to predevelopment conditions, depending on the extent of the impervious surface. Note that a limitation of the SCS-CN method lies in the fact that it does not take temporal distribution of rainfall and its intensity into consideration (United States - Soil Conservation Service, 1958).

For the considered four time periods leakage rate increased from 1880 to 2009. In 1880, no water mains existed in Dübendorf and therefore leakage is zero. The leakage rate increased to 70.4 mm/a in 1955 and reached a maximum in 1980 with 121.7 mm/a. In 1980 drinking water consumption and consequently the waste water production were higher than today (2009). Indeed, in 2009 the leakage rate decreased to 94.7 mm/a, which is however still higher than calculated for the time period 1955.

Table 4 – Deterministic assessment of ET_a, R_{off} and leakages for each land use class.

	% of class	ET_a [mm/a]	R_{off} [mm/a]	Leakages [mm/a]			
				1880	1955	1980	2009
Agricultural areas	**100**	**678.5**	**16.1**	**0**	**0**	**0**	**0**
Cultivated land	24.0	672.4	17.4	0	0	0	0
Grassland	76.0	680.4	15.7	0	0	0	0
Forests	**100**	**776.4**	**6.2**	**0**	**0**	**0**	**0**
Urban areas	**100**	**252.5**	**257.6**	**0**	**70.4**	**121.7**	**94.7**
Building areas	68.2	216.1	163.8	0	70.4	121.7	94.7
Gardens	7	736.4	7.0	0	70.4	121.7	94.7
Streets	24.8	216.1	586.4	0	70.4	121.7	94.7
Unproductive areas	**100**	**461.8**	**0.00**	**0**	**0**	**0**	**0**

4.2. Groundwater recharge assessment

In Figure 5 groundwater recharge rates and percentages of urban area are shown for each of the 17 watersheds for the time steps 1880, 1955, 1980 and 2009. As the degree and time of urbanization are varying strongly from one watershed to another, the presented detailed assessment allows a better understanding of the relationship between the extent of urban area and groundwater recharge changes. A strong positive linear relationship can be observed between groundwater recharge and the extent of urban areas, assuming homogeneous soil conditions. Watersheds that are highly urbanized (mostly in the north-west part of Dübendorf, such as Chriesbach 1, Chriesbach 2 and Sagentobel 1) exhibit significantly higher groundwater recharge than watersheds that remain predominantly rural (mostly in the southern part of Dübendorf, such as Wisbach, Schlossbach and Geerenbach). The correlation coefficient between groundwater recharge and the percentage of urban area for the four considered time steps is 0.97, which indicates a strong linear relationship. A similar correlation was observed by Eshtawi (2016) between percolation and the relative change in urban area in a study area of the Gaza Strip, Palestinian territories.

