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Plant transpiration in constructed treatment wetland: Effects on water budget and management consequences

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ABSTRACT

Plant transpiration is an important feature of wetlands with biological and hydraulic impacts. The global objective of this study was to question the influence of transpirational water losses on constructed treatment wetland water budget for a variety of wetland design and time of the year. Biomass and transpiration field measurements were carried out in constructed treatment wetlands (CTWs) submitted to oceanic climate and used for waste- or stormwater management. Measurements were carried out during spring, summer and fall. Biomass and transpiration rate were both significantly affected by season and site configuration, although the effect appears more sharply for season than for site. Transpiration can reach 26% of the incoming flow during the warmest part of the year for wastewater management CTW, when the effect on adjacent water courses is likely to be the most significant. The impact on multi-monthly water budget plummets to 2% of the incoming water volume. For stormwater CTW, transpiration can lead to strong water scarcity, virtually emptying all available water in these stochastically fed systems. As transpiration also plays a significant role in biogeochemical processes in wetlands, it seems important to design this type of ecological infrastructure in close relation with the pursued objectives, be it either the quality of outlet water (emphasis on treatment efficiency) or the quantity of outlet water (emphasis on flow regulation).

1. Introduction

Urban population is growing, and cities will soon become the major living place for mankind, with estimated 80% of the world population living in urban areas by 2050 (Grimm et al., 2008). The resulting population densification will generate increased pressure on all components of these urban social-ecological systems (SES). For instance, urban water bodies are already facing these disturbances as presses (pollutant fluxes in wastewater flow) or pulses (pollutant fluxes and water volume carried in stormwater flow). These disturbances were acknowledged long ago and led to building various centralized infrastructures to reach the state of "sanitary cities" (Grove, 2009; Melosi, 2008). As aquatic ecosystems, such as rivers, lakes, wetlands, and coastlines, provide cities with vital ecosystem services (e.g., the provision of drinking water, regulation of water quality and quantity, support for habitats, transportation), we should find novel ways to mitigate disturbances these systems face, now and in the future. Ecologically-based solutions are often the most adaptable and resilient ways to move cities towards more sustainable trajectories (Walaszek et al., 2018a). Constructed treatment wetlands (CTWs) are one use of Urban Ecological Infrastructure (UEI) (Daniel L. Childers et al., 2019), gaining in popularity as a solution to urban wastewater and stormwater management challenges.

CTWs are preferable to high capital engineered wastewater treatment plants because they have low energy and maintenance requirements and are relatively cheap to construct (Nivala et al., 2013; Walaszek et al., 2018a, 2018b; Wallace and Knight, 2006). CTWs combine physical and biogeochemical processes, in a slightly engineered manner, to achieve flow and volume regulation as well as pollutant mitigation (Bois et al., 2019). Their efficiency is framed by regulations or guidelines related to wastewater and stormwater management that are focused on either pollutant concentrations or loads (Directive, 2000; Federal Water Pollution Control Act, 2002; WHO,

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2011), in an effort to account for both environmental and sanitary impacts of polluted water. When concentration is the legal parameter, water budgets need to be well characterized, especially in these open systems where water quantity is likely to change significantly due to infiltration, evaporation, and plant transpiration (Brown et al., 1988; McLaughlin and Cohen, 2014; Sanderson and Cooper, 2008). Of these three mechanisms of water loss, transpiration typically receives the least attention in whole-system water budgets. Although recent studies have shown that under arid climates, for instance, plant transpiration is high enough to drive surface hydrology in a CTW, which in turn made this wetland more effective at processing nutrients and pollutants (Bois et al., 2017; Sanchez et al., 2016; Treese et al., 2020). The effect of plant transpiration on the water budgets of CTWs in more mesic climates has been the subject of past studies (Bialowiec et al., 2014; Borin et al., 2011; Consoli et al., 2018), yet rarely on actual full-scale systems with direct and instant transpiration measurements. Notably, CTWs that manage stormwater may face water scarcity challenges between storms and inflow events; this could impair the functioning of the wetland plants, adapted to more hydric conditions. Similarly, the importance of plant transpiration to wastewater CTW water budgets in more mesic climates is less well known (Beebe et al., 2014). In these systems, evapoconcentration of solutes in the wetland soils may result in poor system performance and failure to meet regulatory goals, as shown in pilot-scale systems (Bialowiec et al., 2014; Tuttolomondo et al., 2016). As these systems are slightly engineered, specific features may lead to significant differences of functioning.

We propose to study here the impacts of seasonal dynamics on transpiration and subsequent water budget. We thus quantified instant and integrated transpirational water losses in five actual full-scale CTWs submitted to mesic climate (France) in terms of magnitude and dynamics as potentially influenced by i) climatic parameters (e.g., seasonal assessment), ii) hydraulic design (e.g., surface vs. subsurface flow, permanent or temporary inflow) and iii) hydrochemistry (e.g., stormwater or wastewater feeding). We also assessed the influence of transpiration on the water budget of these wetlands, with regards to design specifications, seasonal variability and temporal dynamics. Based on these analyses, we propose management adjustments that account for the influence of transpiration on water budgets in mesic-climate CTWs.

2. Material & methods

2.1. Site description

We chose study sites presenting a variety of purposes and designs. Five sites built to treat either wastewater or stormwater and designed as surface flow or horizontal/vertical subsurface flow systems were thus studied (Table 1). All study sites are located in urban or peri-urban places in Alsace (Fig. 1), with oceanic climate (Fig. SM1). Site E (Figs. 1 and 2) receives screened wastewater and was designed to reduce Biological Oxygen Demand (BOD), suspended solids, nitrogen and phosphorus concentrations (i.e., secondary treatment). Site L (Figs. 1 and 2) receives secondarily treated wastewater and is meant to further lower nitrogen, phosphorus, and potentially micropollutants concentrations (tertiary treatment). Sites S1, S2 and S3 (Figs. 1 and 2) were designed to treat urban stormwater runoff and reduce discharging volume into an adjacent stream while also lowering micropollutant concentrations. As S# systems are stochastically fed, a backup water volume, proportional to the wetland surface, is kept at their bottom to alleviate potential plant water stress during the dry season (Table 2). Main hydraulics and sizing characteristics are indicated in Table 2 and were retrieved from local water agency.

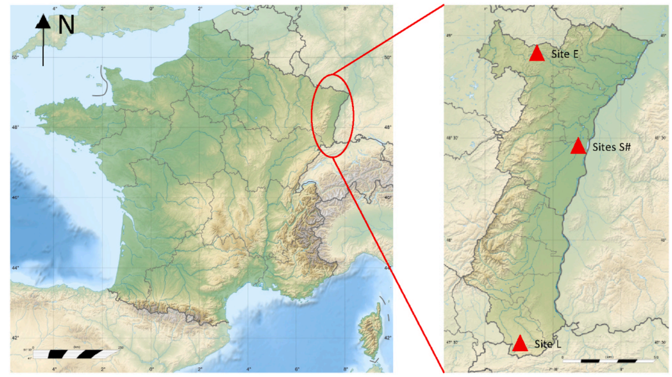

Fig. 1. Location of study sites.

