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Chapter 4

THE INFLUENCE OF EVAPOTRANSPIRATION ON WASTEWATER-CONSTRUCTED WETLAND TREATMENT EFFICIENCY

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Abstract

Owing to low investment and maintenance costs, there has been a growing interest in applying plants in wastewater treatment. Plants commonly used in constructed wetlands (CW) include: cattail, reed, rush, yellow flag, manna grass, and willow.

In a CW, application of plants brings several benefits: creating aerobic conditions in the otherwise anaerobic rhizosphere, providing carbon compounds into the rhizosphere, uptaking pollutants (e.g., nutrients and heavy metals) from treated wastewater; improving the hydraulic conditions of wastewater flow through CW beds, and also increasing the available surface for growth of microbial biofilms. Hydrophytes also have great transpiration potential. Numerous studies have shown the importance of evapotranspiration during hot periods in natural wetlands and also in constructed wetlands. Evapotranspiration affects treatment efficiency in CWs: it increases the concentration of dissolved compounds due to decreasing water volume. Therefore, having regard to the mode of operating (VSSW or HSSW), temperature and influent characteristics (e.g., HLR and wastewater influent loads), the removal efficiency calculated as a comparison between initial and final concentration is lower, than expected from mass balance. Given results from systems in colder (Poland) and warmer (Portugal) climate conditions shows that the difference in methodology of removal efficiency calculation is significant, even if the CWs are operating in different modes. Usually, in the literature removal efficiency is expressed on the basis of concentrations, mostly due to lack of flow rate monitoring. Unfortunately, this may seriously underestimate treatment performance of CWs. This study suggests the need for routine monitoring of flow rate, or evaluation of potential evapotranspiration, to estimate removal efficiency of a CW based on mass balance

Keywords: Constructed wetlands, evapotranspiration, treatment efficiency, concentration, load, water balance

1. Introduction

1.1. Natural and Constructed Wetlands

Wetland ecosystems are characterised by the presence of standing water at or above the soil surface for all or part of the growing season. They form ecotones between terrestrial and aquatic systems and act as sources, sinks and transformers of nutrients and carbon (Mitsch and Gosselink, 2007). The importance of natural wetlands for nature conservation and for the ecosystem services they provide (e.g., water purification, hydrological buffering, production of valuable plant and animal products), has long been recognised (Haslam, 2003). Wetland plants (helophytes) are adapted to tolerate saturated, anoxic soils and because of ample supplies of light, water and nutrients in wetlands, rates of primary productivity and heterotrophic activity are typically high (Brix, 1997). Water plants are more productive than terrestrial ones due to greater resistance to environmental changes and high photosynthesis efficiency (Tchobanoglous, 1987). The use of ecological systems such as constructed wetlands (CWs), is recognised as an economical and technically sustainable solution for wastewater treatment making it safe to discharge into the environment (Christensen, et al.; 1992, Schnoor, et al.; 1995).

CWs are artificial complexes of water, matrix, vegetation and the associated invertebrate and microbial communities designed to simulate the ability of natural wetlands to remove pollutants from water (Brix, 1997; Kangas, 2004), being a good example of ecological engineering (Mitsch and Jorgensen, 2004). This method for treating a variety of wastewaters such as sewage, landfill leachate, mine leachate, urban storm-water, agricultural run-off, is comparatively simple to design, construct, and operate (e.g., Moshiri, 1993; Vymazal, 2005; Randerson, 2006; Randerson et al., 2007; Kadlec and Wallace, 2008). CWs are also suitable for advanced and polishing treatment if water reuse is an option (Masi and Martinuzzi, 2007; Marecos do Monte and Albuquerque, 2010; Pedrero et al., 2011).

Plants commonly used in constructed wetlands include: cattail (*Typha latifolia* L.), reed (*Phragmites australis* Trin ex Steudel), rush (*Juncus effusus* L.), yellow flag (*Iris pseudacorus* L.), mannagrass (*Glyceria maxima*), and giant reed (*Arundo donax L.*). Many authors proposed also willows (*Salix sp.*) to be used in CWs with high efficiency (Aronsson, 2000; Aronsson and Perttu, 2001; Bialowiec et al., 2007; Duggan, 2005; Elowson, 1999; Kowalik and Randerson, 1994; Perttu and Kowalik, 1997; Randerson, 2006).

Vymazal and Kropfelova (2008) showed that willow treatment systems also can achieve zero discharge of water due to evapotranspiration (ET).

1.2. Pollutant Removal in Constructed Wetlands

Aquatic macrophytes are adapted to grow in anoxic substrates by developing aerenchyma tissue which enables transfer of atmospheric oxygen via stems to roots, much of which then leaks into the substrate (Brix, 1997; Williams et al., 2010a; Williams et al., 2010b; Randerson et al., 2011). Oxygen released from roots creates local aerobic conditions and changes the redox status in the otherwise anoxic or anaerobic rhizosphere, which induces growth of aerobic heterotrophic and autotrophic bacteria (nitrifiers), and the aerobic breakdown of organic material (Sorrel and Armstrong, 1994; Brix, 1997; Bialowiec et al., 2012a). Hence the plant root-soil interface plays a significant role in the removal of pollutants, especially of nitrogen (N) from CWs (Reddy et al., 1989; Mesquita et al., 2013).

The release of oxygen into the rhizosphere is related to some extent to photosynthesis, light intensity, stomata aperture, and temperature, as well as to the plant species concerned (Stein and Hook, 2005). Removal of ammonia through nitrification is advantageous not only because it helps to reduce N loads, but also because the final N specie (NO_3^-), is much less toxic and more bioavailable to plants than NH_4^+ (Jones et al., 2006).

Plant uptake also plays an important role in increasing N removal, especially in treatment wetlands containing fast-growing plants such as willows (Aronsson, 2000; Randerson, 2006). Organic compounds, such as sugars, alcohols and acids that are in the wastewater or released by plants into the rhizosphere can help in nitrate removal by acting as a carbon source for denitrifying bacteria (Brix, 1997).

The majority of microbial processing that occurs in wetlands is attributed to biofilms made up of communities of algae, bacteria, protozoa and invertebrates. It has been shown that up to 90% of organic and inorganic N can be removed from wastewater by biofilms (Welander and Henrysson, 1998; Albuquerque et al., 20012).

To conclude, the plants in a CWs bring following benefits: plants may create aerobic conditions in an otherwise anaerobic rhizosphere, which increase heterotrophic and autotrophic aerobic bacteria biodiversity; plants can provide carbon compounds into the rhizosphere that may be utilized for microbiological processes; plants may uptake pollutants (N, P, heavy metals) from treated wastewater; plants may improve the hydraulic conditions of wastewater flow through the CWs bed, and also may increase the available surface for biofilm growth.

Macrophtyes have also a great potential for water loss through transpiration (Bialowiec and Wojnowska-Baryla, 2007; Headly et al., 2012). Thus, water losses to the atmosphere via ET, can be high (Borin et al., 2011), especially under warm and windy conditions (Bialowiec et al., 2006; Albuquerque et al., 2009a).

Numerous studies have shown the importance of ET during hot periods in both natural and CW systems. ET may affect treatment efficiency of CWs by reduction of wastewater volume passing through the system (Bialowiec et al., 2006).

Usually, treatment efficiency of wastewater in CWs is calculated based on the comparison of concentration of pollutants in inlet and outlet without considering the contribution of evapotranspiration in the water balance. This approach is used in conventional wastewater treatment plants. In a CWs, where water loss is typically high, the calculation of pollutant removal efficiency using results of concentrations may lead to significant errors. With ET, the concentration of dissolved compounds increases due to decreasing water volume, hence removal efficiencies calculated with and without the water balance are not the same, and this problem of difference is assessed and discussed below.

This chapter presents an overview of the importance of ET in CWs, comparing different methods of treatment efficiency calculations and discussing the influence of ET on removal rates taking in account different both operating modes and climates conditions. Two case studies are presented, one from field work experiments carried out in an horizontal flow system under hotter conditions (work developed in Portugal), and second, as a comparison from laboratory scale experiments developed in a vertical flow system under colder conditions (work developed in Poland).

1.3. Evapotranspiration of Reed and Willow Systems

Evapotranspiration (ET), includes water which vaporises from soil (evaporation), and moisture which passes through vascular plants to the atmosphere (transpiration), (Mitsch and Gosselink, 2007), along a gradient of decreasing water potential, constrained by resistances encountered in the soil, plant and air (Stewart, 1984).

Assuming an adequate water supply to wetland plants, the rate of ET is proportional to the difference in vapour pressure (water potential in air), between the wet soil or leaf surfaces and that in the overlying air. The water flux is affected by factors including wind speed, air and leaf surface temperatures and air humidity. Hence ET varies with season, typically being highest in summer.

Evapotranspiration rate (ET_r) is described by Equation (1) (Mitsch and Gosselink, 2007).

$$ET_r = c \cdot f(u) \cdot (e_w - e_a) \tag{1}$$

where ET_r ($L\ T^{-1}$) is the rate of ET, c the mass transfer coefficient ($L\ T^2\ M^{-1}$), f(u) is a function of wind speed ($L\ T^{-1}$), and e_w and e_a are vapour pressures at the water/plant surface, and surrounding air, respectively. Fundamental units are presented for explaining equations. However, common units are used throughout the text.

ET_r is increased by meteorological factors, which steepen the water potential gradient such as increased solar radiation and leaf surface temperature (increasing saturation vapour pressure at the evaporating surface), and by decreased air humidity or increased wind speed (reducing vapour pressure close to the leaf surface). ET may be limited by plants closing their stomata during periods of stress due to root anoxia, the presence of toxic substances or high salinity in the substrate (Mitsch and Gosselink, 2007). Hence actual ET is usually less than potential ET (measured as evaporation from a water-filled pan, E_{pan}), depending on variation due to wind and shade effects, or on the physiological state of the vegetation. The physical composition of the substrate (e.g., particle size, specific surface area, water absorption and permeability), may also affect ET either directly or indirectly via plant growth.