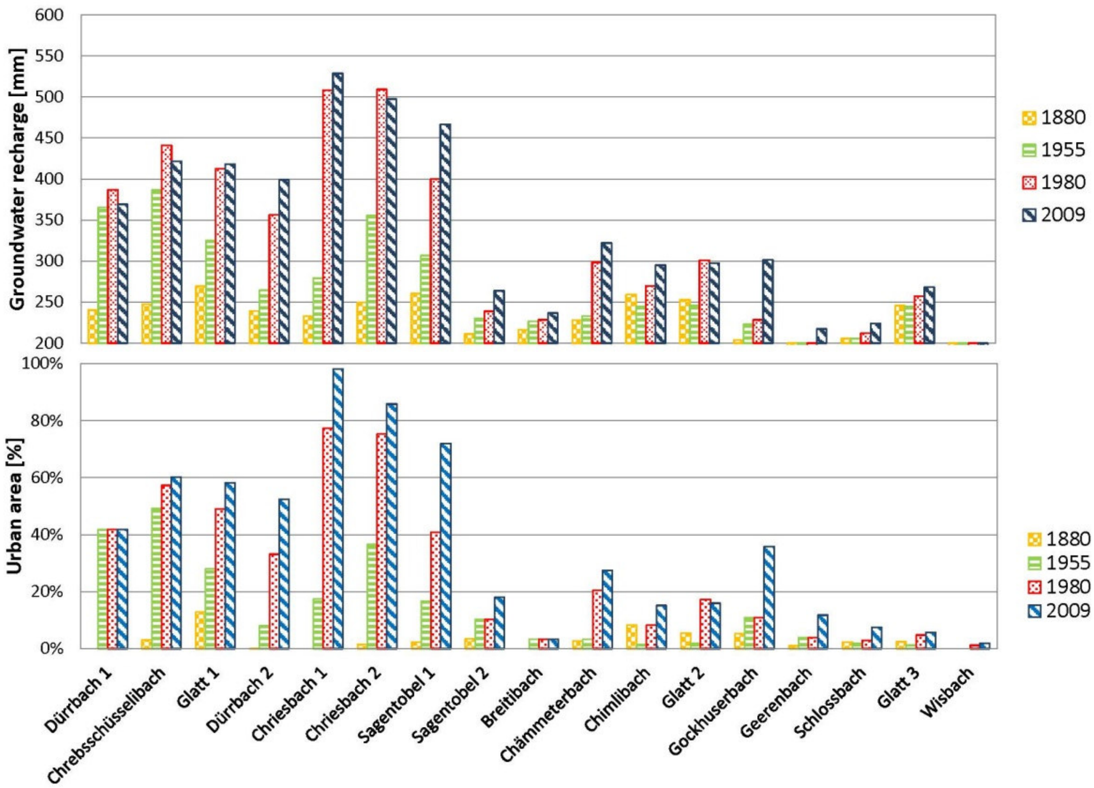

Figure 5 – Deterministic groundwater recharge rates and percentage of urban area for each of the 17 watersheds for the time steps 1880, 1955, 1980 and 2009. For all time steps, the built surface (buildings and streets) is assumed to account for 94% of the urban area.

In the following, these results are aggregated for the whole municipality of Dübendorf to demonstrate the change in the water balance components for the four considered time periods. By increasing the percentage of impervious area from 6% in 1880 to 44% in 2009, a significant decrease in ET_a (from 676.8 mm/a in 1880 to 494.1 mm/a in 2009) can be observed. This decrease is significantly higher than the increase in R_{off} (from 29.1 mm/a in

1880 to 121 mm/a in 2009). These changes, combined with the additional water leaking from both the drinking water and waste water mains (from 0 mm/a in 1880 to 42 mm/a in 2009), led to a substantial increase in groundwater recharge, from 241.5 mm/a in 1880 to 374.2 mm/a in 2009, which represents a 55% increase. On average an increase of 1.0 mm/a occurred between 1880 and 2009. The highest increase in groundwater recharge took place between 1955 and 1980 (2.4 mm/a). For comparison, the annual groundwater recharge increase was only 0.7 mm/a for the time period 1880 to 1955 and 0.7 mm/a between 1980 and 2009.

In a few watersheds, however, groundwater recharge rates slightly decreased from 1980 to 2009 (Dürrbach 1 and Chrebsschüsselibach), even though the percentage of urban area experienced a slight increase. This phenomenon can be explained by the higher leakage rate in 1980. Leakage rate in 2009 is slightly lower than in 1980 due to a more efficient water resource management and as the extent of urban area did not change significantly between 1980 and 2009 in these catchments, this reduction in leakages is leading to a small decrease in recharge rates. Considering that groundwater recharge has increased by 55% from 1880 to 2009 whereas leakages were constituting at the maximum 11% of groundwater recharge, it can be concluded that the principal factor influencing groundwater recharge is the increase in urban area and the subsequent decrease in evapotranspiration. Even though leakages have also an effect on groundwater recharge rates, their contribution is minor in the case of Dübendorf compared to the additional infiltration induced by the strongly reduced evapotranspiration of impervious areas. Note that an increase in groundwater recharge does, however, not mean that groundwater availability in urban areas increased as well. Indeed, the increase in groundwater exploitation due to urbanization and population growth can be higher than the surplus of groundwater recharge (Howard, 2002). Furthermore, the presence of contaminants has often been recorded in urban areas and high concentration load of organic micropollutants can be typically found (Leschik et al., 2009a; Moeck et al., 2016b; Schirmer and Schirmer, 2008). It can be stated that, by converting agricultural areas and forests into urban areas, the reduction in ET_a is more significant than the increase in R_{off}, leading to an increase in groundwater recharge. A conceptual model for the four time periods summarizes the results and shows the changes in the corresponding fluxes of the water balance (

