Urban water storage capacity inferred from observed evapotranspiration recession

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Key Points:

- A new method is proposed to infer urban water storage capacity from evapotranspiration recession.
- Rapid decline of evapotranspiration after rainfall at all sites reflects strong water-limitation.
- Water storage capacity in cities is an order of magnitude smaller than in rural and forested areas.

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Abstract

Water storage plays an important role in mitigating heat and flooding in urban areas. Assessment of the capacity of cities to store water remains challenging due to the extreme heterogeneity of the urban surface. Traditionally, effective storage has been estimated from runoff. Here, we present a novel approach to estimate water storage capacity from recession rates of evaporation during precipitation-free periods. We test this approach for cities at neighborhood scale with eddy-covariance latent heat flux observations from thirteen contrasting sites with different local climate zones, vegetation cover and characteristics, and climates. We find effective water storage capacities to vary between 1.5 and 20 mm corresponding to e-folding timescales of 2.5 to 12 days. According to our results, urban water storage capacity is at least one order of magnitude smaller than the observed values for natural ecosystems, resulting in an evaporation regime characterised by extreme water limitation.

Plain Language Summary

Urban water storage plays an important role in mitigating urban flooding and affects urban heat via cooling through evapotranspiration. Determining the amount of water that can be stored in a city remains challenging due to the variability in urban landscapes. The methodology presented estimates this water storage based on how evapotranspiration declines over time during periods without precipitation. The estimated storage capacities amount to 1.5–20 mm, which is an order of magnitude smaller than in natural ecosystems.

1 Introduction

Cities face weather-related risks magnified by climate change, such as heatwaves and flooding (Wilby, 2007). At the same time, urbanization is expected to further increase the already large share of the world population living in those cities to 68% in 2050 (United Nations, 2018). As for heatwaves, global temperatures are projected to increase, with the high temperatures exacerbated in urban areas where air temperatures are typically higher than in the rural surroundings due to the Urban Heat Island effect (UHI) (Oke, 1982; Santamouris, 2014; Oke et al., 2017). These high temperatures in cities lead to increased energy demands, health issues and excess mortality (Stone Jr & Rodgers, 2001; Gabriel & Endlicher, 2011; Laaidi et al., 2011; Santamouris, 2015; Gasparrini et al., 2017). The UHI originates from the difference between the rural and urban energy balances due to lower albedo, less vegetation, higher heat storage capacity and anthropogenic heat release in cities (Oke, 1982). By endorsing a higher latent heat flux via the evaporation of water complemented by shading, urban vegetation is often given a central role in attempts to reduce the UHI (Ennos, 2010). Indeed, higher vegetation fractions are associated with lower urban air and canopy temperatures (e.g. Gallo et al., 1993; Weng et al., 2004; Theeuwes et al., 2017), although in specific situations vegetation can cause higher temperatures (Meili et al., 2021a). It has been shown that actively expanding the vegetation fraction as part of urban renewal indeed has the potential to mitigate the UHI (Wei & Shu, 2020). Since vegetation-mediated cooling strongly depends on water availability for evapotranspiration (ET) (Avissar, 1992), there is a need for methods to analyze and evaluate the effective storage in urban environments.

Urban areas not only alter the surface energy partitioning, but affect the water balance and make cities more prone to flooding. The high impervious surface fraction in cities results in more storm water being discharged as runoff in urban areas, which on top of that accumulates faster (Arnold Jr & Gibbons, 1996; Fletcher et al., 2013). Consequently, urbanization decreases water availability for ET and thus indirectly contributes to the UHI (Taha, 1997; Zhao et al., 2014), and together with heavy rainfall events leads to annual flood volumes that are 2–9 times higher than in rural areas (Paul & Meyer, 2001; Hamdi et al., 2011; Zhou et al., 2019). Due to the high population density and concentration of capital in cities, these floods cause considerable damage (Tingsanchali, 2012). Floods and the associated damage are likely to be further increased by the ongoing intensification of the water cycle expressed in among other things more precipitation (Huntington, 2006). Solutions have been proposed under various names such as Water Sensitive Urban Design (WSUD) (Wong, 2006), Low Impact Development (LID) (Qin et al., 2013), Sustainable Drainage Systems (SWS) (Zhou, 2014) and sponge cities (Gaines, 2016). The common ground of these concepts is their focus on increasing infiltration and effective storage capacity, of which the latter at urban landscape scale is crucial for their performance (Graham et al., 2004; Qin et al., 2013). Therefore, an evaluation of the effective storage in urban environments is also needed in the light of flooding events.