ET from wetlands may be predicted empirically by, for example, equations of Thornthwaite, Penman, or Hammer and Kadlec (Mitsch and Gosselink, 2007), which provide consistent estimates based on meteorological variables. The method of Thornthwaite requires only values of average monthly air temperature. To allow for differences between vegetation types a local estimate of potential evaporation (E_{pan}), or reference crop ET, may be multiplied by an appropriate crop coefficient K_c, an approach used in predicting agricultural irrigation need (Kadlec and Wallace, 2008).

However, specific water balance data are required to calibrate K_c for local conditions. In a CWs (where it may be assumed that there is no loss of water by seepage and zero net storage over a period), ET_r ($L\ T^{-1}$) may be calculated simply by difference between measured fluxes of water, as in Equation (2) (Surface et al., 1993):

$$ET_r = Q_{inf} + P - Q_{eff} \tag{2}$$

where Q_{inf} is the influent flow rate ($L\ T^{-1}$) over the CW area, P the precipitation ($L\ T^{-1}$), and Q_{eff} the effluent flow rate ($L\ T^{-1}$) over the CW area.

Annual water budgets, measured in a variety of natural wetland habitats show a range of values for ET_r between 40 and 1500 mm y^{-1}, reflecting regional differences in climate (precipitation and solar radiation), and vegetation type (Table 1).

Small-scale wetland areas such as CWs, frequently show enhanced ET_r (the so-called "clothesline" and "oasis" effects), due to a strong influence of advection from the relatively warm, dry air of the surrounding terrestrial microclimate (Borin et al., 2011; Kadlec, 1989; Kadlec and Wallace, 2008). High rates of water loss from treatment wetlands in central Europe, planted with reed, are quoted by Siuta (1996), 1000 to 1400 mm y^{-1}, Wieuner et al. (1999), approximately 1500 mm y^{-1}, and for sludge drying reed beds in Denmark by Nielsen (1993), approximately 1500 mm y^{-1}. Borin et al. (2011), found ET_r for reed bed CWs in Italy to be 6 to 7 times higher than that for a reference crop (based on Penman calculations), when averaged over the growing season.

ET varies greatly with season in temperate regions, and with maturity of the vegetation stand, since it is positively correlated with plant stem/leaf biomass. The effect of vegetation is particularly marked in subsurface flow CWs and in sludge drying reed beds. In a study of ET_r in reed CWs beds treating landfill leachate.

Surface et al. (1993), found the greatest decrease in leachate volume (43%), occurred between May and July in the second year of growth when stem biomass was greater, whereas volume reduction of only 21% occurred in the winter period (from October to April).

Table 1. Examples of annual ET values in natural wetlands

ET_r [mm y^{-1}]	Location of wetland	Reference
40	Fen, N. Wales, UK	Gilman, 1982
64	Prairie pothole, N. Dakota	Shjeflo, 1968
67	Swamp, N. Carolina	Richardson, 1983
72	Swamp, S. Illinois	Mitsch, 1979
86-99	Swamp, Florida	Pride et al., 1966
93	Okefenokee swamp, Georgia	Rykiel, 1984
102	Bog, Massachusetts	Hemond, 1980
1030	Okavango delta, Botswana	Wolski, in Mitsch and Gosselink, 2007
1100	Wetland, Victoria, Australia	Raisin, 1999
1524	Valley wetland, US	Williams et al., 1987

Extremely high average ET_r of 40 and 57 mm d^{-1} respectively for giant reed and common reed planted CWs in Morocco were measured over the winter to spring period, compared with 7 mm day^{-1} for an unplanted gravel bed (El Hamouri et al., 2007). In a Danish sludge drying bed water was lost from the fully grown reed stand at 25 mm d^{-1} on hot, summer days, the reduction in sludge volume being 3 times greater in summer than in winter, and it was 2 times greater in the 2nd year of the study due to increased reed biomass (Nielsen, 1993). Even in cool, wet summers ET greatly exceeded summer rainfall.

In horizontal subsurface flow (HSSW) reed beds treating horticultural runoff in sub-tropical Australia, average ET_r values increased from 7 mm d^{-1} to 10.6 mm d^{-1} in the 2nd year in response to growth of the plants (Headley et al., 2012). ET_r were more variable than corresponding pan evaporation data (ratios ET/E_{pan} 1.9 in the first year, and 2.6 in the second year), reflecting the important effect of vegetation. In the hot, dry, windy summer climate of north-western US, average water loss by ET_r from sludge-drying reed beds (between May and June), was 6.4 mm d^{-1} (similar to that for local hay crops), which amounts to only 10% of total hydraulic loading (5 710 mm d^{-1}), (Burgoon et al., 1997). This circumstance was due to water stress by the reeds, causing increased resistance to water flux through the soil to the plant roots and increased stomata resistance.

Transpiration by reeds and other macrophytes such as willows can provide a cost-effective method for dewatering sludge and slurry residues, alternative to mechanical systems (Nielsen, 1990). Water in raw sewage sludge (typically with 5% of dry matter), consists of pore water (67%), capillary water (25%), and absorption/bound water (8%), (Nielsen, 1993). Removal of pore water occurs through gravity drainage (increasing dry matter to only 15%).

In the field, ET by reed can easily remove capillary water (raising dry matter to 50%). The remaining water content is unavailable, as it is held at a water potential greater (more negative), than the roots and rhizomes can generate, hence 50% dry matter is the usual goal for sludge dewatering in CWs. ET accounted for 80% of the total dewatering (1500 mm after 260 days), close to potential ET for the climate. The SALIMAT technique developed by De Vos (1994), laid out a mat of willow rods across a sludge bed, which grew into a willow stand effective at dewatering the initial deposit, and later surface additions of sludge over subsequent years. De Maeseneer (1997), points out that high concentration of heavy metals such as zinc and cadmium may occur in willow leaves, depending on the origin of the sludge, which may restrict its after-use in agriculture.

A particularly important aspect of landfill leachate disposal in soil-plant CWs systems is the reduced leachate volume due to ET by planted macrophytes, typically willows, poplars and reeds (Dobson and Moffat, 1995), during periods when rainfall input is relatively low. Agopsowicz (1994), determined that ET by young (3-month), willow sprouts supplied with landfill leachate was 1.6 to 1.8 times higher than the average precipitation rate in Poland, (about 600 mm).

In a 2-year study of a soil-plant system with young willow sprouts (*Salix amygdalina* L.), Bialowiec et al. (2003, 2007), found ET_r values of 2-3 mm d^{-1}, which were up to 5 times higher than for a bare soil surface, and resulted in volume reduction between 80 and 90%. Similar ET_r values were found for reeds in soil-plant systems, at 1 to 3 mm d^{-1} in the 1st year of growth and 3 to 5 mm d^{-1} in the 2nd year, reflecting an increase in reed stem biomass (Bialowiec and Wojnowska-Baryla, 2007).

Where the vegetated CWs area is sufficiently large, there may be no effluent discharge, at least during summer months, for example in a wastewater treatment reed bed in Poland (Bialowiec and Randerson, 2010a; Toczylowska et al., 2000). In a marsh-pond-meadow system with recirculation, implemented for municipal wastewater treatment in Kentucky (Choate et al., 1993), as a result of ET and seepage no outflow occurred over 4 years except following one heavy rain event. In sensitive locations where stringent conditions for effluent quality prevent discharge to environment as in Denmark, zero-discharge wetland systems using willows to reduce effluent volume have been employed (Gregersen and Brix, 2001). Conversely, in areas of low rainfall, the aim may be to minimize ET losses in a CWs, as treated effluent may be required to be re-used (El Hamouri et al., 2007; Green et al., 2006; Headley et al., 2012; Masi and Martinuzzi, 2007).

The use of soil-plant systems for landfill leachate disposal calls for an informed selection of plant species and soil types. US-EPA (2000), recommends willows, poplars and grasses for volume reduction in landfill leachate treatment. Bialowiec and Agopsowicz (2007), tested the tolerance of different willow species to pollutant concentration in landfill leachate. They determined the species *S. amigdalina* L., *S. viminalis* L., *S. purpurea* to have the highest ability for ET and high tolerance of pollutant concentrations in the soil solution.

The use of different kinds of landfill leachate for irrigating short rotation willow plantations has been tested in Sweden (Dimitriou and Aronsson, 2003, 2005; Dimitriou et al., 2006; Hasselgren, 1992), Finland (Ettala, 1992), Great Britain (Alker et al., 2003), and in Poland (Agopsowicz, 1994; Białowiec, 2005; Białowiec and Kasinski, 2009). Young rooted willow plants were able to survive in landfill leachate solutions with electrolytic conductivity (EC), values up to 5 mS cm^{-1} if they were cultivated in high concentrations from the beginning, whereas naïve plants were killed when EC exceeded 3 mS cm^{-1} (Bialowiec and Randerson, 2010b).

Common reed also has a high tolerance to chronic and periodic salinity up to a value of 1.8% (< 2.5% Cl^-), (Michiel et al., 1998). Mauchamp and Mesleard (2001) determined LC50 value of salinity as 1.5% for young reed seedlings and that 2.5% salinity inhibited their growth. Lissner et al. (1999) showed that reed transpiration was reduced to 2838 mm y^{-1}, 50% of that in zero salinity. Total ET_r values for reed plants growing in sand with landfill leachate applied at different hydraulic loading rates, ranged from 624 to 1150 mm y^{-1} in the 1st year, of growth, and from 1419 to 1803 mm y^{-1} in the 2nd year, reflecting an increase in shoot biomass. They are similar to values described by Williams et al., (1987); Raisin et al., (1999); and Wieuner et al., (1999); as 1524, 1100 and 1500 mm y^{-1}, respectively.