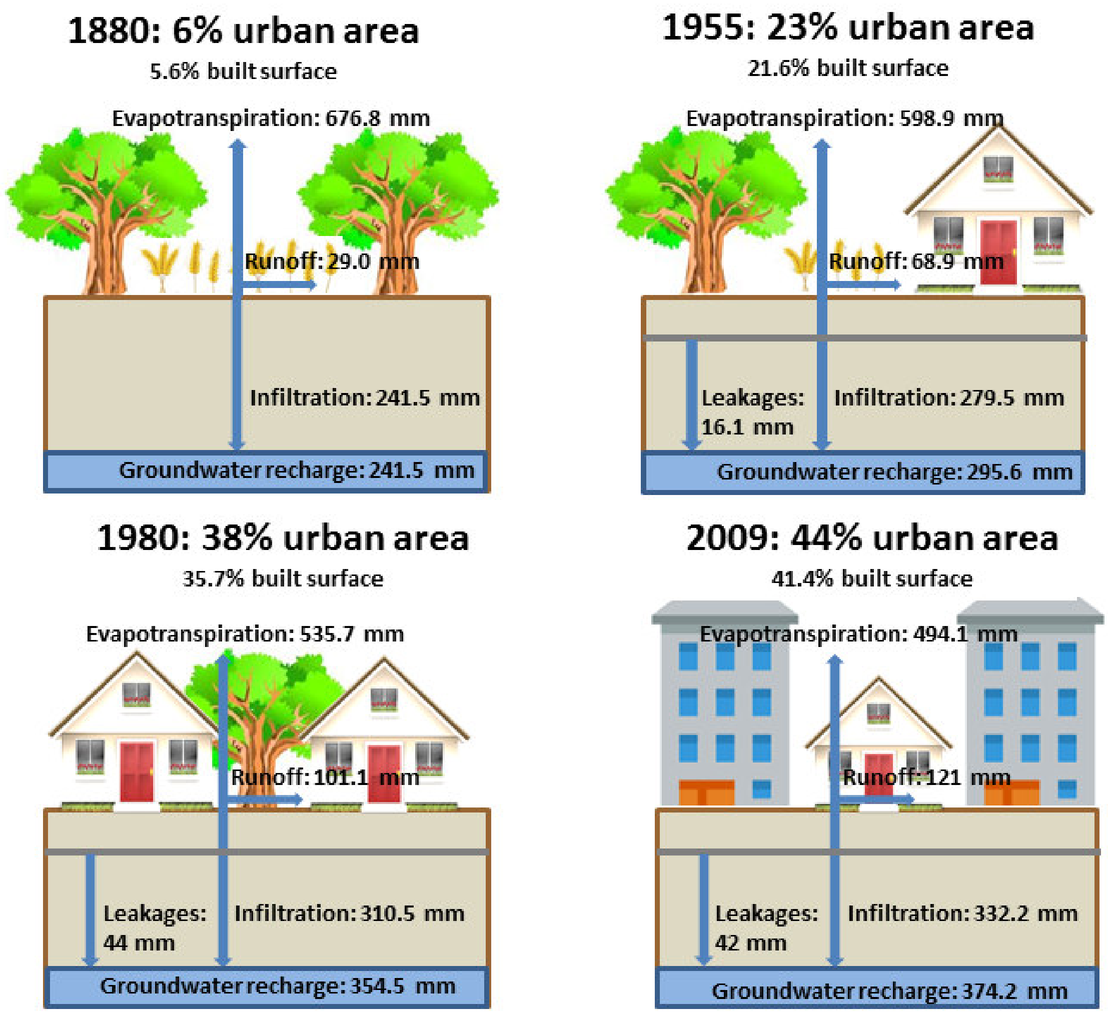

Figure **6**).

