Carbon and Nitrogen Dynamics and Greenhouse Gases Emissions in Constructed Wetlands Treating Wastewaters: A Review

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Abstract

Globally, constructed wetlands (CWs) are a developing technology for the removal of pollutants (e.g. carbon- C and nitrogen- N) from wastewater. The reported removal efficiency of C and N in CWs is very inconsistent and does not in itself reveal whether the removal processes are due to physical attenuation or whether the different species have been transformed to other reactive forms. The existing international literature could not provide an explanation for such low removal efficiencies and typically does not consider the fate of the removed nutrients. A comprehensive review is therefore necessary to assist with the understanding of removal mechanisms and process kinetics of C and N in CWs, which underpin their environmental benefits. Herein, we provide an overview of the current knowledge and discussion concerning the biogeochemical processes that control nutrient removal in CWs used for treating wastewater and the likely impacts of these processes on aquatic and atmospheric environments. This paper does not only identifies knowledge gaps that currently inhibit a holistic understanding of the fate of various C and N species but also itemises methodologies to elucidate the necessary data to eliminate such knowledge gaps. It highlights that nutrient removal efficiencies in CWs vary with CW type, vegetation type, climate, season, treatment region and other management practices. No single CW type shows an optimum performance in removing C and N pollutants. Horizontal flow CWs tend to show higher performance for nitrate (NO_3^-) removal as they provide good conditions for denitrification but cannot remove ammonium (NH_4^+) due to limited ability to nitrify NH_4^+. Vertical flow CWs show higher performance for NH_4^+ removal but their denitrification ability is low. Surface flow CWs decrease N_2O emissions but increase CH_4 emissions; subsurface flow CWs increase N_2O and CO_2 emissions but decrease CH_4 emissions. Mixed species perform better than monocultures in increasing C and N removal and decreasing GHG emissions but experimental evidence is still scarce. Lower hydraulic loading with higher hydraulic retention time enhances nutrient removal but more experimental evidence is required to determine an optimum design. A conceptual model is presented which highlights the current state of knowledge and suggests specific experimental work across CW, vegetation and wastewater type, hydraulic loading rate and retention time and hydrologic pulsing headings. We recommend that further research on process based C and N removal and on the balancing of end products into reactive and benign forms is critical to the assessment of the environmental performance of CWs.

Keywords: Carbon, nitrogen, constructed wetlands, pollution swapping, nitrous oxide, methane

1. Introduction

Increasing anthropogenic loading of reactive nitrogen (Nr; all forms of nitrogen except di-nitrogen gas, N_2) along the nitrogen (N) cascade in the environment raises many critical concerns for human health, drinking water quality (Gray, 2008), coastal and marine water degradation causing eutrophication, as well as algal blooms and hypoxia (Conley et al., 2009; Rabalais et al., 2010). There are natural sinks for Nr along the N cascade (Galloway et al, 2003; Tanner et al., 2005) but other sinks that are rehabilitated or artificial, may also be introduced and evaluated at key locations e.g. constructed wetlands (CWs) (Gold et al., 2013) or permeable reactive interceptors (Fenton et al., 2014). This review focuses on CWs treating wastewater which are now an emerging technology used globally as a mitigation option for nutrient retention at the delivery end of the transfer continuum (Dunne, et al., 2005; Søvik et al., 2006; Tanner and Kadlec, 2013; Tanner and Sukias, 2011). Although CWs have a proven potential for the removal of organic carbon (C) and N, studies have shown that removal efficiencies have been highly variable, due to inadequate observations of C and N transformations and their removal mechanisms (Seitzinger et al., 2002) and lack of improved adoption and placement (Gold et al., 2013). Constructed wetlands are complex bioreactors which facilitate a number of physical, chemical and biological processes but often continue to be evaluated as a 'black box' in terms of process understanding (Langergraber, 2008). Many investigations target single contaminant remediation, whilst disregarding the reality of mixed contaminants entering and leaving such systems. They do not consider the dynamic of pollution swapping (the increase in one pollutant as a result of a measure introduced to reduce a different pollutant) driven by transformational processes within and around the system. The CWs can have negative climatic impacts (Ström et al., 2007), act as an active source of greenhouse gas (GHG) (IPCC, 2014; Clair et al., 2002; Mander et al., 2008; Mitsch and Gosselink, 2000) and enhance pollution swapping (Reay, 2004). There are many pathways by which the removed N can contribute to water and air pollution: accumulation and adsorption in soils, leaching to groundwater (nitrate-NO_3^- and ammonium-NH_4^+), emission to atmosphere via nitrous oxide- N_2O and ammonia and/or conversion to di-nitrogen (N_2) gas. The kinetics of these pathways/processes varies with CW types and management practices and therefore needs to be studied quantitatively for their sustainable use. The impacts of these factors on C and N cycling and associated GHG emissions, especially in response to a changing climate, are poorly known.