2.2. Plant transpiration

Globally, water transpiration by plants is the multiplication of plant specific water transpiration rate – the water volume transpired per biomass and time units (expressed in L. g_{dw}^{-1}. h^{-1}), – by biomass per surface unit in the system (expressed in g_{dw}. m^{-2}). It is thus expressed in L.m^{-2}. h^{-1}. The global transpirational water losses for each system, expressed in L.h^{-1}, are obtained by multiplying the above value by the system area.

Biomass measurements: Plant biomass was estimated combining field measurements with allometric biomass models for the dominant species in each CTW, following established methods (D. L. Childers et al., 2006; Daoust and Childers, 1998; Weller et al., 2016). On the field, five 0.25 m^2 quadrats were randomly located in each system. Measurements of all plants morphological parameters in each quadrat were made in a non-destructive way, based on the parameters determined by the allometric models. It was conducted at key points of the vegetative growth: spring (maximum growth phase), a few weeks after plants appearance, at the heart of summer around July/August (peak biomass phase), and in late October for fall (senescing phase). These moments are strongly dependent on sites, so field expertise is strongly required to determine them. The allometric models were specifically determined for both encountered species, using around 30 plants samples for each species, and allowed to link morphological variables with global plant biomass. It was eventually combined with the field measurements to produce estimates of live plant biomass, in g_{dw}. m^{-2}. Other herbaceous and woody plants were present at L and S# sites, but in fewer numbers; we

Table 1
Features of the constructed wetlands. SF: Surface Flow; VSSF: Vertical Subsurface Flow; HSSF: Horizontal Subsurface Flow. *: main vegetation at the time of the study. Filling is indicated from top to bottom. na: not applicable.[a]: data from (Laurent et al., 2015).

	E	L	S1	S2	S3
Influent	Wastewater		Stormwater		
Purpose	Secondary treatment	Tertiary treatment	Management and treatment		
Hydrology	VSSF	SF	VSSF	HSSF	VSSF
Vegetation*	*P. australis*	*T. domingensis*	*P. australis*	*P. australis*	*P. australis*
Shape	Square	Meandering	Rectangle	Rectangle	Rectangle
Dimensions (LxWxD, in m)	15 × 14.8 × 0.8	38 × 1.7 × 0.4[a]	11x8x0.8	45 × 6.5 × 0.8	20 × 5 × 0.8
Waterproofing	Liner	Na	Clay		
Filling	Gravel, coarse gravel	Na	Fine sand, fine gravel, coarse gravel	Fine gravel	Sand, fine gravel, coarse gravel

Fig. 2. Pictures of study sites. Left: site E; middle: site S; right: site L.

Table 2
Hydraulic and design characteristics of the constructed wetlands. *: nominal theoretical values, altered after years of operation, except[a]: Laurent et al., (2015).

	E	L	S1	S2	S3
Inflow (m^3 d^{-1})	94	103	Na	Na	Na
Mean Residence Time* (h)	~12	~1.5[a]	>120	>120	>120
Backup Volume (m^3)	na	na	5.9	31	6.5
Surface (m^2)	222	63	90	294	100
Configuration	Serial				
Operation	Self-priming siphons	Gravity-induced flow	Self-priming siphons		
Starting operation year	1999	2009	2012		

only estimated biomass of the dominant macrophytes for this study.

Transpiration measurements: Transpiration measurements were also made during spring, summer, and fall. The instantaneous transpiration rate (TR, expressed in L. g_{dw}^{-1}. h^{-1}) was measured (Sanchez et al., 2016). Briefly, a LI-6400XT handheld infrared gas analyzer (IRGA; LI-COR, Lincoln, Nebraska, USA) was used on leaves from 30+ randomly selected plants of the dominant species, at 0.5 m height increments, in each CTW. The IRGA also logged weather parameters, including photosynthetically active radiation (PAR, in $W.m^{-2}$), air temperature (T, in °C), and relative humidity (HR, in %). TR values were averaged for every hour during which measurements were made. These average TR values were scaled in time using hourly data from meteorological stations located at each CTW site, using a multiple linear regression of TR as a function of T, HR, and PAR (sites L, S1, S2 and S3), and of TR as a function of T and HR for site E (PAR measurement was not available for this site). These daily transpiration values ($mm.d^{-1}$) were scaled in space, to the vegetated area of each CTW, using the biomass data (in g_{dw}. m^{-2}) and the total area of each wetland (in m^2), to generate a total daily transpirational water loss for each CTW (in $L.d^{-1}$).

Annual plant transpiration: the calculations for the annual transpired volume were made according to already published work (Sanchez, ibid.). Briefly, a linear model TR = f (T, HR, PAR) was established for every site using the three measurements sessions carried out in spring, summer and fall. We chose 2/3 of the field values for model calibration and 1/3 for model validation. The obtained R^2 were 0.79, 0.94, 0.89, 0.85 and 0.84, for E, L, S1, S2 and S3 respectively. For stormwater wetlands (S1, S2 and S3 sites), RMSE (Root Mean Square Error) was computed for the validation step (it was equal to 0.89, 1.8 and 1.3 for S1, S2 and S3 respectively). The transpiration rate was then upscaled to the whole vegetative season by using the meteorological data recorded during this whole period of time, on each site. To finalize the calculation, we assumed that nighttime transpirational water loss was negligible. Notably, this assumption is one reason that makes our transpiration estimates conservative (Sanchez, ibid.). We linearly interpolated biomass values to compute values over the year, and subsequently computed yearly transpirational losses by combining yearly transpiration rate and yearly biomass values. Eventually, the RMSE value was used to estimate the value range of these yearly transpirational losses for stormwater CTWs (S1, S2 and S3).

2.3. Water budget

Plant transpiration can play a significant role in CTW biological and hydrodynamic functioning (Bois et al., 2017; Sanchez et al., 2016; Weller et al., 2016). To assess how significant water transpirational losses played a role in the studied constructed wetlands, transpiration values were compared with the global water balance of each study system. We detail below the associated model, assumptions and computation used to perform this comparison.

Global assessment: the water balance for any system can be written as:

$$\frac{\Delta V}{\Delta t} = Q_{in} - Q_{out} + R \times S_{system} - I - Ev - ET \quad (1)$$

- $\frac{\Delta V}{\Delta t}$: water volume variation (m^3. h^{-1}).
- Q_{in}: inflow (m^3. h^{-1}).
- Q_{out}: outflow (m^3. h^{-1}).
- R: rainfall ($m.h^{-1}$).
- S_{system}: system area (m^2).
- I: in- or exfiltration (m^3. h^{-1}).
- Ev: evaporation (m^3. h^{-1}).
- ET: evapotranspiration (m^3. h^{-1}).

In- and outflow were obtained using monitoring data from local water agencies. For wetlands E and L, RxS_{system} was negligible compared to the other factors (especially Q_{in}), and was not included in water budget calculations. For S# wetlands, rainfall over the experiment period was 303.6 mm, which is not negligible compared to other terms; RxS_{system} was thus included in water budget calculation. Similarly, as the wetlands at sites E and S# were constructed to be impervious, infiltration (I) was negligible. At site L, the natural soil was kept when building the wetland; water infiltration in soil can thus occur. Additionally, the latter wetland was the only one with surface flow, which made open water evaporation (Ev) not negligible in this case. It was calculated using the Rohwer equation (1931):

$$Ev = 0.484*(1 + 0.64*u)*\left(1 - \frac{RH}{100}\right)*e^{17.27*T/(T+273.15)} \quad (2)$$

where Ev is expressed in $mm.h^{-1}$, u is the wind speed in $m.s^{-1}$, RH and T defined as above. Ev was then converted in $m.h^{-1}$ for water budget calculation. The water budget was calculated for the total study duration for all wetlands (see below for additional precision). Start and end dates for the modelling work were from April 1st to October 31st of 2015; these dates engulf the field sessions.