Estimating the water storage capacity remains challenging, as water sources for ET in urban landscapes are spatially heterogeneous (Sailor, 2011). Previous studies have focused on ET from individual sources (e.g. Gash et al., 2008; Starke et al., 2010; Pataki et al., 2011; Ramamurthy & Bou-Zeid, 2014), as well as on their combined behaviour at street or neighborhood scale (e.g. Christen & Vogt, 2004; Jacobs et al., 2015; Meili et al., 2020, 2021b). In order to study ET on a neighborhood scale (order of hundreds of meters to 1–2 kilometers), flux measurements of latent heat with an associated footprint, such as those obtained by eddy covariance or scintillometry, are becoming increasingly popular. Examples of cities for which these measurements have been taken and analyzed are Arnhem (Jacobs et al., 2015), Basel (Christen & Vogt, 2004), Helsinki (Vesala et al., 2008), Melbourne (Coutts et al., 2007b), Seoul (Hong et al., 2019) and Singapore (Roth et al., 2017). These measurement sites are chosen such that their footprint covers an area as homogeneous as possible to enable research on the influence of the city design on the surface energy balance partitioning. ET originates from the urban water storage, so ET observations contain information on this storage. However, the link between neighborhood-scale observations and urban water storage has not been exploited, despite the relevance of this water storage.

The concept providing this link between neighborhood-scale ET observations and urban water storage is a recession analysis of the observed ET. From the 1970s, recession analysis has an extensive track record in groundwater and hill slope hydrology linking discharge to water storage (e.g. Brutsaert & Nieber, 1977; Kirchner, 2009; Troch et al., 2013). Similarly, daily ET can be linked to water storage during a drydown, a period without precipitation creating water-limited conditions. Recession analysis of ET decay reveals the timescale over which ET declines by 63% (*e*-folding time), and reflects the available storage and resilience to droughts (Wetzel & Chang, 1987; Salvucci, 2001; Saleem & Salvucci, 2002). The reflected storage is defined by the methodology as the dynamic water storage capacity available to the atmosphere for ET, which includes soil moisture, intercepted precipitation and open water varying from lakes to puddles. In studies using daily ET over natural ecosystems, Teuling et al. (2006) and Boese et al. (2019) found timescales ranging from 15 days for short vegetation to 35 days for forest ecosystems, and corresponding storage capacities in the range between 30 and 200 mm, with most sites in the range of 50 to 100 mm. A similar analysis at the global scale by McColl et al. (2017) determined that the timescales of drydowns in surface soil moisture with satellite imagery on 36 km resolution and found timescales ranging from 2 to 20 days. Although valuable insight can be obtained from a comparison of urban and rural ET dynamics, recession analysis has not yet been applied to urban ET.

In this study, we test and adjust the methodology developed to estimate water storage capacity based on observations of daily ET in natural ecosystems for application in urban environments. This allows us to infer dynamic water storage capacities at the neighborhood-scale from latent flux observations that provide ET from eddy-covariance flux towers. The methodology is tested for neighbourhoods in different cities located across a range of climate conditions and with different urban land cover and structure, which are both likely to affect the storage capacity. The results from the different cities are compared to each other to quantify the effect of the site characteristics (e.g. vegetation fraction) on the water storage, and to natural ecosystems for quantification of the effect of urbanization.

2 Data and Methods

We analyze latent heat fluxes and auxiliary meteorological data obtained from eddy covariance flux towers at thirteen sites in eleven different cities to estimate water storage. Table 1 lists the most important characteristics of each site, including the Köppen-Geiger climate and local climate zones as described by Stewart and Oke (2012), and key references. In these references, all observation sites and measurement details are fully described. The sites were chosen based on the length of the data record (minimum of a year), adequate flux footprints representing typical urban neighborhoods, and the availability of observed precipitation and latent heat fluxes. All sites are located in reasonably flat terrain. Most sites were located in mid-latitude climates, except Mexico City that has a subtropical climate and Singapore with a tropical climate. Helsinki, Łódź and Seoul have a continental climate. Vegetation fractions in the associated footprints vary in the range of 6–56%.

Observations were reported in averaging periods of 10–30 min depending on the measurement protocol of each site. In this study, hourly averages were used to determine the timing of rainfall and 24-hour averages were used for the recession analysis. For all sites the quality control of the observed heat fluxes was performed by individual researchers responsible for their ET flux observation site. Although the exact methodology of the quality control differs per site, all fluxes have been properly tested in accordance with procedures published in literature (Aubinet et al., 2012).