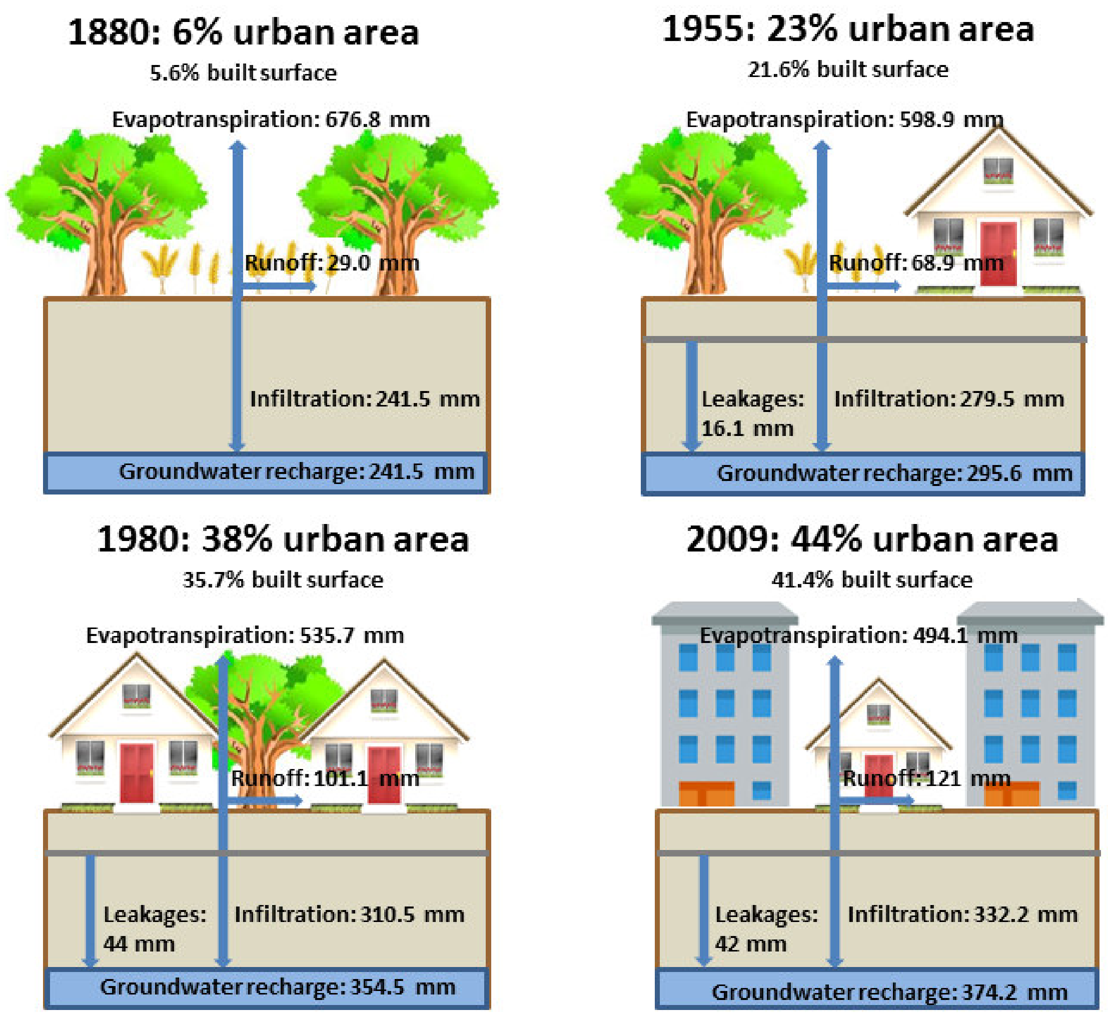

Figure 6 - Actual evapotranspiration, runoff, leakages and groundwater recharge for the four years of investigation (1880, 1955, 1980 and 2009) for the whole Dübendorf municipality.