Water budget features: we discriminated analyses depending on the feeding type of each system. Wastewater CTWs (sites E and L) receive regular inflow; it makes sense to first compare hourly transpiration rate with incoming flow. To get a more global picture, we then upscaled hourly results to values for the total study duration to assess plant transpiration contribution to the water budget during the vegetative season. Conversely, by design, stormwater management CTWs (sites S#) are stochastically fed, only during storm events; we chose to first compare whole system transpiration losses with the water volume entering these wetlands during the study period (labelled V_{in}), that is equal to the sum of $\int Q_{in} * dt$ (inflowing volume from runoff) and RxS_{system} (water volume from rain). We eventually compared transpiration values over the study duration with the total volume in the system (labelled V_{tot}), equal to the sum of the entering water volume plus the backup volume in these systems (equal to 5.8, 31 and 6.5 m^3 for S1, S2 and S3 wetlands respectively) to tackle the issue of water stress in stormwater wetlands ($V_{tot} = V_{in} + V_{backup}$).

2.4. Data analysis

From the data we gathered on the field (biomass, transpiration rate and climatic parameter), we tested several hypotheses. We used ANOVA to investigate the effect of season, study site and plant type on biomass and transpiration rate values. Both "unweighted means" and "weighted means" methods were employed to take the different sample sizes into account when looking at the plant effect (15 biomass values and 63 TR values for *Typha domingensis* plants vs. 60 biomass values and 396 TR values for *Phragmites australis* plants). Student or Wilcoxon test (given the distribution of data) were used to investigate the differences of biomass and transpiration between each site and Hill-Smith analysis to determine which parameter, from season or site, exerts the most significant effect on biomass and transpiration. All these analyses were performed with R software (R Core Team, 2020). For all p-value interpretation, the threshold value was set to 0.05.

3. Results & discussion

3.1. Biomass

Field recorded biomass values are summarized in Table 3. ANOVA results indicate significant effects ($p < 0.05$) of both season and site, but not of the plant species.

Let us first consider the effect of season on biomass; different patterns were observed on each study site. On site E, biomass rose sharply from spring to summer under the effect of nutrients brought by wastewater and receded towards fall. Yet biomass was maintained at a high value from summer to fall, as it remained more than tenfold spring biomass; no field observation other than the start of senescence could explain the decrease observed in the fall. On site L, biomass was stable between spring and summer and increased towards fall. Plant biomass was not significantly different between spring and summer, as other vegetation types developed during the same period and reduced the available space for macrophytes (Miller and Zedler, 2003). On site S1, biomass increased from spring to summer and reached values not significantly different from spring biomass during fall. Field observations showed that other vegetation was settling along the year, decreasing the space available for macrophyte vegetation. On site S2, biomass decreased from spring to summer and increasing again towards fall. The observed decrease from spring to summer might be related to water stress, as will be detailed later in the study (cf. § water budget). On site S3, biomass steadily increased from spring to summer and to fall. This constant significant growth along the year was supported by non-competitive growth of other species within the wetland, as seen on the field. This strong difference of biomass on a given site along the vegetative season is already known for various types of wetland plants (Engloner, 2009) and was observed in constructed treatment wetlands under different locations (resp. Australia, Argentina, and Czech Republic) and climates (resp. Mediterranean, humid tropical and continental), with similar biomass values (resp. 0.5–2.7, 0.2–1.1 and 0.5–5 $kg_{dw}.\ m^{-2}$) (Adcock and Ganf, 1994; Hadad et al., 2006; Vymazal and Kröpfelová, 2005).

If we now consider the effect of site on biomass, we observe that a baseline value was recorded for 4 out of 5 sites during spring, the only significant difference residing between L and S3 sites. During summer, the overwhelming biomass for site E is the feature that stood out significantly ($p < 0.05$) from other sites' biomass; for the latter, biomass values were not significantly different. During fall, biomass on S1 site was significantly lower than on the other sites, while plant biomass did not differ significantly for the remaining sites. This strong difference in biomass for sites sharing the same plant type (i.e., sites E, S1–S3) but with different geographic localization (E vs. S1–S3) comes most probably from the feeding type (Table 1: continuous wastewater at E, intermittent stormwater at S1–S3). Indeed wastewater, either primarily (Molle et al., 2005) or secondarily treated (Licata et al., 2019) contains much more N- and P- than stormwater (Schmitt et al., 2015), which are limiting factors for macrophyte growth in wetlands and play. The high biomass value on E site (2 $kg_{dw}.\ m^{-2}$) was also observed for different treatment wetlands handling various kinds of wastewater (HSSF wetlands handling tertiary treated wastewater (Adcock and Ganf, 1994), FWS wetland handling industrial/wastewater (Hadad et al., 2006) and HSSF wetland handling wastewater (Vymazal and Kröpfelová) (see former paragraph). Thus, it seems that there is a significant effect of the site configuration, mainly due to the type of influent, not necessarily due to the climate.

Table 3
Biomass across sites and seasons.

Biomass ($g_{dw}.\ m^{-2}$)	Spring	Summer	Fall
E	7.7E+01 ± 2.5E+01	2.0E+03 ± 1.9E+02	9.6E+02 ± 2.2E+02
L	2.1E+02 ± 2.7E+01	2.7E+02 ± 3.3E+01	1.1E+03
S1	9.2E+01 ± 1.8E+01	3.6E+02 ± 5.7E+01	1.9E+02 ± 3.2E+01
S2	1.6E+02 ± 5.2E+01	3.4E+01 ± 6.6E+00	6.2E+02 ± 1.2E+02
S3	6.5E+01 ± 1.9E+01	3.9E+02 ± 8.9E+01	6.8E+02 ± 1.7E+02

3.2. Transpiration

This section is based on the results obtained during the field measurement sessions. We will first focus on the transpiration rate (TR, expressed in $L.\ g_{dw}^{-1}.\ h^{-1}$) to inform the effect of several drivers (season, site configuration and plant species) on individual plant transpiration. We will then analyze daily transpiration results (expressed as $mm.d^{-1}$ and computed as mentioned in Material and Methods section) to question the hypothesis of seasonal and configuration influence on global water transpirational losses.

3.2.1. Transpiration rate

Plant transpiration rate globally varied between 0.6 and 4 $mL.g_{dw}^{-1}.\ h^{-1}$ (Fig. 3, Table SM2). The results of an ANOVA analysis performed on these transpiration rate data showed a significant effect ($p < 0.05$) of both seasons and sites, but quite surprisingly not of the plant species, contrary to what was previously shown (Busch, 2000; Pedescoll et al., 2013; Sanchez et al., 2016). This least effect from the plant species may be due to the comparatively greater influence of other factors in our study (see paragraph thereafter on the difference of transpiration between study sites).