1.4. Influence of Climatic Conditions on Treatment Efficiency

Removal efficiencies in CWs systems are normally viewed on an event mean basis, in which flows, concentrations and loads are averaged over the duration of the event.

The event average concentration is defined as the mass of a compound divided by the mass of water involved in the event as in Equation (3) (Kadlec, 2010).

$$C=\frac{\int_1^n \left(Q_{inf(i)} \cdot C_{inf(i)}\right)dt}{\int_1^n Q_{inf(i)}\,dt} \qquad (3)$$

where C is the event average concentration of the compound ($M\,L^{-3}$), $C_{inf(i)}$ the concentration of the compound in the event i ($M\,L^{-3}$), $Q_{inf(i)}$ the incoming flow rate in the event i ($L^3\,T^{-1}$) and n the number of events.

Therefore, the percentage of concentration reduction of a compound (or removal efficiency, RE) may be computed based on the difference between inlet and outlet concentrations divided by the inlet concentrations (Equation (4)).

$$RE = \frac{(C_{inf} - C_{eff})}{C_{inf}} \cdot 100 \tag{4}$$

where C_{inf} and C_{eff} are the average concentrations (M L^{-3}) of a compound in the influent and the effluent, respectively.

Pollutant removal efficiency values above 90% are normally achieved for suspended solids and organic matter (based on biochemical oxygen demand (BOD) and chemical oxygen demand (COD) concentrations) and up to 60% for N (either total nitrogen (TN) or ammonia nitrogen (NH_4-N)) (Vymazal, 2007; Kadlec and Wallace, 2008; Vymazal and Kropfelova, 2008; Vymazal, 2009) in gravel-based beds. However, values under 50% for COD were found in HSSW beds (Table 2) in Israel (Avsara et al., 2007), Portugal (Albuquerque et al., 2009a) and Spain (Osorio, 2006), countries influenced by the Mediterranean climate or moderate continental climate, and associated with the presence of significant concentrations of slowly biodegradable organic matter discharged from livestock facilities.

Table 2. Removal efficiencies for HSSW in warm climate regions

Operating conditions				Removal efficiency (%)				Reference
OLR (g COD m^{-2} d^{-1})	HLR (cm d^{-1})	HRT (d)	SSA (m^2 e.p.$^{-1}$)	COD	TN	NH_4-N	TSS	
9.4-22.3	8.5-13.8	4.8-9.0	2.5	66.7	76.0	78.6	56.4	Albuquerque et al. (2009a)
26.4-52.7	7.3-14.9	2.5-5.0	—	64.2	—	55.1	90.4	Avsara et al. (2007)
2.2-34.1	14.0-15.6	3.0-4.3	1.2	94.0	60.0	85.0	84.0	Masi and Martinuzzib (2007)
18.4-54.5	18.0	3.0	1.0	43.0	—	25.0	73.0	Osorio (2006)
38.1	3.6	5.0	2.3	78.0	35.0	22.6	78.0	El-Khateeb and El-Gohary (2002)
5.0 – 20.0	2.0-20.0	5.0-14.0	3.0-6.0	> 85.0	> 70.0	> 80.0	> 80.0	*Worldwide experience:* EPA (1999), Kadlec and Wallace (2008); Kadlec et al. (2000), Korkusuz (2005), Kowalik et al. (1995), Vymazal and Kropfelova (2008)

OLR: organic loading rate; HLR: hydraulic loading rate; HRT: hydraulic retention time; SSA: specific surface area; e.p.: equivalent-population.

Hunter et al. (2001), Sun and Austin (2007) and Cheng et al. (2011) reported removal efficiencies for TN between 30 and 54% in HSSW operated in cold climate conditions. Avsara et al. (2007) and Albuquerque et al. (2009a) have found ammonia removal efficiencies of 55.1 and 78.6%, respectively in HSSW systems operated in warm climate regions (the average water temperatures were above 20ºC), which show higher nitrification and denitrification rates for higher temperatures.

Pinton et al. (2007) and Bialowiec et al. (2012a) have reported fluctuations in ammonia removal, which was explained by the variation of dissolved oxygen (DO), in the rhizosphere, but also due to organic exudates by roots. These exudates normally include organic N, which hydrolysed to ammonia, thus increasing the concentrations of this inorganic N form in the rhizosphere. Therefore, the oxidation of ammonia occurs faster when the ionized form is available and increases again after a few days due to hydrolysis of organic nitrogen compounds released by the plants. Stecher and Weaver (2003) and Mander et al. (2000) also found high variation in ammonia removal efficiency (from 12 to 85%) in HSFW beds, suggesting that there is a higher influence of temperature and plant density on the activity of nitrifying biomass.

The hydraulic loading rate (HLR) is a critical parameter for the operation of HSSW. The infiltration of stormwater flow into the sewer network may increase the HLR throughout the beds. If loading is too high (>30 cm d^{-1}), the bed tends to clog causing wastewater to back up or hydraulic short-circuiting, which may lead to the discharge of low quality effluents into water streams (Wallace and Knight, 2006).

Amado et al. (2012) found significant infiltration flow rates into the HSSW (96% higher than the design maximum value for the bed) for stormwater runoffs above 4000 m^3 d^{-1}, which led to a high variability of the HLR into the bed. As a result, the removal of organic matter, solids and especially nitrogen was lower than expected and the quality of the final effluent was negatively affected. Therefore, the variation of the HLR and the change in the bed media characteristics may influence the bed's stability and performance. Ammonia is more difficult to remove than organics since nitrifiers are autotrophic microorganisms, which are very sensitive to changes in humidity, temperature and wastewater characteristics and loadings (Kadlec and Wallace, 2008; Vymazal and Kropfelova, 2008). Nitrifiers have a slow respiration rate and stoichiometrically require 4.57 mg O_2 per mg NH_4–N removed (full nitrification to NO_3) and 1.71 mg O_2 per mg NH_4–N removed (partial nitrification to NO_2) (Paredes et al., 2007; Kadlec and Wallace, 2008).

However, Sun et al. (2003) observed that the reduction in level of ammonia in a vertical subsurface flow reed bed was not balanced by increases in nitrite and nitrate contents in the influent.

Therefore, the large differences in removal efficiencies and their variability over time may be related to water losses. Finlayson and Chick (1983) calculated the RE for COD, TSS, TKN and TP (Table 3), taking into account the influent and effluent concentrations (Equation (4)) and the influent and effluent mass loads as in Equation (5).

Percentage removal rates calculated for mass loads are higher than those for concentrations, because Equation (5) is more sensitive to flow rate fluctuations caused by variation in ET.

$$RE=\left(\frac{C_{inf}\cdot Q_{inf}-C_{eff}\cdot Q_{eff}}{C_{inf}\cdot Q_{inf}}\right)\cdot 100 \tag{5}$$

where $C_{inf}*Q_{inf} = M_{inf}$ (influent mass load, M T^{-1}) and $C_{eff}*Q_{eff}= M_{eff}$ (effluent mass load, M T^{-1}).

Nevertheless, Kadlec (2010) states that concentration reduction and mass removals calculated using event average concentrations and mass flows, are inadequate to characterize CW performance under event-driven operation. These two measures did not include internal mixing, storage and flushing, temperature, and inter-event water quality conditions. Nevertheless, they may be used as an approach to quantify CW performance.

Light–expanded clay aggregates (LECA) have been used to improve treatment capacity, which is especially useful in cold climate regions, since they present both higher porosity and specific surface area, which allows a better biofilm adhesion, and may require smaller bed areas than the conventional gravel substrate (Vilpas et al., 2005; Scholz, 2006; van Deun and van Dyck, 2008; Albuquerque et al. 2009b; Bialowiec et al. 2011, Bialowiec et al. 2012b).

Tracer experiments on LECA-based HSSW (Albuquerque, 2012) have showed the presence of stagnated areas and the occurrence of internal recirculation, due to the development of clusters of biomass, roots, rhizomes and solid material, which seems not to interfere with the removal of organic mater and nitrogen.

However, at the start-up these beds may present low performance if upstream infiltration is not controlled as concluded by Amado et al. (2012).

Table 3. Treatment performance based on concentrations and mass loads for HSSW

Compound	Concentrations			Mass loads		
	C_{inf} (mg L^{-1})	C_{eff}(mg L^{-1})	RE (%)	M_{inf} (kg d^{-1})	M_{eff} (kg d^{-1})	RE (%)
COD	642	378	41	35.3	13.8	61
TSS	214	82	62	11.8	3	75
TKN	257	190	26	14.3	7	51
TP	19.8	15.2	23	1.1	0.6	49

Adapted from Finlayson and Chick, 1983.
C_{inf}: influent concentration,
C_{eff}: effluent concentration;
M_{inf}: influent mass load;
M_{eff}: effluent mass load;
RE: removal efficiency.

Therefore, CW beds subject to transient high loads and significant ET variations should be designed to accept organic and solid loads, with the inclusion of advanced primary treatment systems (e.g., filter screens or high-rate clarification), in order to reduce the surface loading rate (Albuquerque et al., 2009a).

In arid areas, salt concentration may increase in the treated effluent to a level which prevents its re-use for irrigation (Morari and Giardini, 2009). In the low rainfall (< 650 mm), high evaporation (> 2000 mm), environment of central Queensland, Australia, water losses in CWs designed for bacterial sulphate reduction in coal mining wastewater caused the opposite effect, that of doubling SO_4 concentration after 70 days of treatment (Tyrrel et al., 1997). For these reasons, CWs are often declared to be inappropriate for arid climates.

Despite the lack of published research on ET rates from CWs i, it is clear that wetland designs for arid climates need to be different from those in conventional temperate environments (Headley et al., 2012).