4.3. Uncertainty analysis

Table 5 shows the calculated groundwater recharge rates for the four investigated time periods. The mean recharge rate is 265.7 mm/a in 1880, 337.8 mm/a in 1955, 457.1 mm/a in 1980 and 483.1 mm/a in 2009. This amounts to an increase of groundwater recharge of 217.4 mm/a between 1880 and 2009. If, however, only the minimum change in recharge is considered, which can be seen as the most conservative assumption, it can be shown that recharge increase is still 48.5 mm/a between 1880 and 2009. This shows how uncertain the recharge estimation in urban areas is and that results should not be based on one specific value, rather different parameter combinations should be included to take uncertainty into account.

Table 5 – Groundwater recharge rates with descriptive statistics for the considered four time periods

	Groundwater recharge [mm/a]			
	1880	**1955**	**1980**	**2009**
Mean	265.7	337.8	457.1	483.1
Median	266.2	344.7	457.1	502.4
1st Quartile	248.3	318.1	426.4	451.9
3rd Quartile	285.0	364.7	502.0	533.8
Min	163.7	178.2	210.1	212.2
Max	331.9	433.6	606.1	632.6

In Figure 7 the groundwater recharge rates for the four time periods are shown. It can be observed that with increasing time groundwater recharge increases. Apart from the recharge rates, the uncertainty range (e.g. Inter Quartile Range IQR, Figure 7) also increases. This is due to increasing water use associated with more uncertain leakage parameters. Uncertainty in the parameters expressing drinking water leakages (DWL) and waste water leakages (WWL) affects recharge rates less for earlier time periods because the percentage of urban area is smaller.

Overall, the results are in good agreement although it can be noticed that the mean groundwater recharge rates are slightly higher for the period 1980 and 2009 within the MC analysis compared to the deterministic calculation. This result highlights the need to take uncertainty in water budget calculation into account. However, even if the obtained values are affected by uncertainties due to rough estimations and measurement errors, the groundwater recharge increase linked with urbanization is incontestable.

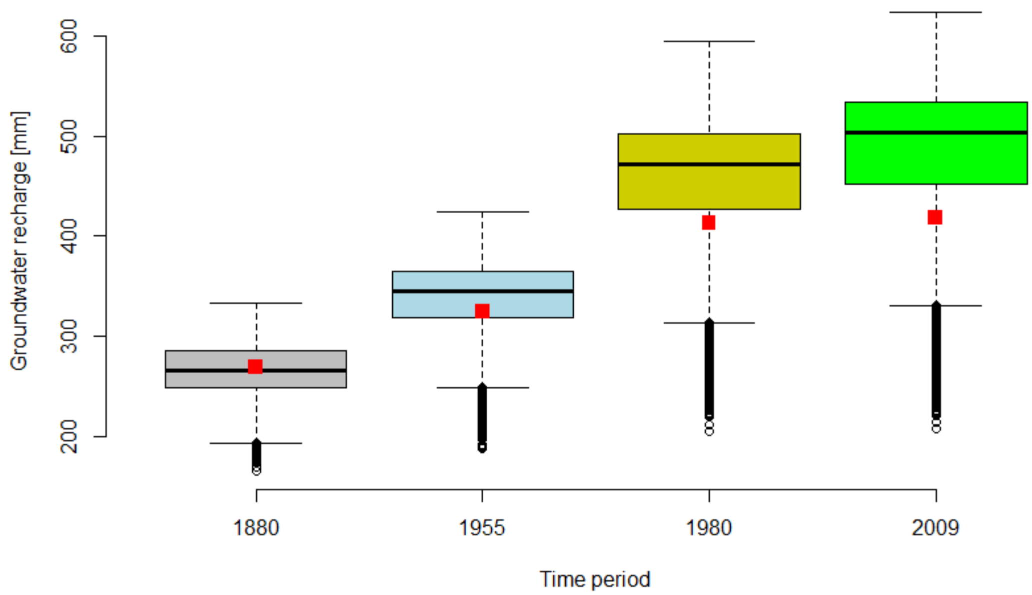

Figure 7 – Groundwater recharge rates for the four years of investigation (1880, 1955, 1980 and 2009) for the watershed Glatt 1. The boxplot indicates the uncertainty range where the filled box shows the 25th to 75th percentile of the recharge data (Inter Quartile Range IQR) and the black line within the IQR displays the mean value. Outliers are shown as single points whereas the ranges of the lower and upper whisker are indicated with dashed lines. The red filled rectangle shows the recharge rate from the calculation without the uncertainty analysis.