When looking at the effect of season on plant transpiration rate, we see that for all sites, the transpiration rate increased significantly from spring to summer ($p < 0.05$) for all plant species (*P. australis* for sites E and S#, *T. domingensis* for L site), about four times. It then decreased from summer to fall, although with various amplitudes for each site. On site E, transpiration rate was thus constant over summer and fall

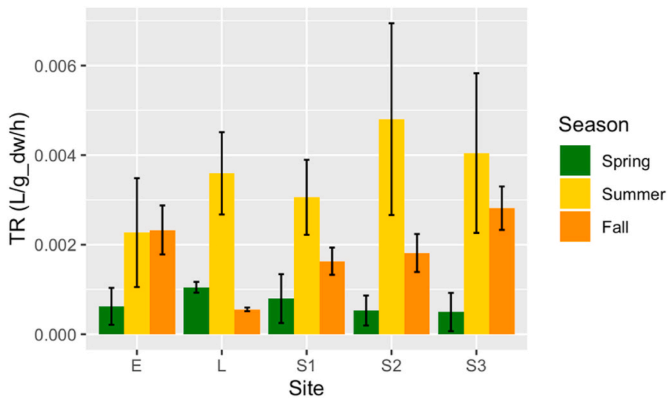

Fig. 3. Field transpiration rate per mass unit. E, L and S# refer to the study sites.

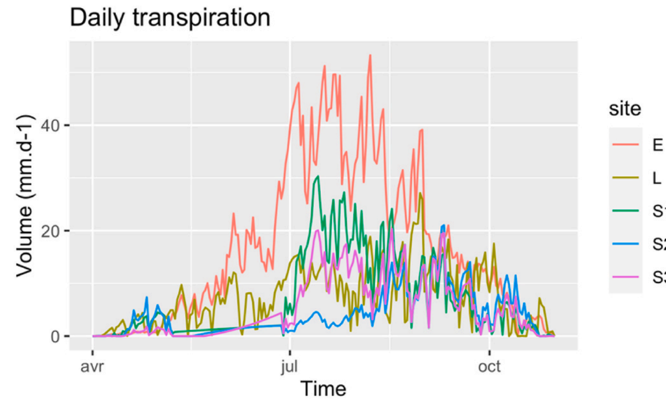

Fig. 4. Daily transpiration in mm.d^{-1} over the year.

measurements. For each L, S1, S2 and S3 sites, though, transpiration rate was significantly lower during fall than during summer. The spring transpiration rates were significantly different than fall ones for each site. The seasonal dynamics we observed in transpiration rates across the seasons was expected, as transpiration is a biological mechanism mainly driven by temperature, radiation, humidity and their dynamics (Licata et al., 2019; Sánchez-Carrillo et al., 2001).

We then looked at the transpiration differences between the study sites. Following pairwise Wilcoxon tests as the datasets were not normally distributed, we obtained similarity groups (Table SM3). There are significant differences, which first seems not so surprising (Tuttolomondo et al., 2016; Zhou and Zhou, 2009), but they could also be due to site configuration or climatic variability during field sessions. To answer this question, we performed a Hill-Smith analysis with transpiration rate (TR), temperature (T), relative humidity (RH) and photosynthetically active radiation (PAR) as quantitative variables and site (site.#) as a qualitative variable. 31% of the global inertia is explained on the first axis of the analysis, 17% on the second axis. Factorial map (Fig. SM2) shows that transpiration rate (TR) is mainly correlated with climatic variables temperature (T) and relative humidity (RH) and only secondarily with PAR and site location. The differences we observe between sites are strongly correlated with PAR and only weakly with TR, T and RH.

Eventually, in our study the main reasons for differential transpiration rates seem to be climatic parameters (i.e., T and RH) rather than by plant type or CTW hydraulic and influent type (sites differ in terms of plant species, hydraulic feeding and influent type, cf. Table 1).

3.2.2. Daily transpiration

We previously discussed the field results we obtained for the two factors that determine transpiration: biomass and transpiration rate. We saw a clear effect of season on both factors, and a sharp effect of site for biomass. We will now look at the year-long transpiration values, obtained through biomass and transpiration rate scaling as explained in the Material and Methods section, and expressed in mm of transpired water per day.

First, we see an obvious seasonal pattern (Fig. 4), with transpiration values ranging from almost 0 to nearly 60 mm d^{-1} at the highest. This seasonal influence was observed on each study site. Low transpiration in the spring resulted from joint low biomass values and transpiration rate; high – or very high for site E – transpiration values during summer resulted from both large biomass values and high transpiration rate. Transpiration decreased in early fall because of biomass and – for sites L, S1, S2 and S3 – transpiration rate decrease. As biomass slightly changes at a daily scale, corresponding dynamics were due to weather changes (e.g., radiation or temperature evolution). These transpiration values and dynamics are similar to what was found wastewater treatment wetlands, at mesoscale with 0–48 mm d^{-1} for *T. latifolia* plants (Licata et al., 2019), pilot-scale with 0–23 mm d^{-1} for various types of macrophytes (Pedescoll et al., 2013) or full-scale with summer averages of 30, 16, 0.6, and 9.6 mm d^{-1} for *Typha* spp., *S. acutus* + *S. tabernaemontani*, *S. californicus*, and *S. americanus* (Sanchez et al., 2016).

If we look at the differences between sites, it might be interesting to consider the distribution of transpiration values (Fig. 5). It is quite scattered for site E, while it is much narrower and in closer ranges for the other sites. As for daily transpiration, the only site standing out is the E site. As the feature that differentiates this site from the others is the use of primary treated wastewater, it seems quite logical that the nature of feeding effluent influences the transpiration status of plants; from what we discussed in 3.1. and 3.2.1. Sections, this is primarily linked with biomass differences. Globally, the calculated average transpiration values over the study period are close – 0.48 (L), 0.54 (S3), 0.63 (S1) to 2.0 (S2) and 3.8 mm d^{-1} (E) –, and comparable with natural wetlands under Mediterranean climate (Eichelmann et al., 2018) where evapotranspiration was reported through eddy flux-based method at roughly 2.7–3.1 mm d^{-1}.

Having looked at biomass, transpiration and the effects of season and site configuration on transpiration, we will now discuss the influence of transpiration on CTW functioning. We showed in the above paragraphs that transpiration is strongly impacted by these two parameters; we will now see how transpiration contributes to the water budget, discriminating wastewater and stormwater management CTW.