2. CASE STUDIES

2.1. Influence of Evapotranspiration on Treatment Efficiency in Horizontal Subsurface Flow Systems

2.1.1. Material and Methods

System Design and Control

The experiment was performed on a HSSW located at Capinha in the Beira Interior in the center of Portugal near the border with Spain. A moderate Mediterranean climate with continental influence characterizes the region (annual average parameters: temperature = 14.5 °C; precipitation = 780 mm; ET = 700 mm). ET is normally higher than precipitation from June to September (summer time). The wastewater treatment plant (WWTP) of Capinha was designed for 800 e.p. and receives domestic, stormwater and livestock streams. It includes a bar racks channel, screening channel, flow rate measurement through a Parshall device, Imhoff tank and two parallel HSSW, each 50 m long and 15.5 m wide (total surface area, 773 m^2), filled with gravel and colonized with common reeds (*Phragmites australis* Trin ex Steudel) (Figure 1). The total depth of each bed is 1 m and the submerged operating depth is 0.65 m. . The beds were designed for flow rates from 45 to 90 $m^3\ d^{-1}$, HLR from 7 to 15 cm d^{-1}, HRT from 4.5 to 9 d, SSA of 2.5 m^2 p.e.$^{-1}$and COD concentrations from 300 to 500 mg L^{-1}. Capinha HSSW was monitored over a period of 19-months (January 2007 to July 2008), including flow-measurement and the collection of bi-monthly inflow and outflow samples (39 samples collected at sampling points (1) and (2) in Figure 1) to determine the following parameters: pH, air and soil temperatures, DO, COD, TN, NH_4-N, nitrite nitrogen (NO_2-N) and nitrate nitrogen (NO_3-N). All analyses were carried out according to standard methods (APHA-AWWA-WEF, 1999).

Treatment Efficiency

Water balance was estimated according to Equation (2), considering Q_{inf} as the influent flow rate ($m^3\ d^{-1}$), P as the precipitation over the CW area ($m^3\ d^{-1}$) and Q_{eff} as the effluent flow rate ($m^3\ d^{-1}$). Removal efficiency of COD, NH_4-N and TN in the HSSW was calculated, based on concentrations (Equation (4)) and on loads (Equation (5)). The two measures of RE were compared and correlations between ET_r and flow rate, and between ET_r and COD, NH_4-N and TN removal efficiencies were computed.

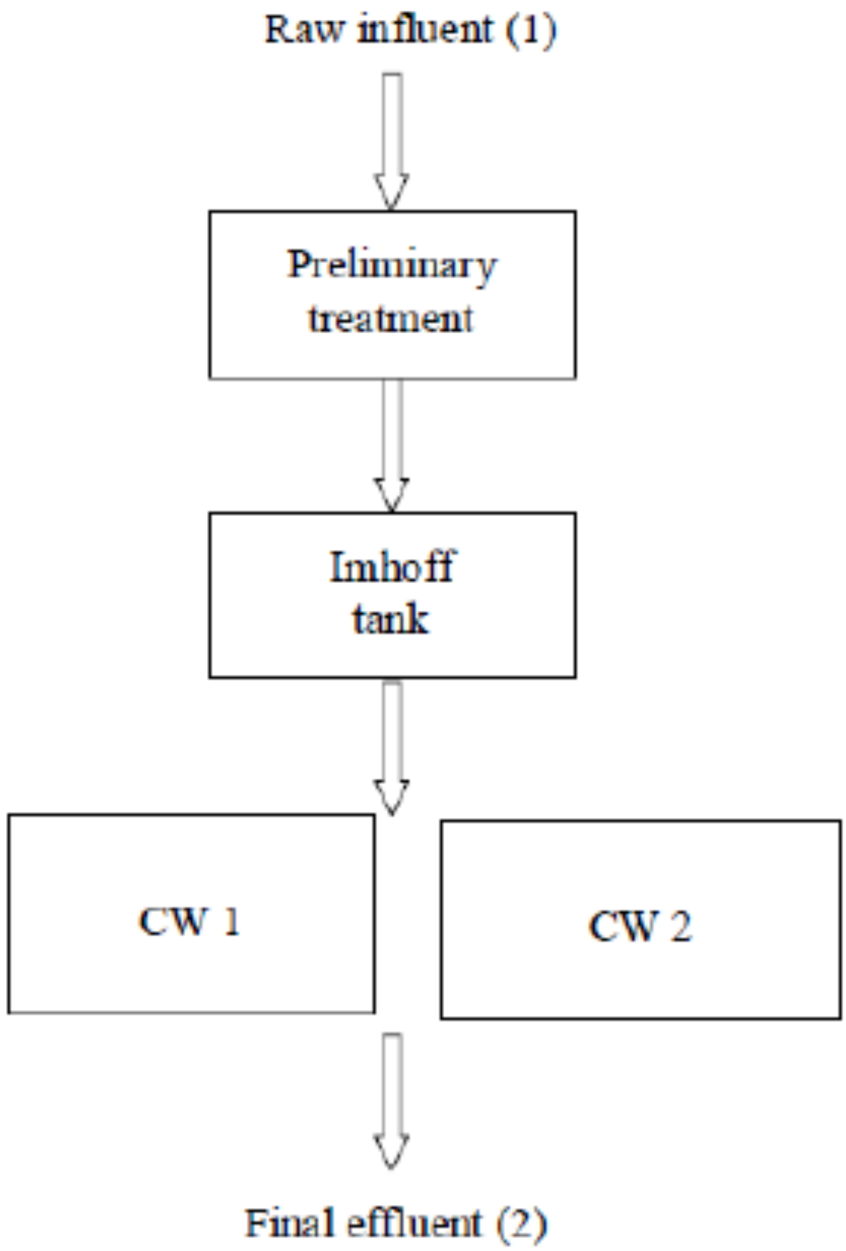

Figure 1. Schematic representation of the WWTP of Capinha (Portugal).

2.1.2. Results and Discussion

The results for the 19-months monitoring period are presented in Table 4. Nitrite and nitrate concentrations are very low and will not be analyzed. Table 5 shows RE for COD and TN based on concentrations (Equation (4)) and mass loads (Equation (5)) for the range of HLR observed during the monitoring period (73 to 145 mm d^{-1}), which was within the design limits (70 to 150 mm d^{-1}).

Table 4. Results for the monitoring of Capinha HSSW (January 2007 to July 2008)

Parameters	Influent	Effluent
pH	6.7 – 7.3	6.5 – 7.0
COD (mg L^{-1})	379.6 SD±75.4	102.9 SD±20.3
NH_4-N (mg L^{-1})	22.2 SD±5.5	8.5 SD±2.6
NO_2-N (mg L^{-1})	0.5 SD±0.3	0.0 SD±0.0
NO_3-N (mg L^{-1})	2.2 SD±1.4	0.2 SD±0.4
TN (mg L^{-1})	29.2 SD±4.5	9.3 SD±2.2
DO (mg L^{-1})	0.2 SD±0.1	0.5 SD±0.2
Air temperature (° C)	18.4 SD±9.2	
Soil temperature (° C)	17.5 SD±8.7	
ET_r (mm d^{-1})	14.0 SD±6.1	
P (mm d^{-1})	1.7 SD±4.5	

Note: average values and standard deviation (SD). Number of samples: 39.

Table 5. Treatment performance based on concentrations and mass loads for the Capinha HSSW (January 2007 to July 2008)

Parameters	Concentration RE (%)	Mass Load RE (%)
COD	72.5 SD±4.7	75.9 SD±5.0
NH_4-N	60.1 SD±13.0	64.7 SD±12.5
TN	67.4 SD±9.6	71.3 SD±9.4

Note: average values and standard deviation (SD). Number of samples: 39.

ET_r increased as air temperature increased and the highest value of daily ET_r, 37 mm d^{-1}, was obtained on 30.09.2007 with HLR of 87.8 mm d^{-1} (this value is one of the lowest HLR recorded during the 19 months). The results also show that ET_r influenced the RE of organic pollutants and nitrogen (Table 6), since the mass removal increased as ET_r increased, especially for HLR between 73 and 100 mm d^{-1} (average COD removal was 74.4% and 78.5% for concentrations and mass loads, respectively; and average TN removal was 70.9% and 75.5% for concentrations and mass loads, respectively). For HLR between 100 and 146 mm d^{-1} the RE of both parameters decreased (average COD removal was 70.8% and 73.5% for concentrations and mass loads, respectively; and average TN removal was 64.3% and 67.5% for concentrations and mass loads, respectively). These results show also that HLR over 100 mm d^{-1} leads to a decrease in CW treatment performance.

Over the 19-months monitoring period ET made a significant contribution to the water balance at the Capinha HSSW (Figure 2). On all sampling days ET_r was higher than precipitation, which is typical for a moderate temperate climate. ET_r varied between 4.3 and 17.6 mm d^{-1} during winter time (December to February; average 11.5 mm d^{-1}) and between 13 and 37.1 mm d^{-1} in the summer period (July to September; average 20.4 mm d^{-1}). The effluent values were lower than the influent on all sampling days. The difference between inflow and outflow was dependent on ET_r, and was highest during the summer months.