Following Teuling et al. (2006), the change in landscape dynamic storage (S) with time t is given by:

$$\frac{dS(t)}{dt} = P(t) - q(t) - \mathrm{ET}(t) \tag{1}$$

where P is precipitation and q is drainage/runoff, all in $\mathrm{mm\,d^{-1}}$. During multi-day drydowns with no rain, both P and q typically become much smaller than the other terms. Neglecting them reduces Equation 1 to:

$$\frac{dS(t)}{dt} = -\mathrm{ET}(t) \tag{2}$$

In conditions of water-limitation, daily ET becomes a function of (soil moisture) storage. This dependency is often assumed to be linear (Williams & Albertson, 2004; Dardanelli et al., 2004):

$$\mathrm{ET}(t) = f(S(t)) = cS(t) \tag{3}$$

in which $c = 1/\lambda$ is a proportionality constant. Combining Eq. 2 and Eq. 3 and solving the differential equation leads to an exponential response of ET (Williams & Albertson, 2004; Dardanelli et al., 2004):

$$\mathrm{ET}(t) = \mathrm{ET}_0 \exp\left(-\frac{t - t_0}{\lambda}\right) \tag{4}$$

Table 1. Summary of the measurement sites and the outcomes of the regression. The climate statistics are long-term means (1999-2019). The start and end of the indicated ranges for the parameters are respectively the 5^{th} and 95^{th} percentile of the median distribution from the bootstrapping re-samples. The value in brackets is the median itself. (LCZ: 1 = compact high-rise, 2 = compact mid-rise, 3 = compact low-rise, 5 = open mid-rise, 6 = open low-rise, F_v: surface fraction covered by vegetation in a 500 m radius around the measurement site, z_s: height of sensors above ground level, z_H: mean building height, ET_0: initial evapotranspiration, *lambda*: e-folding timescale, $t\frac{1}{2}$: half-life, S_0: effective, dynamic water storage capacity)

City	Lat. (deg)	Lon. (deg)	Köppen-Geiger climate	Avg. temp. (deg C)	Ann. prec. (mm)	LCZ	F_v (%)	z_s (m)	z_H (m)	Start	End	Source	Dry-down	Days	ET_0 (mm d^{-1})	λ (day)	$t\frac{1}{2}$ (day)	S_0 (mm)
Amsterdam	52.37	4.89	Cfb	9.2	805	2	15	40	14	05/2018	10/2020	Ronda et al. (2017) Steeneveld et al. (2019)	14	56	0.99–2 (1.5)	3.4–7.3 (3.8)	2.4–10 (2.6)	5.6–17 (6.7)
Arnhem	51.98	5.92	Cfb	9.4	778	2	12	23	11	05/2012	12/2016	Jacobs et al. (2015)	48	195	0.62–0.95 (0.73)	2.6–4.6 (3.3)	1.8–3.2 (2.3)	2.4–3.9 (3.1)
Basel (AESC)	47.55	7.60	Cfb	10	778	2	27	39	17	06/2009	12/2020	Lietzke et al. (2015)	125	523	0.76–1 (0.83)	4.4–5.7 (5)	3.1–3.9 (3.5)	3.5–4.8 (4)
Basel (KLIN)	47.56	7.58	Cfb	10	778	2	27	41	17	05/2004	12/2020	Schmutz et al. (2016)	160	685	0.9–1.2 (1)	5–7 (5.7)	3.5–4.9 (3.9)	4.7–7.7 (6.1)
Berlin (ROTH)	13.32	52.46	Cfb	9.1	570	6	56	40	17	06/2018	09/2020	Vulova et al. (n.d.)	8	36	0.46–0.95 (0.74)	4.1–12 (6.7)	2.8–8.1 (4.7)	2–9.7 (4.9)
Berlin (TUCC)	13.33	52.51	Cfb	9.1	570	5	31	56	20	07/2014	09/2020	Jin et al. (2020) Vulova et al. (n.d.)	38	153	0.31–0.78 (0.47)	3.3–5.1 (3.9)	2.3–3.5 (2.7)	1.2–3.6 (2.7)
Helsinki	60.33	24.96	Dfb	5.1	650	6	54	31	20	01/2006	12/2018	Vesala et al. (2008) Karsisto et al. (2016)	45	197	1.1–1.8 (1.6)	3.7–5.3 (4.3)	2.5–4 (3)	5.6–11 (7.8)
Łódź	51.76	19.45	Dfb	7.9	564	5	31	37	11	07/2006	09/2015	Fortuniak et al. (2013)	59	269	0.74–1.4 (1)	3.6–5.3 (4.5)	2.5–3.8 (3.1)	3.4–6.7 (4.2)
Melbourne (Preston)	-37.73	145.01	Cfb	14.8	666	5	38	40	6	08/2003	11/2004	Coutts et al. (2007b) Coutts et al. (2007a)	3	12	0.55–2.1 (1.5)	2.6–14 (9)	1.8–9.5 (6.2)	5–21 (5.5)
Mexico City	19.40	-99.18	Cwb	15.9	625	2	6	37	9.7	06/2011	09/2012	Velasco et al. (2011) Velasco et al. (2014)	8	49	0.69–1.5 (1.2)	5.4–17 (11)	3.8–11 (7.6)	5.8–22 (9.5)
Seoul	37.54	127.04	Dwa	11.9	1373	1	40	30	20	03/2015	02/2016	Hong et al. (2019) Hong et al. (2020)	11	66	0.74–1.7 (1.1)	2.8–11 (6.7)	2–7.3 (4.7)	3.1–9.9 (6.7)
Singapore	1.31	103.91	Af	26.8	2378	3	15	24	10	03/2013	03/2014	Velasco et al. (2013) Roth et al. (2017) Harshan et al. (2017)	7	41	1.2–1.6 (1.3)	7–20 (10)	4.8–14 (7.2)	8.7–24 (12)
Vancoucer	49.23	-123.08	Csb	9.9	1283	6	35	28	5	05/2008	07/2017	Christen et al. (2011)	63	293	1–1.4 (1.2)	6.8–9.1 (7.7)	4.7–6.3 (5.4)	6.6–8.9 (7.7)