4.4. Comparison with other case studies

The validation of calculated groundwater recharge changes is challenging and is certainly associated with large uncertainty as demonstrated. Various artificial effects in an urban setting make a systematic evaluation often impossible (Vazquez-Sune et al., 2010). Even if groundwater measurements are available as time series which cover current and historical periods, it does not guaranty that increasing recharge rates are identified. Typically, increasing industrial and drinking water extraction exceed or buffer the surplus of groundwater recharge (Jackson et al., 2001). Alternatively, changes in surface water discharge, especially in the baseflow might indicate changes in recharge rates (Rutledge and Daniel, 1994, von Freyberg et al., 2015). However, in urban settings construction of surface and underground infrastructure, lowering of riverbeds as well as banks with boulders, cement and removed riparian vegetation can lead to a wide range of changes in surface water discharge which cannot be associated with groundwater recharge changes. A promising method to indirectly identify changes in recharge is certainly the application of remote sensing method such as GRACE (Rodell et al., 2009, Scanlon et al., 2012, Shamsudduha et al., 2012). However, the spatial resolution and technical restrictions still hamper a systematical differentiation of changes in water storage between natural landscapes and urban settings. Moreover, remote data are typically just available for more current time periods where changes in recharge are small compared to historical pre-urban conditions. For instance, for our study site the mean recharge change between 1980 and 2009 is only 19 mm/a. Therefore, we use analog studies of recharge estimation in urban areas.

There is only a limited number of studies on urban groundwater recharge (Lerner, 2002). Undoubtedly, a direct comparison might be limited due to, amongst other, different climatic conditions, urban area development and water operator strategies. However, the provided values from urban groundwater recharge studies are generally exhibiting the same trend as the ones calculated in our research and can be used to assess the plausibility of our results.

For instance, Hooker et al. (1999) observed an increase of about 41% in groundwater recharge rate compared to the pre-urban rate for Wolverhampton, UK. In our study, a groundwater recharge rate increase ranging between 29% and 67% was found. The increase in groundwater recharge in Dübendorf is therefore in the same order of magnitude than the one obtained by Hooker et al. (1999).

Similarly, Appleyard (1995) estimated that recharge accounts for 15-25% of the annual average rainfall of 860 mm in non-urban areas and for 37% in residential areas in Perth, Australia. In the present study, recharge (deterministic calculation) was accounting on average for 26% of the annual precipitation (P = 947 mm) in 1880, when Dübendorf was only slightly urbanized (6% urban area) and 40% in 2009 (44% urban area).

Knowing that recharge varies significantly according to urban density, climatic conditions, land use and topographic settings, the obtained values in our study seem coherent with those found in the literature.

5. Summary and conclusions

Detailed knowledge about groundwater recharge is essential for sustainable water resource management, especially for urban areas where groundwater resources are under increasing pressure. Despite an increasing awareness about the need for a better understanding of urban groundwater recharge rates, only a few studies on this topic have been performed. The difficulty in estimating groundwater recharge rates lies in its high spatial and temporal variability, which makes direct measurements on an urban scale impossible. Compared to natural landscapes, urban areas exhibit new sources and more complex recharge pathways. Indirect approaches such as water budgets have thus to be used, although they present large uncertainties. In this study, an almost linear positive relationship between the percentage of urban area and groundwater recharge rates was observed. For the study site, groundwater recharge has increased within a range of 29% to 67% from 1880 to 2009. This increase is mostly due to the reduction in evapotranspiration that more than compensates the increase in runoff produced by the replacement of natural areas (agricultural areas and forests) by impervious surfaces. Furthermore, water main and sewer leakage contributed to an increase in groundwater recharge from 1880 to 2009.