Table 6. Removal of COD and TN according to HLR and ET_r in the Capinha HSSW (January 2007 to July 2008)

Date	HLR (mm d^{-1})	ET_r (mm d^{-1})	COD removal (%)		TN removal (%)	
			Concentration	Mass Load	Concentration	Mass Load
12/01/07	114.7	16.6	73.0	74.9	46.0	49.9
29/01/07	131.0	17.6	70.8	74.0	71.8	74.8
13/02/07	106.3	6.3	71.9	73.3	69.2	70.7
28/02/07	100.2	7.9	72.2	74.4	54.3	57.9
14/03/07	145.5	16.7	59.6	64.2	74.1	77.1
30/03/07	101.8	10.9	76.1	78.7	77.4	79.8
13/04/07	86.3	19.6	74.3	78.2	79.1	82.3
30/04/07	97.5	11.1	69.2	72.7	62.7	67.0
12/05/07	119.4	18.0	72.7	76.8	56.7	63.2
29/05/07	84.9	9.8	69.4	72.6	65.7	69.3

Date	HLR (mm d^{-1})	ET_r (mm d^{-1})	COD removal (%)		TN removal (%)	
			Concentration	Mass Load	Concentration	Mass Load
14/06/07	79.2	15.2	77.0	79.9	55.9	61.6
30/06/07	73.4	12.9	66.1	72.1	74.8	79.3
15/07/07	82.1	21.8	78.9	84.5	75.1	81.7
30/07/07	89.1	23.3	80.9	85.9	81.1	86.0
13/08/07	75.6	14.0	76.0	80.4	69.5	75.1
30/08/07	73.5	17.4	75.6	81.3	71.1	77.9
14/09/07	81.8	14.2	75.0	79.3	68.1	73.6
30/09/07	87.8	37.1	75.3	78.7	58.0	63.8
12/10/07	96.4	14.4	76.2	79.8	72.7	76.7
30/10/07	114.4	9.2	69.4	71.9	76.9	78.8
14/11/07	126.3	11.5	61.8	65.3	62.0	65.5
29/11/07	106.3	5.8	70.7	72.3	44.4	47.5
12/12/07	132.4	16.2	67.8	71.7	57.8	63.0
29/12/07	120.7	7.4	69.9	71.7	75.1	76.7
14/01/08	113.4	15.6	68.0	70.3	68.1	70.4
30/01/08	125.2	11.7	72.8	75.4	60.9	64.6
12/02/08	130.9	14.1	78.1	80.5	66.4	70.0
28/02/08	114.2	6.9	76.4	77.4	67.6	69.0
12/03/08	96.4	4.3	67.4	68.8	57.6	59.4
30/03/08	100.5	13.7	70.2	74.1	67.3	71.6
14/04/08	103.8	12.8	66.9	71.0	57.0	62.3
29/04/08	93.6	7.4	75.1	77.1	79.6	81.2
14/05/08	124.7	13.8	74.9	77.0	60.5	63.8
30/05/08	105.1	7.8	72.8	74.8	72.2	74.2
14/06/08	94.8	16.6	71.4	76.4	83.7	86.5
29/06/08	87.3	15.3	74.4	78.9	81.7	84.9
13/07/08	79.2	12.9	77.4	81.1	72.3	76.9
30/07/08	84.4	22.7	80.0	85.4	67.6	76.3

A strong positive correlation between ET_r and the percentage decrease of water flow through the HSSW system was found (r=0.91) as shown in Figure 3. Therefore, it seem that an increase of ET_r had a positive effect on water flow decrease, regardless of varying inflow rates. The decrease of wastewater volume due to ET_r had an influence on treatment efficiency, as shown by comparing average RE of COD based on initial and final concentrations (Equaion (4)), with that based on loads (Equation (5)) (Table 4 and Figure 4).

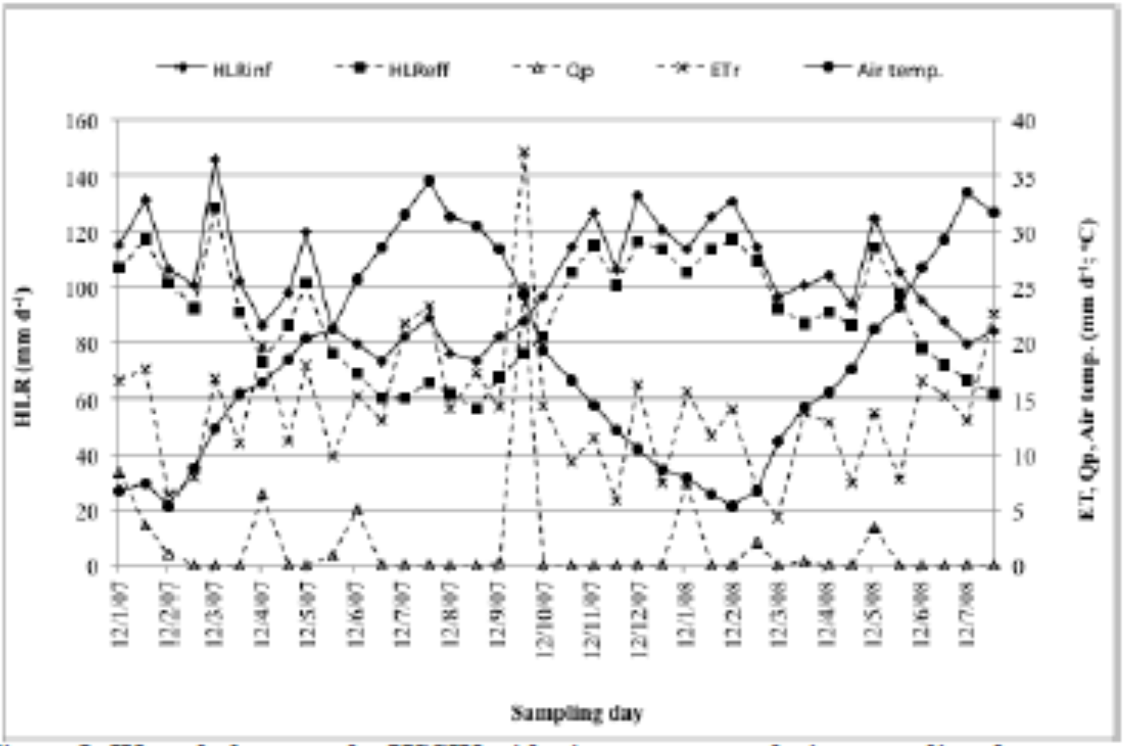

Figure 2. Water balance at the HSSW with air temperature during sampling days.

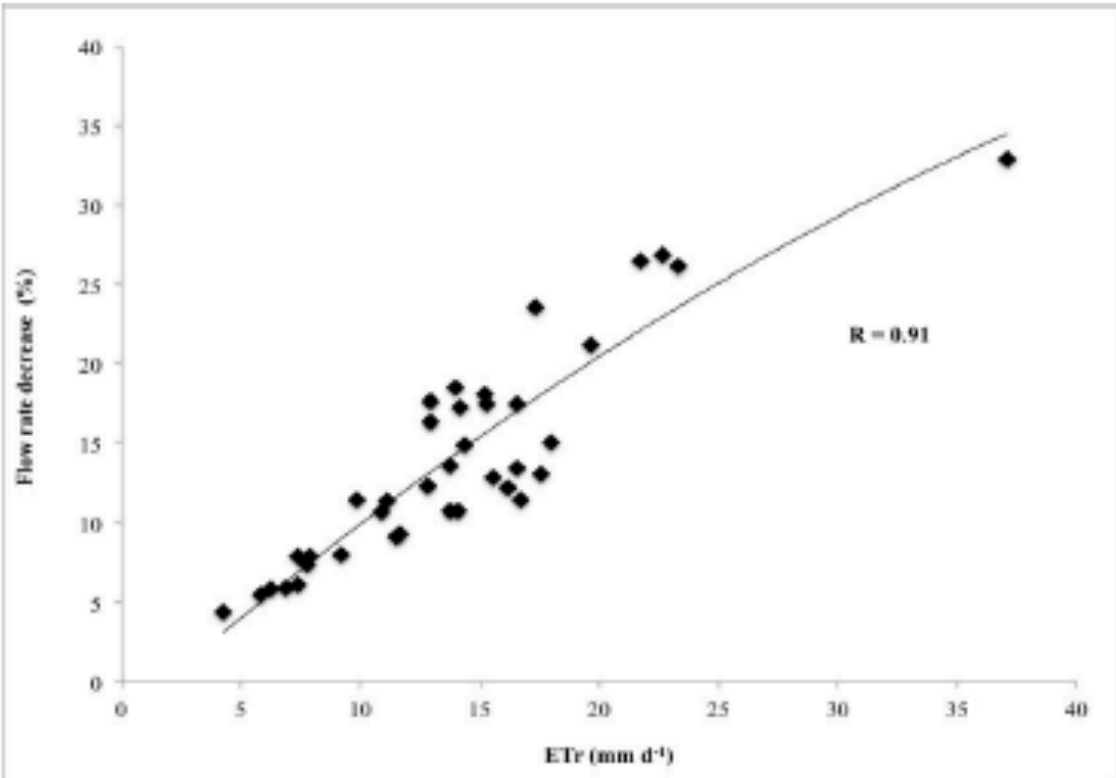

Figure 3. Correlation between ET_r and flow rate decrease for the HSSW.

Similar effects were found in the case of ammonia nitrogen removal (Table 4 and Figure 5), and total nitrogen removal in the HSSW (Table 4 and Figure 6).