where λ is the e-folding timescale (i.e. the time over which ET is reduced by 63%), and ET_0 the initial ET. With these parameters the total dynamic storage volume S_0 in mm that would be depleted during a complete dry down ($t \rightarrow \infty$) is given by:

$$S_0 = \int_{t_0}^{\infty} \mathrm{ET}(t)\mathrm{d}t = \lambda \mathrm{ET}_0. \tag{5}$$

so that S_0 can be estimated using only ET observations. To estimate the parameters λ and ET_0, we identified all periods without precipitation for at least three continuous days, since three data points are the minimum requirement for an exponential fit (Figure 1). We only consider the first ten days of the drydowns to reduce the influence of the tail on ET_0, and to limit the contribution of garden irrigation to ET. In order to preserve the information in ET during the first hours after rainfall (in case of low λ), we start the 24-hour averaging bins directly after the rainfall event, regardless of its magnitude. The bin-average is assigned to the middle of the day (e.g. the first bin is assigned to 0.5 day since rainfall). We exclude hours with an average shortwave incoming radiation below 10 $\mathrm{W\,m^{-2}}$ (i.e. nighttime), since during the night ET tends to be low. Only bins with at least 70% of data for daytime hours were analyzed, and no gap-filling was applied. The effect of implementing the threshold instead of requiring 100% was tested for the monitoring sites in Arnhem and Helsinki. We found that while sample size increased by 53 and 27% respectively, the median of the water storage capacities only changed by 6 and 3%. Given the minimal effect on the results and potential to increase the sample size, the threshold provides more information especially regarding cities with a shorter measurement period without compromising the results.

The ET observations were transformed by taking the natural logarithm to acquire a linear relation based on Equation 4, which is fitted trough every individual drydown with the method of least squares obtaining λ and ET_0. With increasing R^2, the parameters converge until an R^2 of 0.3 (not shown), which is thus our minimum requirement. In addition, the parameters are required to be physically plausible meaning that λ and ET_0 have to be positive, and below 80 days respectively 10 $\mathrm{mm\,d^{-1}}$. Also, the average temperature during a drydown needs to exceed 0°C to exclude snow conditions, which is strict enough, confirmed by a double check against snow records. To quantify the uncertainty of the estimated parameters complying with all criteria, we applied bootstrapping using 5000 re-samples containing 90% of the estimates. The confidence interval is defined as the 5$^{\mathrm{th}}$ and 95$^{\mathrm{th}}$ percentile of the median distribution from the re-samples. With λ and ET_0 the storage capacity is calculated according to Equation 5 (shaded area in Figure 1). The calculated storage corresponds to the total storage capacity, when the storage capacity is assumed to be completely filled after every rainfall event. Drydowns occurring during all seasons are included and analyzed for a seasonal effect, since the water storage available to the atmosphere may change due to for example leaf phenology. For both parameters and S_0, we investigate the relation to Köppen climate, LCZ and vegetation fraction. To investigate the possible impact of day-to-day variation or change in energy availability on the results, we repeated the recession analysis based on evaporative fraction (Gentine et al., 2007) multiplied by the average available energy over the drydown, which we included in the supplementary information (Table S1 and Figure S1 and S2).

3 Results

In Figure 2, the individual drydowns (in grey) show a good resemblance of the characteristic behaviour of the recession confirming the exponential behaviour. In general, ET is quickly decaying after rainfall in all LCZ's represented in our sample, indicating urban ET is generally strongly limited by water availability even on the first day after rainfall. As all cities respond similarly, this confirms the qualitative, decaying relation