3.3. Significance of transpiration in CTW functioning

3.3.1. Contribution of transpiration to the water budget of wastewater treatment wetlands

When looking at the hour scale, field measurements show that transpiration accounts for 0.3 up to 26% of the inflow in the secondary

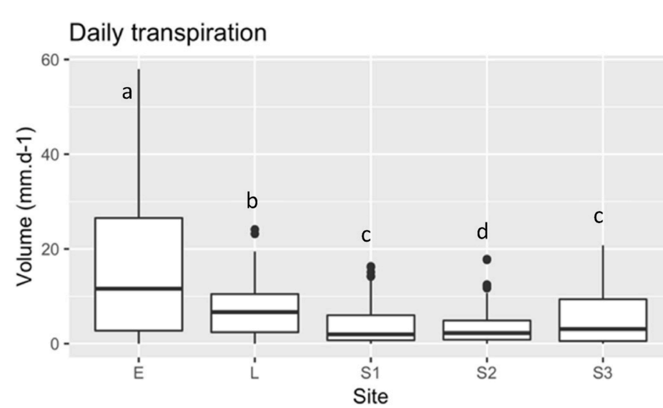

Fig. 5. Daily transpiration in mm.d^{-1} over the year; boxplot for each site. Similarity groups on the top of each boxplot.

treatment wetland (site E, Table 4); this corresponds to 11 L, respectively a cubic meter of transpired water per hour. The latter value, strongly significant for the water budget, occurs during summer, when biomass is at its utmost and temperature is high. Transpirational losses thus represent a quarter of the incoming wastewater during the warmest time of the year, and a tenth during fall. For the tertiary treatment wetland (site L), the hourly transpiration accounts for 0.33 (spring) up to 1.4% (summer) of the inflow; it corresponds to 14, respectively 61 L water transpired per hour. The loss is much lower than for site E, due to the large difference in biomass (see section 3.1). Similar values were found in (Consoli et al., 2018), where transpiration represents between 2 and 25% of the incoming water flow (10-days average) in a HSSF CTW receiving secondary treated wastewater (similar to L site for the influent), although the dominant plant species in this case was *P. australis* (instead of *T. domingensis* for E site). Yet for exhaustive comparison, the estimation of biomass in the study site would be needed.

When scaled up at the annual level though, transpiration reaches 762 m^3, which makes evaporation accounts for only 2.2% of the water budget for E site. When scaled up at the annual level for L site, evapotranspiration is mainly due to evaporation, reaches 387 m^3 and thus accounts for 1.0% of the water budget. This fairly constant impact on water budget for site L results from the high impact of open-water transpiration; without it, the contribution of transpiration on an annual level would be only 0.2% of the total water budget. The globally minored contribution of evapotranspiration to water budget over a year in these continuously fed systems (during nights, and all year long) can be explained by minimal transpiration the night – we even considered it zero for our study – and null values between November to end March due to the absence of alive biomass. The results we found are sensibly smaller than what was found in (Headley et al., 2012), where annual transpirational water losses by *P. australis* reached 6–10% of incoming wastewater, probably due to the oasis effect generated in the 4 m^2 studied wetlands compared to the systems we studied (area between 63 and 222 m^2).

3.3.2. Contribution of transpiration to the water budget of stormwater management wetlands

The results for stormwater-managing CTW (Table 5) show that

Table 4
Evapotranspiration and water budget parameters for wastewater management systems. TR field values are obtained by combining field values of TR ($L.g_{dw}.h^{-1}$) and biomass ($g_{dw}.m^{-2}$); Transpiration values (T) are obtained by combining field values of TR ($mm\ h^{-1}$) and wetland surfaces (m^2); Q_{in} is the incoming flow; $V_{ET} = V_{evaporated} + V_{transpired}$ is the annual evapo-transpired volume; V_{tot} is the annual incoming volume.

	Season	TR (mm. h^{-1})	T (L. h^{-1})	Q_{in} (L. h^{-1})	T/ Qin (%)	V_{ET} (m^3)	V_{tot} (m^3)	V_{ET}/ V_{tot} (%)
E	spring	0.05 ± 0.04	11 ± 8	3917	0.3 ± 0.2	0 + 793	34,313	2.3
	summer	4.55 ± 2.48	1010 ± 550		26 ± 14			
	fall	2.24 ± 0.73	498 ± 162		13 ± 4			
L	spring	0.22 ± 0.03	14 ± 2	4292	0.33 ± 0.05	287 + 100	37,596	1.0
	summer	0.97 ± 0.27	61 ± 17		1.4 ± 0.4			
	fall	0.62 ± 0.05	39 ± 3		0.91 ± 0.07			

hourly transpiration ranges from 0.03 mm h^{-1} during the spring to 1.91 mm h^{-1} during the fall, with corresponding transpiration around a hundred cubic meters per year. There is no open water evaporation here, as all the systems are subsurface flow wetlands. Even when accounting for modelling uncertainties with RMSE (Table 5), the transpired water amount largely surpasses – more than 15 times in average – this backup volume for every site. To assess the potential water scarcity, we compared transpirational water losses with the water volume having flown into the wetland during the study period (V_{in}): these losses represent between 0.8 (S3 site, lower range value) and 7.1 times (S2 site, upper range value) the incoming water volume during this period. This is close to what (Owen, 1995) found, with plant transpiration equal to 93% and 122% of the incoming water on the two seasons of the study, with 4 different plant species accounting for transpiration measurement through mass-balance approach; differences may be due to the differences between the systems (92ha area, and a different climate). As herbaceous and woody plants were present (although in fewer numbers) on S# sites, it is probable that whole transpiration values were actually higher than what we measured. Even supposing that each wetland's backup volume was full before the experiment, as $V_{ET} > V_{tot}$ in the vast majority of cases, there is a strong risk of water scarcity during various dry periods throughout the year. This is a contrasting result with a reported average wetland transpiration equal to 13–42% of inflow (Eger et al., 2017); beyond obvious variability between study systems, this discrepancy might be mainly due to the different methods used for plant transpiration assessment (modelling, PET, evaporation pan in the synthesized articles, but no direct measurement).

3.3.3. General discussion

The transpiration values we measured on the field – up to 58 mm d^{-1}, Figs. 4 and 5 – illustrate quite well the diversity of transpiration values summarized in the literature (Anda et al., 2015; Licata et al., 2019; Pedescoll et al., 2013; Sanchez et al., 2016), that ranged between 0.1 and 57 mm d^{-1} and were measured differently (direct measurement, pan evaporation, computation with Penman-Monteith formula) in various kinds of wetlands. Additionally, the numbers we present here are conservative: we sampled only macrophytes as they represented the dominant plant type inside the CTW, and we did not take night transpiration into account. It is thus very likely that the transpiration effect we show in this study would be magnified if every sources of transpiration could be similarly assessed: instant volume reduction would be higher in wastewater handling CTWs, and water scarcity would be more pronounced in stormwater handling CTWs.

The non-significance of plant effect on biomass and transpiration rate (Sections 3.1. and 3.2.1.) remains surprising, as this variability was largely reported in the literature (Busch, 2000; Licata et al., 2019; Pedescoll et al., 2013; Sanchez et al., 2016; Tuttolomondo et al., 2016) and can lead to a difference of more than one order of magnitude between transpiration values (e.g., Sanchez et al., 2016). This may stem from the combination of two factors: the comparatively greater effects of i) climatic variability and ii) CTW features (hydrological, hydraulic and chemical). An additional artifact might be the fact that in French CTWs, single plant type is often the rule, but does not allow for *in situ* comparative measurements.