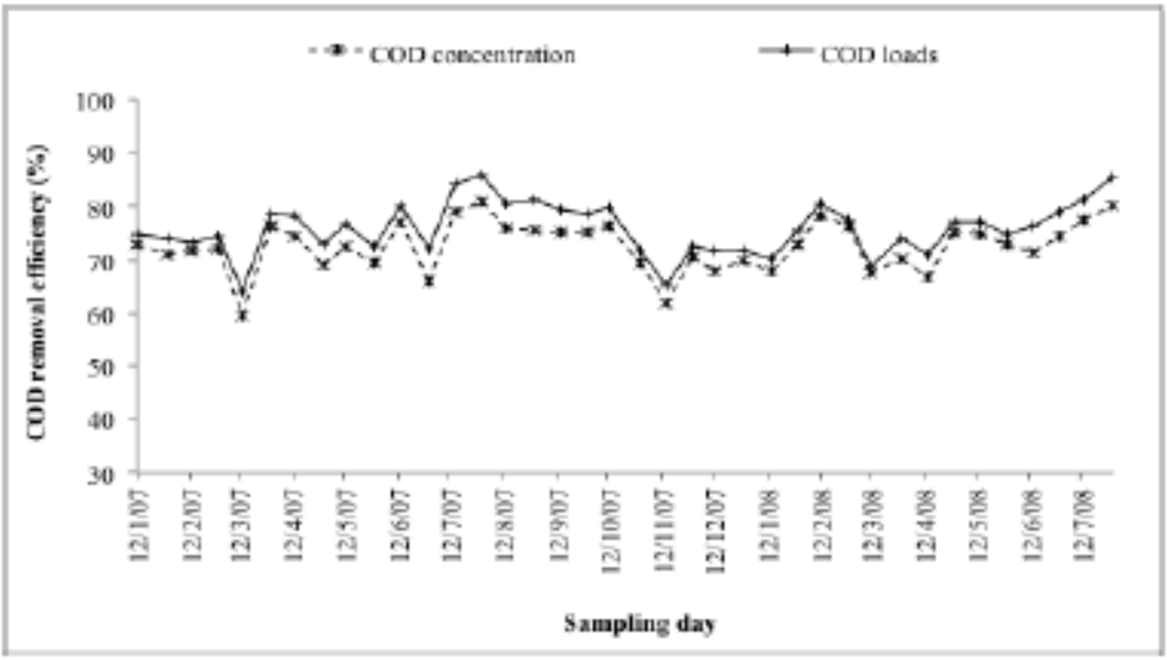

Figure 4. Comparison between COD removal efficiency calculated based on concentrations and on loads in the HSSW of Capinha over time.

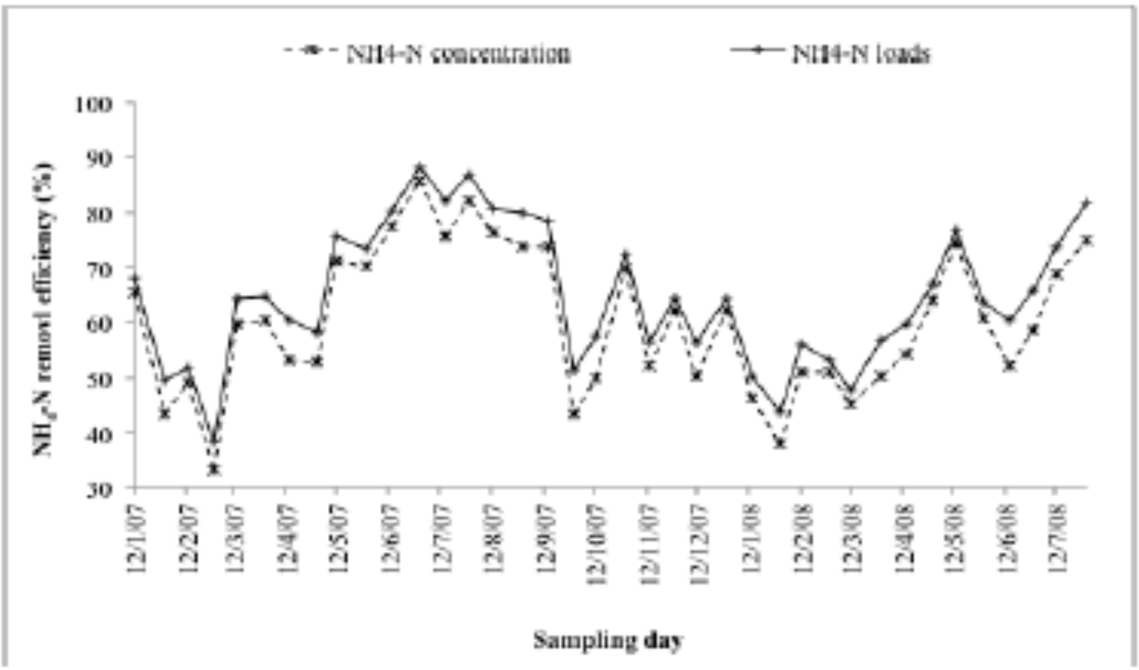

Figure 5. Comparison between NH_4-N removal efficiency calculated based on concentrations and on loads in the HSSW of Capinha over time.

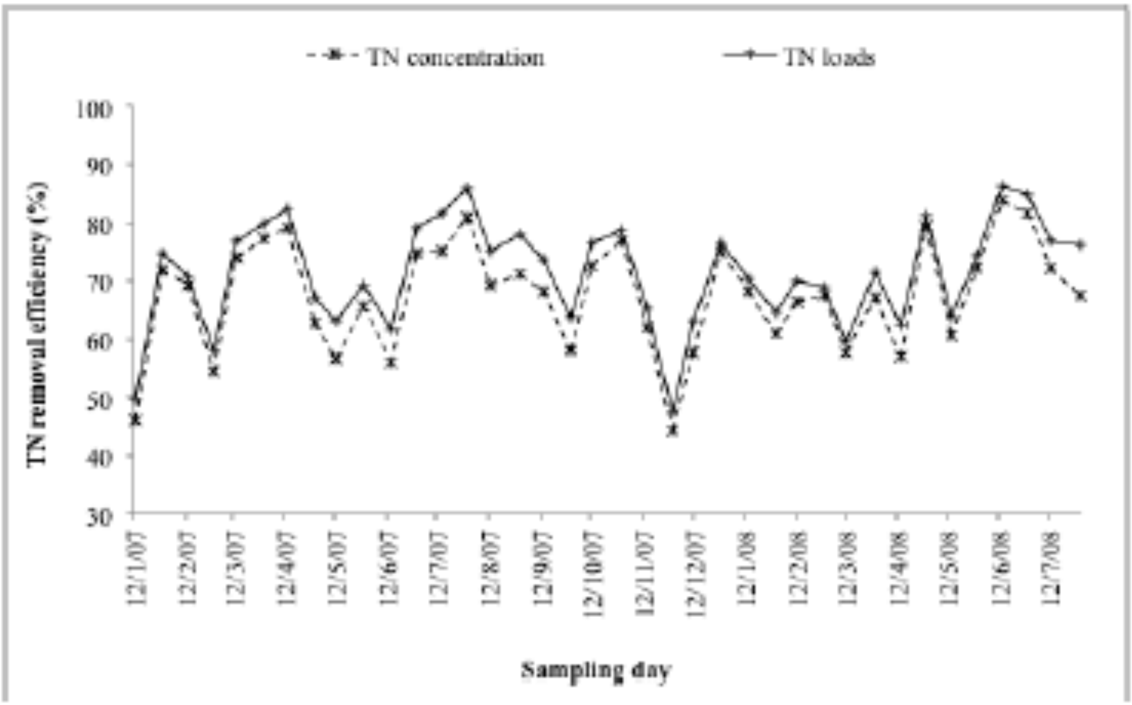

Figure 6. Comparison between TN removal efficiency calculated based on concentrations and on loads in the HSSW of Capinha over time.

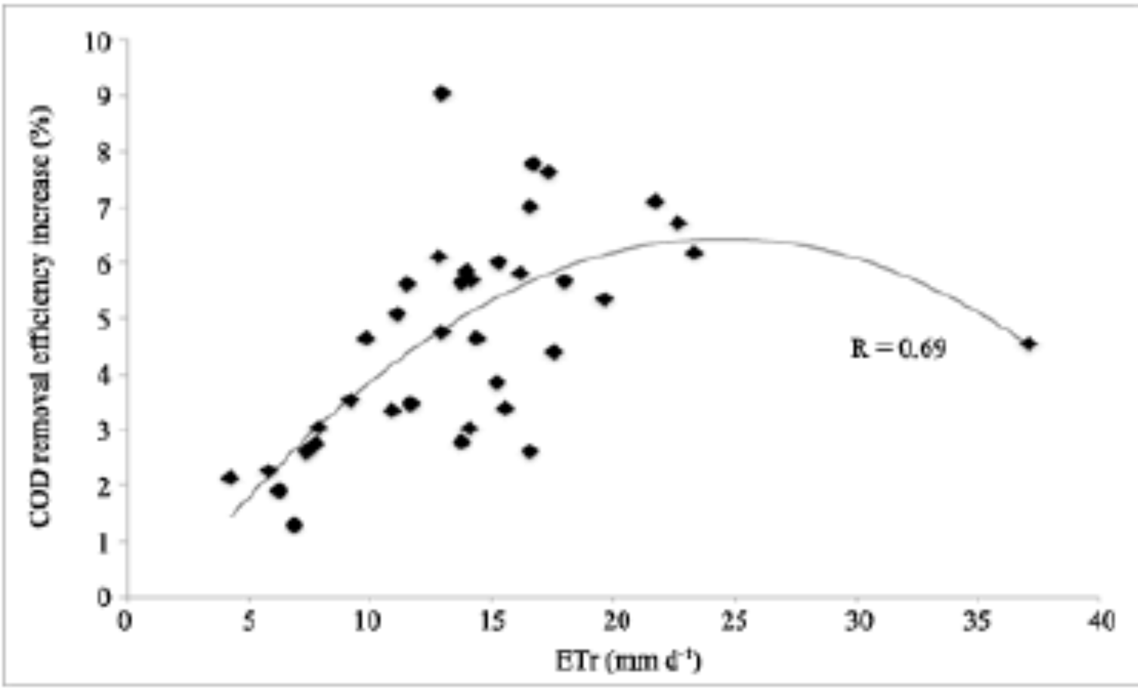

Figure 7. Correlation between ET_r and the increase in COD removal efficiency calculated using mass loads relative to concentrations for the HSSW of Capinha.