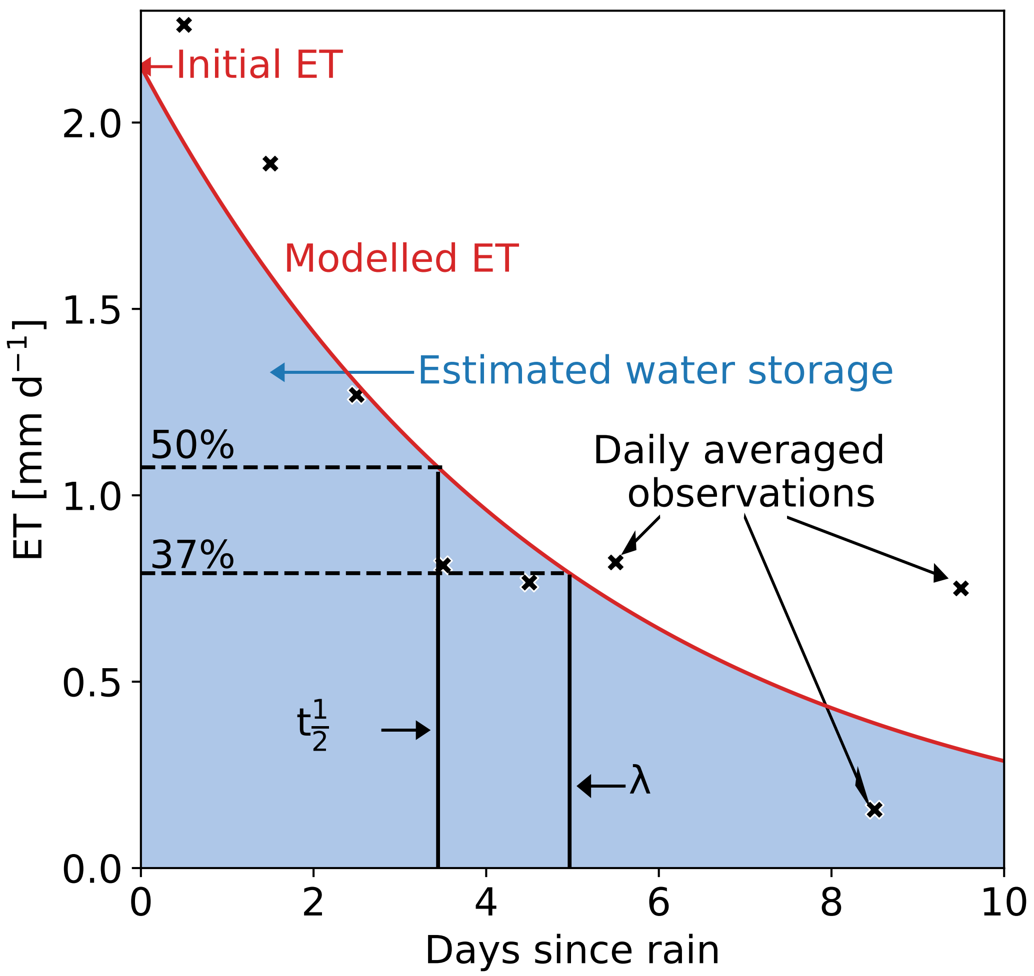

Figure 1. Illustration of the recession analysis. Daily averaged ET versus the number of days since the last precipitation for an example drydown from the Seoul data set with the fitted recession curve. Note that the fit was obtained by a linear fit on log-transformed data (see Data and Methods). In the figure the parameters are indicated.

during a drydown. The uncertainty is visibly higher in cities with shorter measurement periods, since shorter periods inevitably mean smaller samples of drydowns. For Arnhem, Basel (AESC and KLIN), Berlin (Roth and TUCC), Helsinki, Łódź and Vancouver, observations are available for more than two full years resulting in narrow uncertainty bands. In contrast to the uncertainty bands for the sites with records of less than two years (Amsterdam, Melbourne, Mexico City, Seoul and Singapore), which are as wide as the range of observations. In some panels (e.g. Amsterdam and Helsinki), we observe two groups of curves with distinct slopes, for which we found no explanation in seasonality, temperature and pre-drydown rainfall (amount and timing). When vegetation is irrigated in summer, irregularly high ET values can occur long after rainfall (see for examples Helsinki and Vancouver).

In Table 1, an overview of the parameters is given for the 589 drydowns that complied with all criteria. Of the total number of 1611 drydowns, 537 are excluded because of a negative λ and 74 because of a λ above 80 days. All drydowns had a positive ET_0, and only one exceeded 10 $\mathrm{mm\,d^{-1}}$. Snow conditions influenced 133 drydowns, which are thus excluded. Finally, 714 drydowns did not meet the minimum R^2 of 0.3. The remaining drydowns yielded initial evapotranspiration between 0.3–2 $\mathrm{mm\,d^{-1}}$ and e-folding timescales between 2.5–12 days with the majority below 8 days, corresponding to half-lives of 1.7–8.3 and 5.5 days. The related storage capacities appear to be between 1.5–20 mm with the majority below 10 mm. As mentioned before, the length of the measurement period determines the width of the uncertainty, which for S_0 varies from 1.44 mm in Arnhem to 16.03 mm in Mexico City (Figure 2).