We sampled different types of treatment wetlands (Table 1): 2 out of 5 receive partially treated wastewater, 3 out of 5 receive stormwater; one displays surface flow, while the remaining systems display subsurface flows. Yet globally, TR mainly correlated with season (and actually T & HR, section 3.2.1. and Fig. SM2), displaying strong differences between summer and the other seasons, spring and fall being closer than summer due to similar climatic parameters (Fig. 6a, individual map of the Hill-Smith analysis described in Section 3.2.1.). So, seasonality in TR was driven by weather dynamics more than by nature of inflowing water of hydraulics in the systems. Even if hydraulics of the systems is not the main driver of transpiration, we can see from the map of individuals (Fig. 6b) that subsurface-flow systems (E, S1, S2, S3) form a close group,

Table 5
Evapotranspiration and water budget parameters for stormwater management systems. TR field values are obtained by combining field values of TR ($L.gdw.h^{-1}$) and biomass ($gdw.m^{-2}$); T values are obtained by combining field values of TR ($mm.h^{-1}$) and wetland surfaces (m^2); $V_{ET} = V_{evaporated} + V_{transpired}$ is the annual evapotranspired volume; V_{in} is the yearly water volume entering the system; $V_{tot} = V_{in} + V_{backup}$ where V_{backup} is the water reservoir volume at the bottom of the wetlands. Numbers in parentheses correspond to value range computed with model RMSE values (cf. Material and Methods).

	Season	TR ($mm.h^{-1}$)	V_{ET} (m^3)	V_{in} (m^3)	V_{ET}/V_{backup} (−)	V_{ET}/V_{in} (−)	V_{ET}/V_{tot} (−)
S1	spring	0.07 ± 0.05	0 + 132 (101–171)	76.3	23 (17–29)	1.7 (1.3–2.2)	1.6 (1.2–2.1)
	summer	1.11 ± 0.35					
	fall	0.31 ± 0.08					
S2	spring	0.09 ± 0.06	0 + 413 (178–790)	111.3	13 (6–25)	3.7 (1.6–7.1)	2.9 (1.3–5.6)
	summer	0.16 ± 0.08					
	fall	1.12 ± 0.34					
S3	spring	0.03 ± 0.03	0 + 113 (50–210)	63.4	17 (8–32)	1.8 (0.8–3.3)	1.6 (0.7–3.0)
	summer	1.57 ± 0.78					
	fall	1.91 ± 0.59					

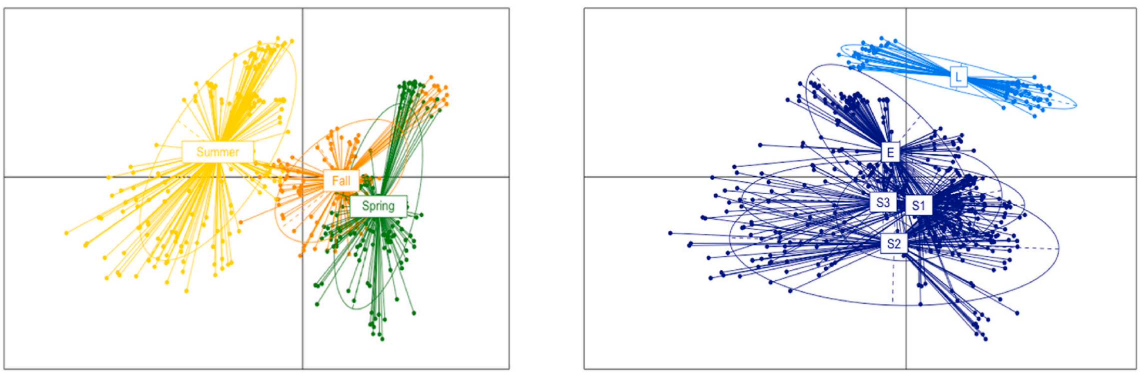

Fig. 6. Individuals map for Hill-Smith analysis. Coloring from seasons (6a) and from hydraulic features (6b).

away from the only surface-flow system we studied (L). Water type does not play a significant role in this clustering, but its main effect lies in biomass production, as discussed before (cf. Section 3.1. and Table 3).

The impact of plant transpiration on the water budget depends on the type of water feeding involved: for continuously fed systems (i.e., the wastewater managing ones), the impact is strong on an hourly basis, as in (Borin et al., 2011) where transpiration accounts for 0.7–7.8 mm d^{-1}, (respectively 0.1–5.1 mm d^{-1}) compared to 5.2 mm d^{-1}, (respectively 3.6 mm d^{-1}) inflow. It is nevertheless levelled over the year, as shown in our study and in (Headley et al., 2012) for instance, where transpiration annually represents only 6–10% of incoming water in pilot-scale wetlands receiving horticulture wastewater. No water scarcity can be endured by plants in these systems, even during the hottest times of the year, as the inflow is almost continuous. On the contrary, for stochastically fed systems (i.e., the stormwater managing ones), the impact of plant transpiration is greater on a yearly basis. As $V_{ET} > V_{backup} + V_{tot}$, the upscaling method is even wrong at this point, as this is physically impossible. What could probably explain this is that plants just stop transpiring when water runs out of bioavailability, and thus endure water scarcity and probably stress for part of the year. Let us underline that this may be an asset more than a drawback, though, if volume reduction is the objective, as significantly achieved in other types of UEI (Ebrahimian et al., 2019).

Finally, an important point is that when plants transpire water, they are also drawing more water, nutrients and pollutants into the soils where the processing takes place (Beebe et al., 2014; Martin et al., 2003). Thus, plants provide a dual role in CTWs: 1) they take up nutrients and pollutants directly (Dodgen et al., 2015) and 2) they draw more nutrients/pollutants into soils for microbial processing when they draw water out of the soils via transpiration. This is particularly striking in this study when we compare stormwater-with wastewater CTWs: the formers have less water, so the downward movement of water and its dissolved content into the soils will be less. But the treatment efficiency expected from these systems is much lower than the one expected from wastewater CTWs, as stormwater is less loaded with nutrients and pollutants than wastewater is. Conversely, higher transpiration in wastewater CTWs participates in better processing of elements, by keeping them longer in the system.

3.4. Management and design recommendations

The transpiration results obtained in this study, and the subsequent effects on hydraulics and treatment efficiency, are likely to change with the ability of planted macrophytes to transpire. Actually, we recorded biomass and transpiration rates for two macrophyte species living in the same wetland, in a site directly aside site L (Table SM4). It appears that during this study, *P. australis* produced more biomass and transpired more than *T. domingensis*. Though this result is not readily expendable to all constructed treatment wetlands as biomass production and transpiration are driven by numerous factors, it appears that the choice of the macrophyte to be settled in CTWs is not neutral and could generate both benefits and costs to the system efficiency.

Sufficient water amount left to ensure flooding conditions is essential for stormwater wetland. Indeed, water scarcity will have an impact on wetland macrophytes (De Wilde et al., 2014; Luo et al., 2008) and might generate the release of chemical compounds (Walaszek et al., 2018) as well. In the stormwater CTWs studied here (S1 to S3), it seems that this backup volume is too small to prevent drought within the system, and the systems may endure extended drought periods throughout the year. Hence, an important point when designing stormwater treatment wetlands would be to ensure correct sizing of this reservoir given rainfall, watershed, climatic parameters and wetland species involved.