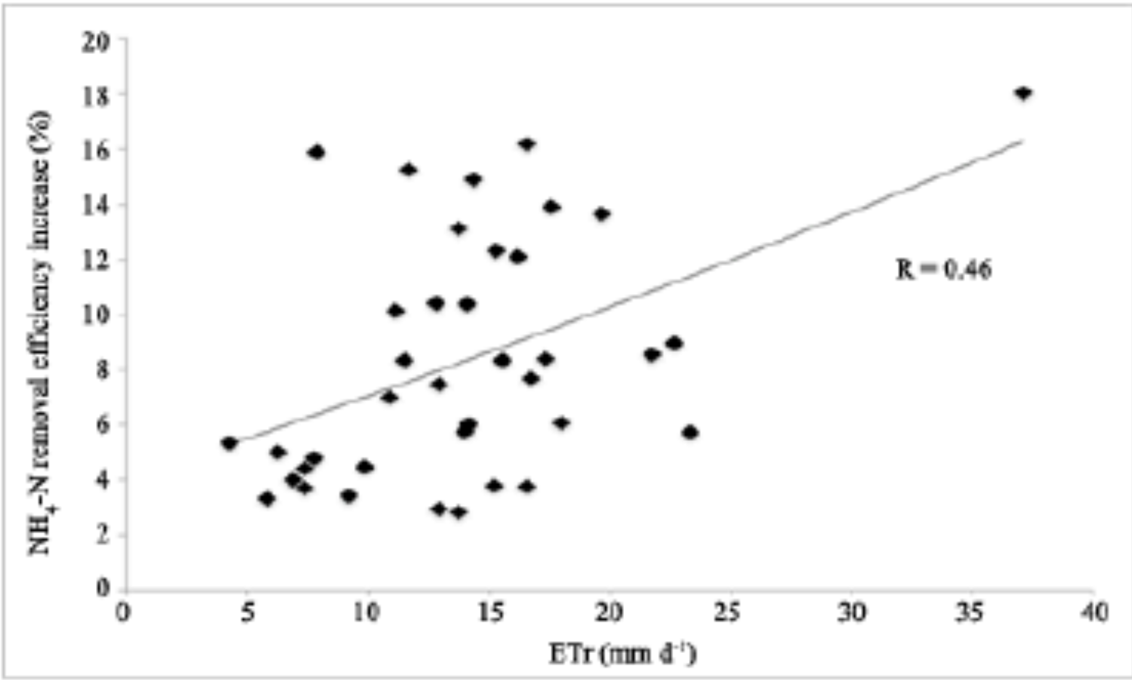

Figure 8. Correlation between ET_r and the increase in NH_4–N removal efficiency calculated using mass loads relative to concentrations for the HSSW of Capinha.

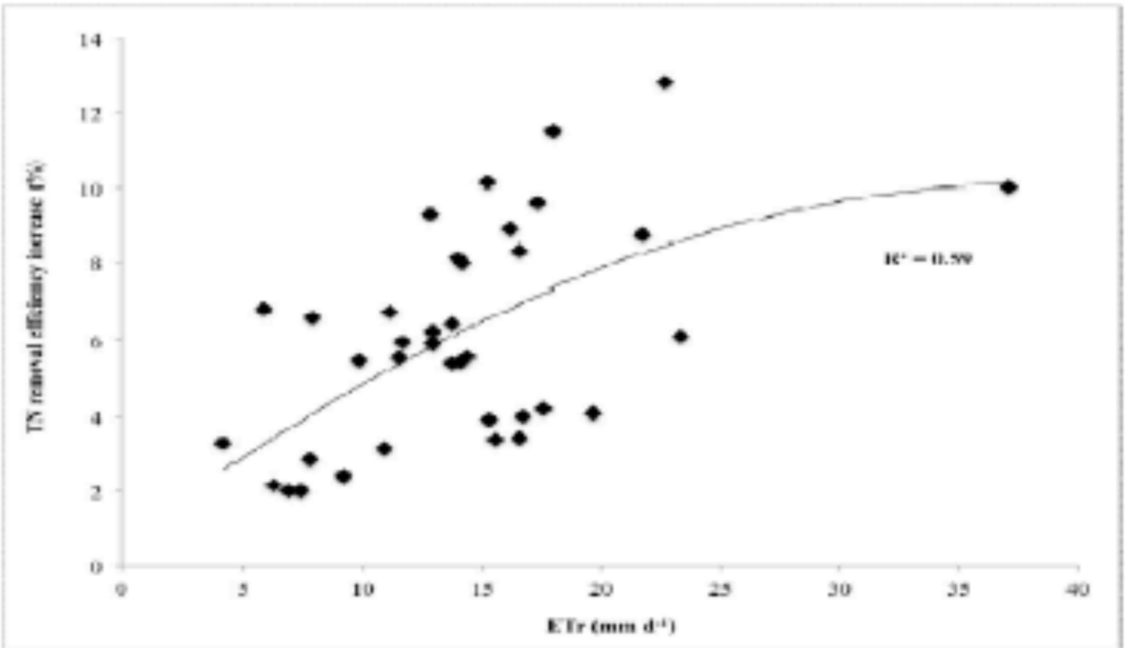

Figure 9. Correlation between ET_r and the increase in TN removal efficiency calculated using mass loads relative to concentrations for the HSSW of Capinha.

During all sampling days RE based on concentrations were lower than RE based on mass loads for the three parameters. The differences were higher during the hottest months (July to September), when ET calculated using mass loads increased more: 5.7%, 7.5% and 7.9% for COD, NH_4-N and TN, respectively. Average RE values based on loads were higher than values based on concentrations by 4.4% (COD), 7.2% (NH_4-N) and 5.6% (TN). Good correlations were obtained between ET_r and the increases in COD, NH_4–N and TN removal efficiencies based on mass loads relative to concentrations (r=0.69, r=0.46 and r=0.59, respectively; Figures 7-9).

2.2. Influence of Evapotranspiration on Treatment Efficiency in Vertical Subsurface Flow Systems

2.2.1. Material and Methods

System Design and Control

A laboratory scale experiment was conducted in (Agopsowicz et al., 2001), using a 1 m high lysimeter (0.6 m in diameter and 0.28 m^3 in volume), as a model of a constructed wetland with vertical subsurface flow (VSSW).

The construction details of two types of lysimeter have been shown on figure 10.

In order to compensate the temperature inside the lysimeter and in the soil, the lysimeter was placed in the ground.

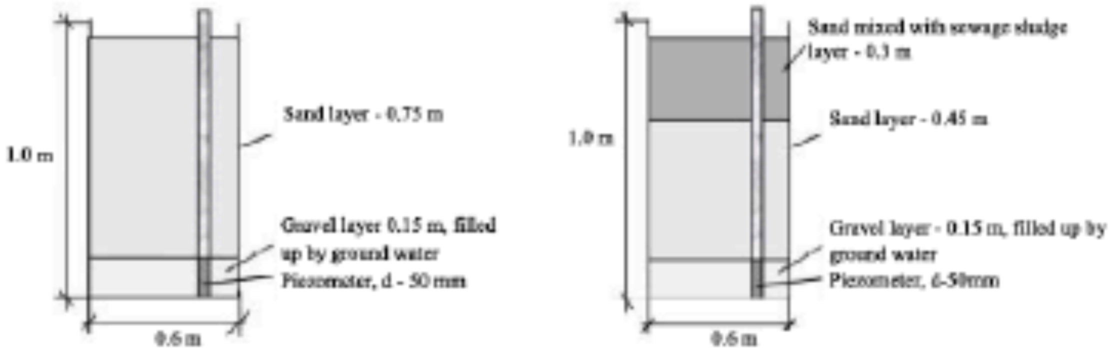

Figure 10. Lysimeters construction.

A municipal landfill in Wola Pawlowska near Ciechanów in Poland, was the source of treated wastewater (landfill leachate), in the experiment. Analyses of 48 landfill leachate samples indicated that average COD concentration was about 1425 mg L^{-1}; BOD_5/COD ratio was 0.17; EC value was approximately 12.3 mS cm^{-1}; ammonium value was 592 mg NH_4 L^{-1}, which comprised about 76% of Kjeldahl nitrogen; orthophosphate concentration was only 3.6 mg $P\text{-}PO_4$ L^{-1}.

The experiment was conducted during two vegetation seasons, as an example of VSSW with reed *Phragmites australis* (Cav.) Trin ex Steudel.

ET and RE of organic compounds in VSSW was examined as a function of:

- Soil type: sand (S) and sand with a dose of 450 mg ha^{-1} as dry matter (dm.) of sewage sludge (C);
- Leachate HLR: 1 mm d^{-1}, 3 mm d^{-1} and 5 mm d^{-1}.

Each variant of the experiment was repeated three times.

The applied sewage sludge had following properties: pH 6.82; moisture 57.8%, organic matter 76.5% (of d.m.) and Kjeldahl nitrogen 1.21% (of d.m.).

A 0.3 m deep layer of a sewage sludge-with-sand mixture was settled at the top of the one type of the lysimeter.

Into each lysimeter 7 reed-stocks were planted at a depth of 10 cm (Reed et al., 1995).

The lysimeters were located under a foil roof, which had been firmly insulated. There was no natural rainfall or natural groundwater inflow into the lysimeters. All the lysimeters were watered in the following ways:

- with landfill leachate according to the prescribed HLR (once per week);
- by simulation of rainfall (distilled water) (once per week);
- and incoming clean ground water (distilled water) according to meteorological data (once per month).

Once per month free drainage water that had gathered inside the lysimeter was pumped out through the gravel layer and the piezometer using a peristaltic pump, prior to weighing. Water masses (kg) were then converted into (mm) units.

Treatment Efficiency

Water balance was estimated according to Equation (2), considering Q_{inf} as the total amount of water added to the lysimeter (mm d^{-1}) and Q_{eff} as the amount of water pumped out from the lysimeter (mm d^{-1}). Q_{inf} was evaluated using Equation (6).

$$Q_{inf} = Q_l + Q_p + Q_s \tag{6}$$

where Q_l is the amount of added landfill leachate (mm d^{-1}), Qp is the amount of precipitation (mm d^{-1}), and Q_s is the amount of added ground water (mm d^{-1}).

Similarly as HSSW system in first case study, the RE of COD in the VSSW was calculated, based on concentrations (Equation (4)) and based on loads (Equation (5)). The two measures of removal efficiency were compared and correlations between ET and COD removal efficiencies were determined.