For all sites, we find a considerable spread in the ET observations (Figure 2), which recurs in the values found for S_0. In Figure 3, S_0 is plotted against the month of the drydown, showing a very distinct seasonal cycle. Both ET_0 and λ, on which S_0 is based, show similar behaviour (not shown). Melbourne is shifted to fit the seasonality, as it is situated on the southern hemisphere. Since Singapore is close to the equator, it is not expected to show seasonal effect, which is confirmed by the distribution of points in Figure 3 with no clear pattern. Any connection between S_0 and site characteristics is overshadowed by the seasonal cycle covering the full range of S_0 (Table 1), as we illustrate in the supplementary material (Figure S3 and S4). It is unfortunately not possible to eliminate the influence of the cycle by focusing on one season due to the steep slope, and not by focusing on one month due to the low data density. Only after omitting half of the cities based on the number of drydowns, a relation between S_0 and site characteristics is visible (Supplementary material Figure S5).

4 Discussion

In contrast with the results from this study found in urban areas, Teuling et al. (2006) found timescales ranging from 15–35 days and storage varying between 30 and 150 mm for forests and grassland with a similar methodology. When the urban parameter values found in this study are compared with these timescales and storage capacities (2.5–12 days and 1.5–20 mm), it is clear that both the timescales and storage capacities are higher in rural areas. McColl et al. (2017) have analyzed soil moisture drydowns in a global study using satellite data with a resolution too coarse to explicitly resolve individual cities, thus resembling rural values. Although their timescales with values from 2–20 days are closer to ours, it must be noted the temporal resolution is one in every three days and their observations only regard the first few centimeters instead of the root zone. Also, the satellite product in their research is known to underestimate the timescales compared to in-situ observations (Rondinelli et al., 2015; Shellito et al., 2016). Hence, the results show that both λ and S_0 are an order of magnitude smaller in cities indicating shorter timescales and lower storage capacities in urban areas regardless of their climate and vegetation fraction.