Treatment efficiency is generally correlated with the hydraulic residence time within the CTW (Kadlec and Wallace, 2008). And generally, reflections seem to be carried in terms of water residence time rather than in terms of water budget. Most residence time calculations are indeed based only on the ratio between system volume and inflow, and seldom account for transpirational losses of water that alter the residence time within the system. For instance, in the case of site E, the hourly transpiration rate peaks at 26% of the incoming flow during the warmest days of summer; increasing the size of the wetland to supposedly get longer residence time would actually result in having very few water flowing out, because of transpiration. This would occur at the time when streams are in a low-flow regime and thus most sensitive to pollution; the global effect would be mostly detrimental.

On the other hand, fostering transpiration could enhance treatment efficiency, as water and its solutes will be diverted from main flow towards a highly active zone – the root zone.

It eventually seems that size and design of CTW may be chosen not only in terms of water residence but also in terms of influence of transpiration on the water budget. The sizing of the CTW would in this case be guided by its purpose, e.g., no, maximal or regulated outflow.

4. Conclusion

The global objective of the study was to question the influence of transpirational water losses on CTW water budget for a variety of design and time of the year. Daily transpiration is shown to be strongly affected by both season and site configuration, with a milder effect of site. As transpiration depends on both transpiration rate and biomass, a strong seasonal effect is also observed. Globally, the hydraulic feature (surface or subsurface flow) of the CTW influences the transpirational losses of the system. The influence of plant species, although undeniable, could be not observed in our study due to contextual reasons (single plant type in each study site).

Subsequently, transpiration significantly impacts on the water budget of CTWs, even for continuously fed systems because of their high biomass (wastewater handling wetlands). The effect can even become quite dramatic for stormwater wetlands, with all available water being virtually transpired over the vegetative season. Eventually, it appears that plant transpiration may result in sensible flow reduction for wastewater CTW during the warmest times of the year, and in substantial water scarcity for stormwater CTW. From a management perspective, it seems thus strongly advisable to adjust the design of such systems depending on the pursued objectives, be it the quality (emphasis on treatment efficiency) or the quantity (emphasis on flow regulation) of outlet water.

To go further, it would be interesting to assess if water scarcity in stormwater CTWs mandatorily leads to plant stress, and to perform modelling scenarios to assess comparative effects of sizing (residence time vs. transpiration) and design (open water vs. marsh zone vs. subsurface zone) on the functioning of constructed treatment wetlands.

Credit author statement

Paul Bois: Conceptualization, Methodology, Validation, Formal analysis, Investigation, Data curation, Software, Writing – original draft, Writing – review & editing, Funding acquisition.Daniel Childers: Conceptualization, Methodology, Validation, Formal analysis, Investigation, Data curation, Writing – review & editing, Funding acquisition. Milena Walaszek: Writing – review & editing. Adrien Wanko: Validation, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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References

Adcock, P.W., Ganf, G.G., 1994. Growth characteristics of three macrophyte species growing in a natural and constructed wetland system. Water Sci. Technol. 29 (4), 95–102.
Anda, A., Soos, G., da Silva, J.A.T., Kozma-Bognar, V., 2015. Regional evapotranspiration from a wetland in Central Europe, in a 16-year period without human intervention. Agric. For. Meteorol. 205, 60–72.
Beebe, D.A., Castle, J.W., Molz, F.J., Rodgers Jr., J.H., 2014. Effects of evapotranspiration on treatment performance in constructed wetlands: experimental studies and modeling. Ecol. Eng. 71, 394–400.
Bialowiec, A., Albuquerque, A., Randerson, P.F., 2014. The influence of evapotranspiration on vertical flow subsurface constructed wetland performance. Ecol. Eng. 67, 89–94.
Bois, P., Beisel, J.-N., Heitz, C., Katinka, L., Laurent, J., Pierrette, M., Walaszek, M., Wanko, A., 2019. Integrated blue and green corridor restoration in Strasbourg: green toads, citizens, and long-term issues. Ecological Wisdom Inspired Restoration Engineering. Springer, pp. 151–169.
Bois, P., Childers, D.L., Corlouer, T., Laurent, J., Massicot, A., Sanchez, C.A., Wanko, A., 2017. Confirming a plant-mediated “Biological Tide” in an aridland constructed treatment wetland. Ecosphere 8 (3), e01756. https://doi.org/10.1002/ecs2.1756.
Borin, M., Milani, M., Salvato, M., Toscano, A., 2011. Evaluation of Phragmites australis (cav.) trin. Evapotranspiration in northern and southern Italy. Ecol. Eng. 37 (5), 721–728.
Brown, R.G., Stark, J.R., Patterson, G.L., 1988. Ground-water and surface-water interactions in Minnesota and Wisconsin wetlands. The Ecology and Management of Wetlands. Springer, pp. 176–180.
Busch, J., 2000. Canopy transpiration rates in eutrophic wetlands dominated by sedges (Carex spp.) differ in a species specific way. Phys. Chem. Earth - Part B Hydrol., Oceans Atmos. 25 (7), 605–610. https://doi.org/10.1016/S1464-1909(00)00072-1.
Childers, D.L., Iwaniec, D., Rondeau, D., Rubio, G., Verdon, E., Madden, C., 2006. Primary productivity in Everglades marshes demonstrates the sensitivity of oligotrophic ecosystems to environmental drivers. Hydrobiologia 569 (1), 273–292.
Childers, Daniel L., Bois, P., Hartnett, H.E., McPhearson, T., Metson, G., Sanchez, C.A., 2019. Urban Ecological Infrastructure: an inclusive concept for the non-built urban environment. Elem. Sci. Anth. 7.
Consoli, S., Milani, M., Cirelli, G., Barbagallo, S., Marzo, A., Vanella, D., Toscano, A., 2018. Energy and water balance of a treatment wetland under mediterranean climatic conditions. Ecol. Eng. 116, 52–60. https://doi.org/10.1016/j.ecoleng.2018.02.029.
Daoust, R.J., Childers, D.L., 1998. Quantifying aboveground biomass and estimating net aboveground primary production for wetland macrophytes using a non-destructive phenometric technique. Aquat. Bot. 62 (2), 115–133.
De Wilde, M., Sebei, N., Puijalon, S., Bornette, G., 2014. Responses of macrophytes to dewatering: effects of phylogeny and phenotypic plasticity on species performance. Evol. Ecol. 28 (6), 1155–1167.
Directive, W.F., 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. Off. J. Eur. Commun. 22 (12), 2000.
Dodgen, L.K., Ueda, A., Wu, X., Parker, D.R., Gan, J., 2015. Effect of transpiration on plant accumulation and translocation of PPCP/EDCs. Environ. Pollut. 198, 144–153.
Ebrahimian, A., Wadzuk, B., Traver, R., 2019. Evapotranspiration in green stormwater infrastructure systems. Sci. Total Environ. 688, 797–810. https://doi.org/10.1016/j.scitotenv.2019.06.256.
Eger, C.G., Chandler, D.G., Driscoll, C.T., 2017. Hydrologic processes that govern stormwater infrastructure behaviour. Hydrol. Process. 31 (25), 4492–4506. https://doi.org/10.1002/hyp.11353.
Eichelmann, E., Hemes, K.S., Knox, S.H., Oikawa, P.Y., Chamberlain, S.D., Sturtevant, C., Verfaillie, J., Baldocchi, D.D., 2018. The effect of land cover type and structure on evapotranspiration from agricultural and wetland sites in the Sacramento–San Joaquin River Delta, California. Agric. For. Meteorol. 256, 179–195.
Engloner, A.I., 2009. Structure, growth dynamics and biomass of reed (Phragmites australis)–A review. Flora-Morphology, Distribution, Functional Ecology of Plants 204 (5), 331–346.
Federal Water Pollution Control Act, 2002.
Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., Briggs, J.M., 2008. Global change and the ecology of cities. Science 319 (5864), 756–760.