Our results highlight that a better understanding of groundwater recharge response to urbanization is a prerequisite to improve the establishment of adequate institutional frameworks, which warrant sustainable groundwater resources management. Indeed, detailed understanding of urban groundwater recharge and its composition is beneficial in order to insure good groundwater quality and drinking water supply in urban areas, for a better assessment of flood events and for a wise urban infrastructure development. We hope that our example will encourage the hydrogeological community to pay more attention to urban groundwater recharge.

6. Acknowledgment

This project has received funding from the European Union's Horizon 2020 research and innovation programme under the Marie Skłodowska-Curie grant agreement No 675120 (project INSPIRATION). This study was further supported by the Competence Centre Environment and Sustainability (CCES) of the ETH domain in the framework of the RECORD Catchment project (Coupled Ecological, Hydrological and Social Dynamics in Restored and Channelized Corridors of a River at the Catchment Scale). This work was furthermore supported by the ZBL-Foundation (Foundation for the Promotion of the "Scientific School Zunker-Busch-Luckner"). We thank Ansel Schirmer (University of Geneva) for his comprehensive literature review and many helpful discussions. We also highly appreciate the useful comments of the two anonymous reviewers and Radu Gogu as well as the associated editor Brian Smerdon and editor Corrado Corradini.

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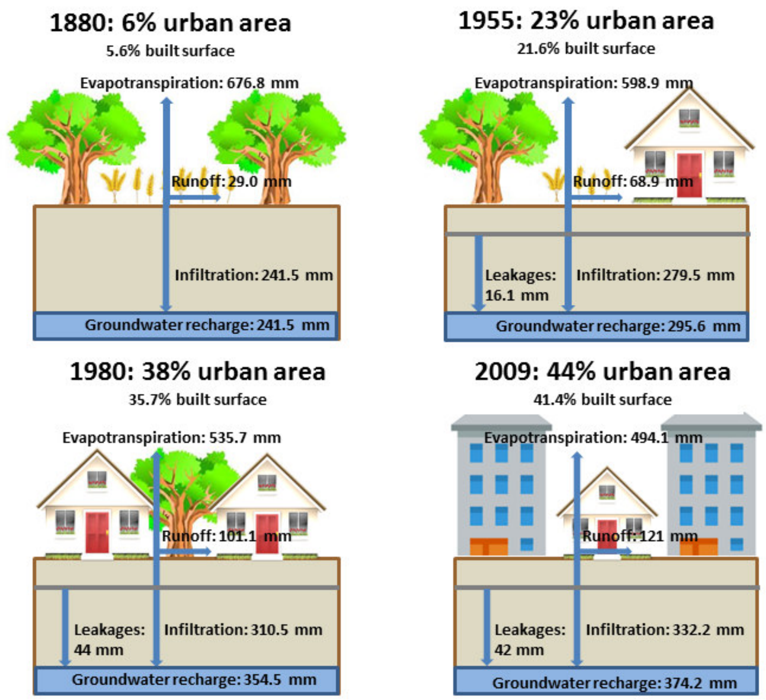
1880: 6% urban area
5.6% built surface
Evapotranspiration: 676.8 mm
Runoff: 29.0 mm
Infiltration: 241.5 mm
Groundwater recharge: 241.5 mm
1955: 23% urban area
21.6% built surface
Evapotranspiration: 598.9 mm
Runoff: 68.9 mm
Leakages: 16.1 mm
Infiltration: 279.5 mm
Groundwater recharge: 295.6 mm
1980: 38% urban area
35.7% built surface
Evapotranspiration: 535.7 mm
Runoff: 101.1 mm
Leakages: 44 mm
Infiltration: 310.5 mm
Groundwater recharge: 354.5 mm
2009: 44% urban area
41.4% built surface
Evapotranspiration: 494.1 mm
Runoff: 121 mm
Leakages: 42 mm
Infiltration: 332.2 mm
Groundwater recharge: 374.2 mm

Highlights

- Urban groundwater recharge rate assessment through water budget calculations
- Urbanization leads to a reduction of evapotranspiration
- Water leakages contribute to recharge rates increase (up to 10% at the study site)
- Increasing groundwater recharge rates due to urbanization at Dübendorf, Switzerland