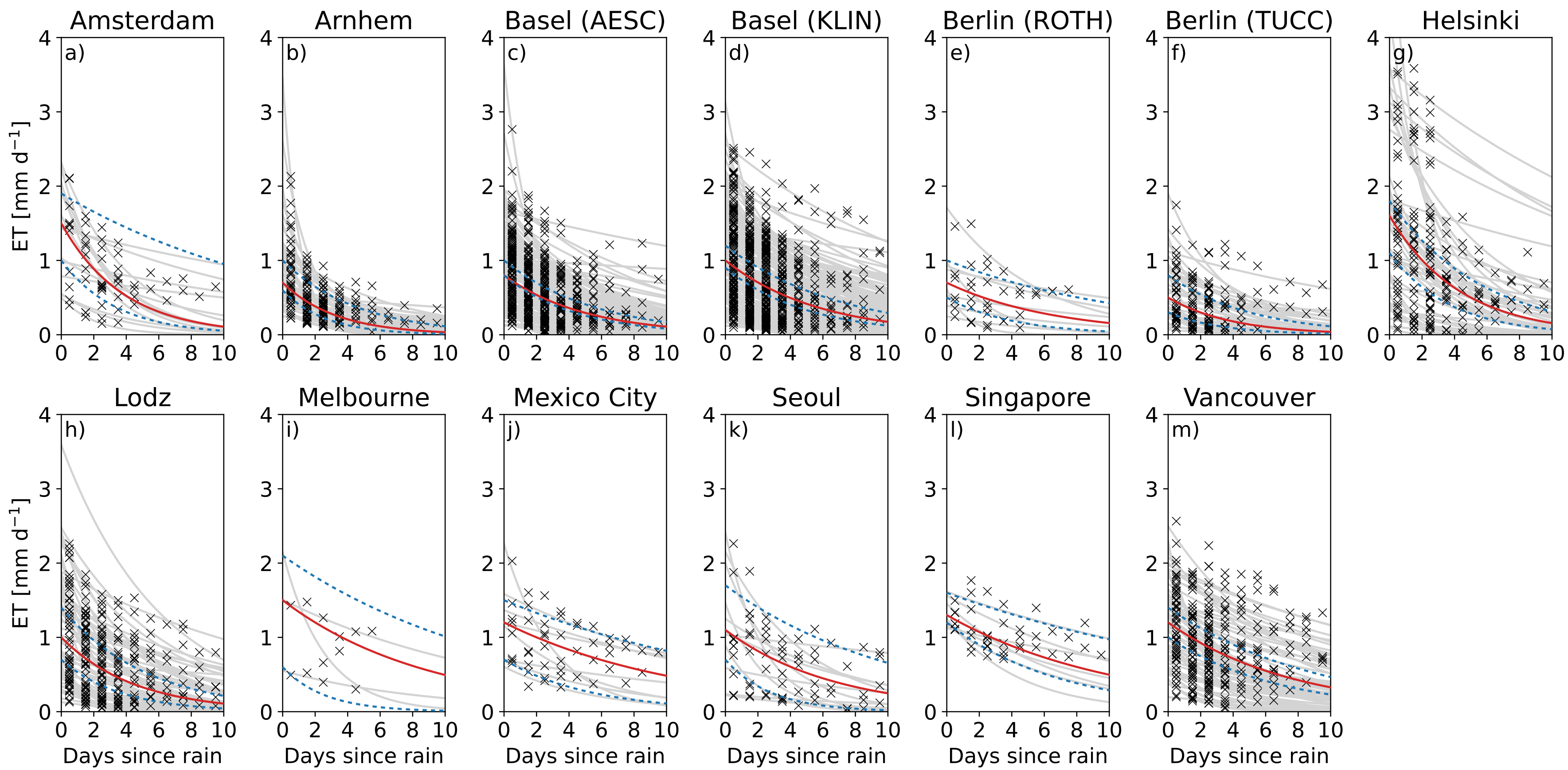

Figure 2. Daily average *ET* versus the day after the last precipitation with in red (continuous) the recession curve using the median parameter values, in blue (dotted) the 5$^{\text{th}}$ and 95$^{\text{th}}$ percentile of the median distribution from the bootstrapping re-samples, and in light grey all individual drydowns. The parameters of the fitted curves are shown in Table 1

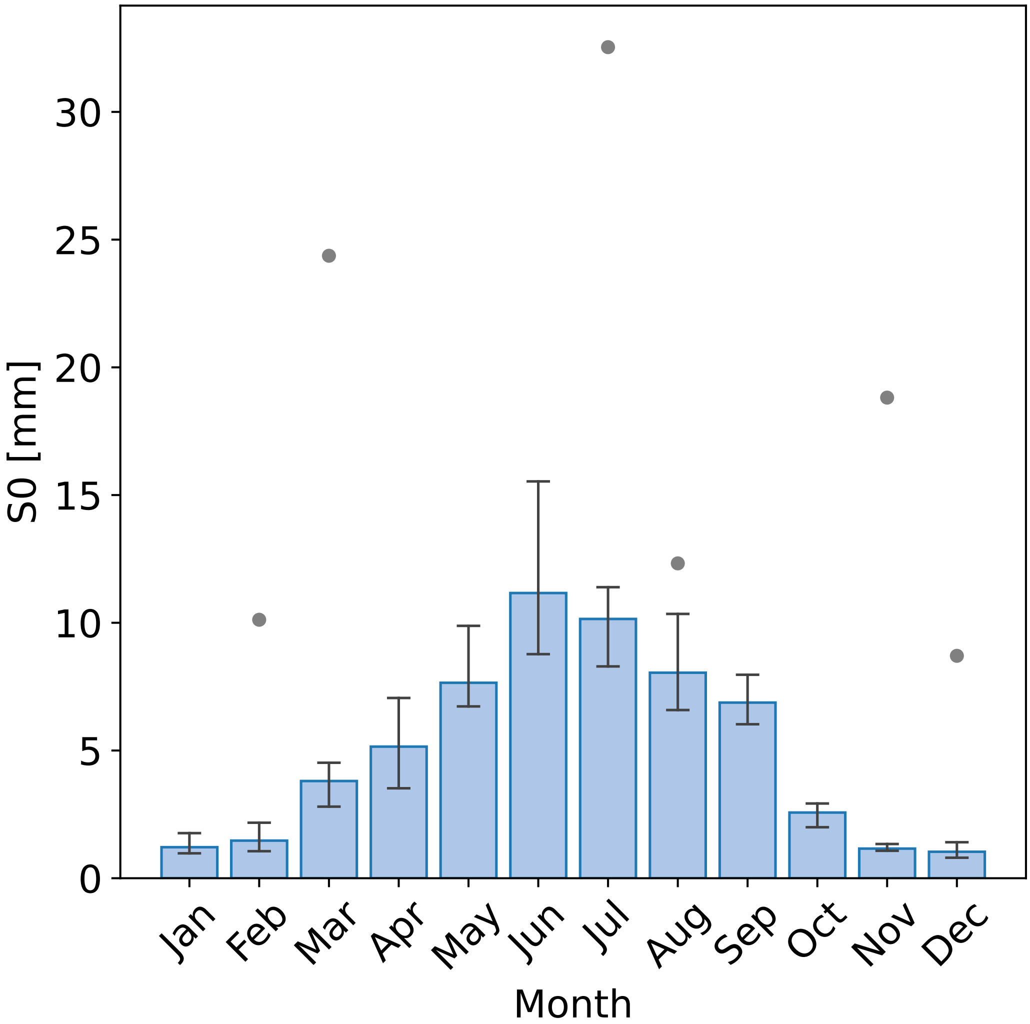

Figure 3. The seasonal cycle of S_0 for the sites on the northern hemisphere (Melbourne is included shifted by half a year) in blue and for Singapore as grey dots. The uncertainty is determined similarly as in Figure 2.