Grove, J.M., 2009. Cities: managing densely settled social-ecological systems. Principles of Ecosystem Stewardship. Springer, pp. 281–294.
Hadad, H.R., Maine, M.A., Bonetto, C.A., 2006. Macrophyte growth in a pilot-scale constructed wetland for industrial wastewater treatment. Chemosphere 63 (10), 1744–1753. https://doi.org/10.1016/j.chemosphere.2005.09.014.
Headley, T.R., Davison, L., Huett, D.O., Müller, R., 2012. Evapotranspiration from subsurface horizontal flow wetlands planted with Phragmites australis in sub-tropical Australia. Water Res. 46 (2), 345–354.
Kadlec, R.H., Wallace, S., 2008. Treatment Wetlands. CRC press.
Laurent, J., Bois, P., Nuel, M., Wanko, A., 2015. Systemic models of full-scale surface flow treatment wetlands: determination by application of fluorescent tracers. Chem. Eng. J. 264, 389–398.
Licata, M., Gennaro, M.C., Tuttolomondo, T., Leto, C., La Bella, S., 2019. Research focusing on plant performance in constructed wetlands and agronomic application of treated wastewater–A set of experimental studies in Sicily (Italy). PLoS One 14 (7), e0219445.
Luo, W., Song, F., Xie, Y., 2008. Trade-off between tolerance to drought and tolerance to flooding in three wetland plants. Wetlands 28 (3), 866.
Martin, J., Hofherr, E., Quigley, M.F., 2003. Effects of Typha latifolia transpiration and harvesting on nitrate concentrations in surface water of wetland microcosms. Wetlands 23 (4), 835–844.
McLaughlin, D.L., Cohen, M.J., 2014. Ecosystem specific yield for estimating evapotranspiration and groundwater exchange from diel surface water variation. Hydrol. Process. 28 (3), 1495–1506.
Melosi, M.V., 2008. The Sanitary City: Environmental Services in Urban America from Colonial Times to the Present. University of Pittsburgh Pre.
Miller, R.C., Zedler, J.B., 2003. Responses of native and invasive wetland plants to hydroperiod and water depth. Plant Ecol. 167 (1), 57–69. https://doi.org/10.1023/A:1023918619073.
Molle, P., Liénard, A., Boutin, C., Merlin, G., Iwema, A., 2005. How to treat raw sewage with constructed wetlands: an overview of the French systems. Water Sci. Technol. 51 (9), 11–21. https://doi.org/10.2166/wst.2005.0277.
Nivala, J., Headley, T., Wallace, S., Bernhard, K., Brix, H., van Afferden, M., Müller, R.A., 2013. Comparative analysis of constructed wetlands: the design and construction of the ecotechnology research facility in Langenreichenbach, Germany. Ecol. Eng. 61, 527–543.
Owen, C.R., 1995. Water budget and flow patterns in an urban wetland. J. Hydrol. 169 (1–4), 171–187. https://doi.org/10.1016/0022-1694(94)02638-R.
Pedescoll, A., Sidrach-Cardona, R., Sánchez, J.C., Bécares, E., 2013. Evapotranspiration affecting redox conditions in horizontal constructed wetlands under Mediterranean climate: influence of plant species. Ecol. Eng. 58, 335–343.
R Core Team, 2020. R: A Language and Environment for Statistical Computing, vol. 887. R Foundation for Statistical Computing, Vienna, Austria, p. 888. https://www.
Sanchez, C.A., Childers, D.L., Turnbull, L., Upham, R.F., Weller, N., 2016. Aridland constructed treatment wetlands II: plant mediation of surface hydrology enhances nitrogen removal. Ecol. Eng. 97, 658–665.
Sánchez-Carrillo, S., Álvarez-Cobelas, M., Benítez, M., Angeler, D.G., 2001. A simple method for estimating water loss by transpiration in wetlands. Hydrol. Sci. J. 46 (4), 537–552.
Sanderson, J.S., Cooper, D.J., 2008. Ground water discharge by evapotranspiration in wetlands of an arid intermountain basin. J. Hydrol. 351 (3–4), 344–359.
Schmitt, N., Wanko, A., Laurent, J., Bois, P., Molle, P., Mosé, R., 2015. Constructed wetlands treating stormwater from separate sewer networks in a residential Strasbourg urban catchment area: micropollutant removal and fate. J. Envion. Chem. Eng. 3 (4, Part A), 2816–2824. https://doi.org/10.1016/j.jece.2015.10.008.
Treese, S., Childers, D.L., Sanchez, C.A., 2020. Long-Term Trends in Nitrogen Removal by an Aridland Constructed Treatment Wetland. Wetlands. https://doi.org/10.1007/s13157-020-01376-4.
Tuttolomondo, T., Leto, C., La Bella, S., Leone, R., Virga, G., Licata, M., 2016. Water balance and pollutant removal efficiency when considering evapotranspiration in a pilot-scale horizontal subsurface flow constructed wetland in Western Sicily (Italy). Ecol. Eng. 87, 295–304.
Vymazal, J., Krőpfelová, L., 2005. Growth of Phragmites australis and Phalaris arundinacea in constructed wetlands for wastewater treatment in the Czech Republic. Ecol. Eng. 25 (5), 606–621. https://doi.org/10.1016/j.ecoleng.2005.07.005.
Walaszek, M., Bois, P., Laurent, J., Lenormand, E., Wanko, A., 2018a. Micropollutants removal and storage efficiencies in urban stormwater constructed wetland. Sci. Total Environ. 645, 854–864.
Walaszek, M., Bois, P., Laurent, J., Lenormand, E., Wanko, A., 2018b. Urban stormwater treatment by a constructed wetland: seasonality impacts on hydraulic efficiency, physico-chemical behavior and heavy metal occurrence. Sci. Total Environ. 637, 443–454.
Walaszek, M., Del Nero, M., Bois, P., Ribstein, L., Courson, O., Wanko, A., Laurent, J., 2018. Sorption behavior of copper, lead and zinc by a constructed wetland treating urban stormwater. Appl. Geochem. 97, 167–180.
Wallace, S.D., Knight, R.L., 2006. Small-scale Constructed Wetland Treatment Systems: Feasibility, Design Criteria, and O&M Requirements, Final Report. Water Environment Research Federation, USA, p. 304.
Weller, N.A., Childers, D.L., Turnbull, L., Upham, R.F., 2016. Aridland constructed treatment wetlands I: macrophyte productivity, community composition, and nitrogen uptake. Ecol. Eng. 97, 649–657.
WHO, 2011. Guidelines for Drinking-Water Quality, vol. 398.
Zhou, L., Zhou, G., 2009. Measurement and modelling of evapotranspiration over a reed (Phragmites australis) marsh in Northeast China. J. Hydrol. 372 (1–4), 41–47.