2.2.2. Results and Discussion

The experiment showed that ET depended on both HLR and soil type. The daily ET_r differed between successive vegetation seasons. In the first year after planting, during the young reed plants' development ET values were lower than in the second vegetation season, when plants were well grown (Table 7).

ET from systems containing sand with sewage sludge mixture was higher than that from the systems with sand alone, throughout the two years of the experiment. The addition of organic material (sewage sludge) improved the growth conditions, resulting in higher transpiration rates. Also, ET increased with the rise of HLR during two vegetation seasons in both kinds of soil. The highest value of daily ET_r, 4.61 mm d^{-1}, was obtained in the second year in VSSW covered by sand with sewage sludge mixture, with HLR 5 mm d^{-1} of landfill leachate.

Similarly to case study with HSSW in Portugal, this experiment showed that ET_r influenced the RE of organic compounds. It has been proved that treatment efficiency, calculated on the basis of both concentrations and loads, decreased with the increase of HLR, in all variants, but treatment efficiency differs in values if calculated based on concentrations or based on mass loads. In systems with sand, under the poor nutritional conditions, water losses (ET_r ranging from 0.98 to 2.72 mm d^{-1}) were too small to influence pollutant removal efficiency, and the higher values were observed for concentrations.

In systems with soil mixture containing sand–with-sewage sludge, especially during the second vegetation season, and at the highest HLR, ET_r caused a marked reduction in removal efficiency based on concentrations compared to that based on mass loads.

Average RE based on concentration was only 28%, whereas the value based on loads reached 69% (Table 7).

It has been found that when ET_r is low (below 2.5 mm d^{-1}), removal efficiency does not depend on ET_r, but when ET_r starts to rise above 2.5 mm d^{-1} water losses cause a rapid decrease of removal efficiency. At higher values of ET_r, final concentrations may be even higher than initial (Figure 11).

Table 7. Performance of reed VSSW on COD removal for landfill leachate treatment depending on the soil type, and hydraulic loading rate in successive vegetation seasons

Vegetation season	Parameters	Soil type: Sand			Soil type: Sand + Sludge		
		HLR 1 mm d^{-1}	HLR 3 mm d^{-1}	HLR 5 mm d^{-1}	HLR 1 mm d^{-1}	HLR 3 mm d^{-1}	HLR 5 mm d^{-1}
I[st] vegetation season	Concentration RE for COD (%)	90.2 SD±2.0	72.1 SD±3.7	56.2 SD±7.7	92.5 SD±2.1	75.3 SD±7.1	53.6 SD±15.4
	Mass load RE for COD (%)	77.88 SD±2.76	61.6 SD±1.5	50.1 SD±4.2	88.3 SD±4.1	75.6 SD±11.0	68.6 SD±13.6
	ET (mm d^{-1})	0.98 SD±0.61	1.00 SD±0.46	1.48 SD±0.48	1.59 SD±0.66	2.22 SD±1.07	2.99 SD±0.89
II[nd] vegetation season	Concentration RE for COD (%)	83.3 SD±18.5	80.9 SD±6.8	53.8 SD±16.8	79.5 SD±25.6	46.9 SD±53.9	28.0 SD±56.1
	Mass load RE for COD (%)	82.3 SD±5.5	79.3 SD±8.2	59.7 SD±5.1	89.1 SD±2.3	86.8 SD±8.2	69.4 SD±9.9
	ET (mm d^{-1})	2.72 SD±1.47	2.56 SD±1.22	2.69 SD±1.17	3.51 SD±1.89	3.95 SD±2.42	4.61 SD±2.33

Note: average values and standard deviation (SD). Number of samples: 5 for each variant in the I[st] year, and 6 for each variant in the II[nd] year.

In contrast, there was no significant dependence of ET_r on COD removal efficiency calculated using mass loads (Figure 12).

Regarding the operating conditions, most of the studies with VSSW (Vymazal, 2005; Vymazal, 2007; van Deun and van Dyck, 2008; Morari and Giardini, 2009; Cheng et al., 2011; Bialowiec et al., 2011) and HSSW (Vymazal, 2005; Vymazal, 2007; El Hamouri et al., 2007; Vymazal and Kropfelova, 2008; Vymazal, 2009; Albuquerque et al., 2009a and 2009b; Amado et al., 2012; Bialowiec et al., 2012b) evaluate the treatment efficiency of beds based on concentrations as presented in Equation (4).

This approach relates to specific conditions within a CW bed, where the influence of plants is not only on pollutants, but also on water. Plants transpire water into the atmosphere, increasing the concentration of pollutants. Simultaneously, microbial degradation is in progress and is dependent on oxygen concentration, HLR, influent loads and temperature, among other factors.

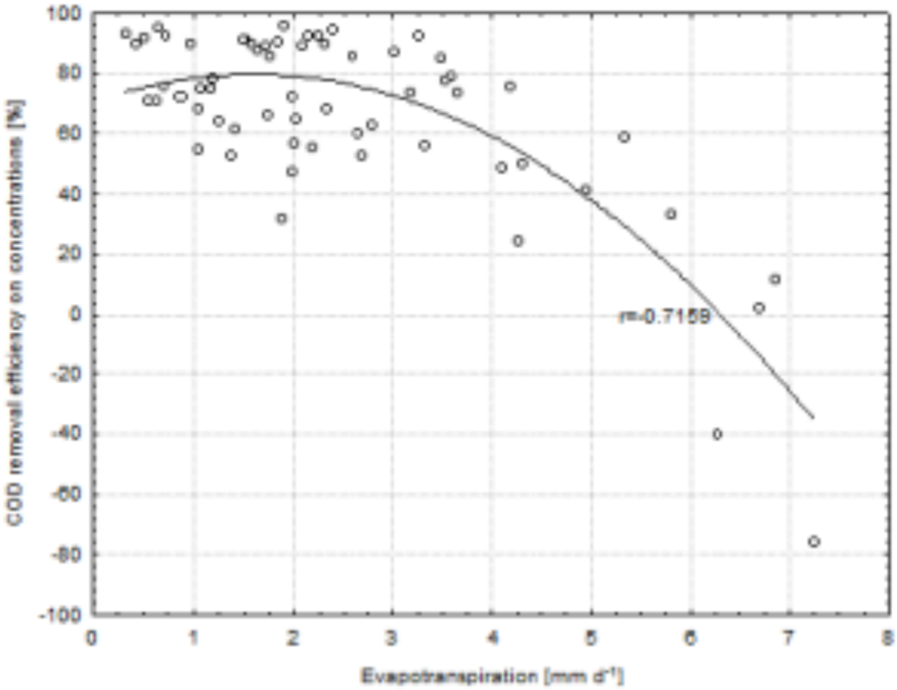

Figure 11. Correlation between ET_r and COD removal efficiency calculated using concentrations for the VSSW.

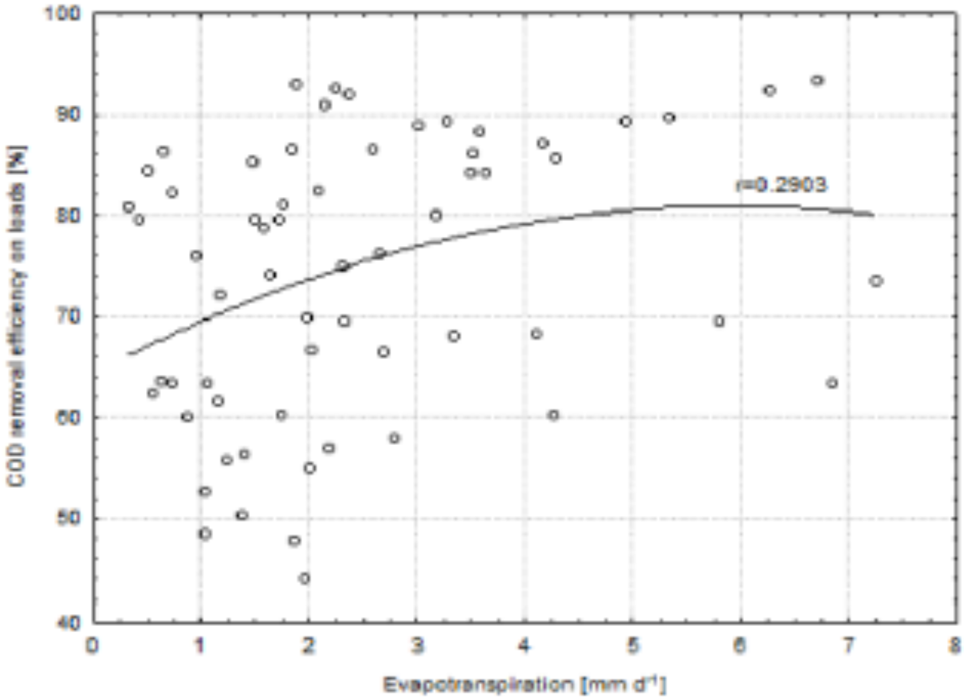

Figure 12. Correlation between ET_r and COD removal efficiency calculated using mass loads for the VSSW.

It could be concluded that two independent processes of reducing the both pollutants and water molecules are taking place. Depending on the relative rate of each process, the final pollutant removal efficiency value differs. When ET is low, the difference between RE calculated through Equation (4) or Equation (5) is small, but with ET rising the RE based on mass loads increases relative to that based on concentrations.

The case studies presented in this work show higher values of RE calculated using mass loads in VSSW, especially for high values of HLR when the both the ET_r and the ambient temperature was lower (Table 7), as well as in HSSW operated at higher both ET_r and temperatures (Table 4), as also found by Finlayson et al. (1987) (Table 3).

In our opinion, the RE calculated through mass loads of compounds (given by Equation (5)) should be used as a more accurate method for performance evaluation in CW than the method based only on influent and effluent concentrations.

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