The applied methodology implies a set of assumptions and limitations that confine its utilization. Since the method is observation-based, the reliability of the measurements is an important factor in this confinement. Eddy covariance is a sophisticated method for measuring fluxes, but comes with a set of potential challenges in cities (Velasco & Roth, 2010; Feigenwinter et al., 2012; Järvi et al., 2018). By carefully selecting locations and applying quality control, these problems are minimized. All sites have an observation height well above the mean building height (see Table 1), and measure in the well-mixed inertial sublayer. This reduces the variability in flux measurements in response to the heterogeneity of the monitored footprint, which is induced by the many, unevenly distributed surfaces with different characteristics and water storage capacities in the urban landscape. The only site in this research that includes a non-homogeneous footprint is Seoul, for which the observations are filtered by wind direction to exclude a nearby forest. Additionally, the long bins and quality control keep the influence of sudden fluctuations to a minimum.

Interception ET was an important consideration in the design of the methodology, since the high ET during the first day of a drydown can be largely attributed to this phenomenon (Savenije, 2004). Interception ET is a concept originally developed for forest environments, where interception is defined as water intercepted by the canopy. However, the concept can also be applied in urban environments with the more general definition of a temporary storage space only filled directly after rainfall (e.g. Grimmond & Oke, 1991; Gerrits, 2010; Oke et al., 2017). The volume of this temporary space is affected by urban design choices, such as the use of flat or sloping roofs. In order to capture the peak at the beginning of the recession caused by interception ET, the starting time of the 24-hour bins is not fixed but made dependent on rainfall, which can be during either day or night. Due to a lack of energy availability at night, more water may be drained and thus less interception might be evaporated. Since all parameters are independent of the starting hour of the bins (not shown), a bias due to the starting hour can be ruled out.

Although ET is expected to behave as described in Equation 4, the anthropogenic moisture flux does not necessarily follow this equation. In urban areas, the anthropogenic moisture flux can contribute substantially to ET, in particular during long dry periods (C. Grimmond & Oke, 1986; Moriwaki et al., 2008; Miao & Chen, 2014). This moisture flux includes processes like transport, heating, cooling (indoor), human metabolism and irrigation, which do not directly depend on rainfall. We expect variations in daily averages of these processes, except for irrigation, to be negligible during one drydown. We have tried to capture these flux by adding a constant base term to Equation 4, but this strongly increased the number of drydowns yielding physically unfeasible parameters. Therefore, we conclude that including this part of the anthropogenic moisture flux would not improve the results. As mentioned earlier, irrigation cannot be expected to be constant, while in some cities (e.g. Vancouver (C. Grimmond & Oke, 1986; Järvi et al., 2011) and Melbourne (Barker et al., 2011)) its contribution to ET is considerable. Two choices in the methodology prevent irrigation from affecting the results. The first one being the maximum duration of a drydown of 10 days, and the second being the requirement that the R^2 has to be above 0.3, which is not achieved if ET rises due to irrigation.

The methodology assumes that at the start of a drydown the storage capacity is completely full. A partly empty storage capacity would lead to an underestimation of the capacity, as less water is available for ET. We have compared the magnitude of the rain event before a drydown with the resulting parameters and found no correlation. Since the storage can be refilled by a series of events separated by dry days, we plotted the found parameters against the Antecedent Precipitation Index (API) (Fedora & Beschta, 1989). The API takes into account rainfall occurring during preceding days (here limited to 20), but also shows no correlations with the parameters. Therefore, the assumption of a completely filled storage is tangible and no selection has been performed based on rainfall event size.

The small storage capacity in cities shows their water balance is altered. Despite the limitations of the methodology discussed above, the provided first estimates show the clear contrast between water storage capacity in urban and rural areas. The presented water storage estimation method has potential to offer an intriguing comparative analyses between less distinctly different urban sites. The data presented here are not sufficient to substantiate, but do indicate their existence. In order to coin the larger potential and establish empirical relations between site characteristics and storage capacity, future research will need to focus on including longer records from more cities and applying soil moisture observations as a reference. The first is because urban flux records are scarce both in number and length, as was previously indicated by Grimmond and Christen (2012), but the availability is improving. Additionally, very wet climates, such as in Singapore, only sporadically satisfy the conditions of the methodology increasing the necessity of long observational records. The second, soil moisture, are available for only three sites in this study (Berlin, Singapore and Vancouver), but could provide the opportunity to estimate storage conditions. To establish robust relations, future research will need to include many more cities with records of at least two years.

5 Conclusion

The timescales of ET recession found in urban environments are considerably shorter than in rural environments. This is related to the storage capacity, which is also found to be lower. Timescales of cities are within 2.5–12 days with the majority below 8 days and storage capacities range between 1.5–20 mm with the majority below 10 mm. Both are an order of magnitude smaller than the values found in rural areas. We were unable to analyze differences between cities to vegetation fraction, local climate zone or climate for two reasons. Firstly, a very strong seasonal cycle in the storage capacities as strong as the total found variation. Secondly, the number of sites is limited, for which no more than one year of data is available for about half of them. When provided with more data, the presented water storage capacity method does have the potential to establish robust empirical relations explaining the differences between cities.

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