Several processes can produce N_2O in CWs: nitrification, denitrification and nitrifier denitrification. It is known that CWs significantly contribute to atmospheric N_2O emissions either directly to the atmosphere from the surface of the wetland as its by-product (IPCC, 2014; Søvik et al., 2006; Ström et al., 2007; Elberling et al., 20011; Van der Zaag et al., 2010) or indirectly via dissolved N_2O in the effluent or groundwater upon discharge to surface waters (Riya et al., 2010). The global importance of this N_2O emission has been increasingly recognised (IPCC, 2014). Indirect N_2O emissions from CWs effluents that discharge directly to aquatic environments have a default emission factor (EF) of 0.005 but with a wide range (0.0005-0.25). The wide range of EF is highlighting the large uncertainty, which warrants further refinement.

Constructed wetlands receive organic C from the wastewater, being used for treatment, and also from fixation by the photosynthetic hydrophytes which are rapidly recycled by respiration but much of this is incorporated into soil as organic C. Soil organic C undergoes the biogeochemical processes that regulate C accretion in soil and microbial respiration, producing carbon dioxide (CO_2). Anaerobic mineralization of organic C by methanogenic archaea can produce methane (CH_4) and exchange with atmosphere (Laanbroek, 2010; Ström et al., 2007; Søvik et al., 2006; Pangala et al., 2010). Constructed wetlands can contribute to the dissolved organic carbon (DOC) load transfer to ground and surface waters which can produce and exchange substantial amounts of CO_2 and CH_4 with atmosphere (Clair et al., 2002; Elberling et al., 2011). The CWs are thus can diminish the environmental benefits of wastewater treatment in the exchange of GHG which need to evaluate. The dynamics of dissolved N_2O, CO_2 and CH_4 in CWs is a key ‘‘loop hole’’ in global greenhouse gas budgets.

Surface emissions of GHG from CWs have been commonly measured by closed chamber method (Johansson et al., 2003, 2004; Mander et al., 2005, 2008) but rarely measured by ebullition and diffusion methods (Søvik et al., 2006). The measured rates have shown high spatial, temporal and diurnal variations due to the change in biogeochemistry of C and N and plant-microbes-soil interaction over time and space. The surface emissions cannot explain the kinetics of production and consumption rates of GHG which we need to know for adoption of better management to mitigate emissions. In addition, subsurface export of dissolved nutrients and GHG, being an important pathway of nutrient loss (Riya et al., 2010), is ignored. Mass balance analysis of the different components of the N cycle and kinetics of their transformation processes occurring within the treatment cells using the isotope-tracing ^{15}N technique can provide mechanistic information for N transformation products (Lee et al., 2009; O’Luanaigh et al., 2010) and

therefore can be used to start to answer such questions. Similarly, ^{14}C application and measurement of C species (e.g. CO_2, CH_4, and DOC) with the soil core incubation could elucidate the C mineralization and CO_2 and CH_4 production and consumption. These methods in combination will be able to provide a comparative analysis of the rates of C and N transformation processes and role of these processes in delivering NO_3^- and or NH_4^+ and DOC to ground/surface waters and N_2O, CO_2 and CH_4 to atmosphere. Past reviews on CWs, though very limited, summarise the performance of different types of CWs on C and N removal (Vymazal, 2007) and surface emissions of GHG (Mander et al., 2014) but have not discussed the mechanisms of nutrient removal and the fate of the nutrients delivered and removed to and from CWs. The objectives of this review are i) to understand the biogeochemical dynamics of C and N in CWs ii) to better understand the fate of various C and N species in a holistic manner, in addition to the conventional influent – effluent balance for nutrient removal iii) to identify the research gaps that need to be undertaken for optimising nutrient removal efficiencies and mitigating GHG emissions and iv) to focus on the advanced methods that can give insights into the understanding of production and reduction of GHG.

2. Removal Efficiency, Hydraulic Loading and Retention Time

In CWs, the efficiency of C and N removal is generally limited and highly variable over CW types, plant types, seasons, climatic regions and management practices. On average, it appears that 50 and 56% of the influent TN and TOC, can be removed respectively, but the removal rates are very inconsistent. Mean (±SE) TN removal efficiencies, obtained from the literature cited, ranged from 31.3 ± 6.3% in surface flow (SF) CWs to 40.4 ± 4.4% in subsurface flow CWs whereas TOC removal ranged from 18.8 ± 9.4% in SF CWs to 56.2 ± 9.5% in vertical subsurface flow CWs (Table 1 and Table 2). In European systems, for example, typical removal percentages of ammoniacal-N in long-term operation are c. 35% but can be enhanced if some pre-treatment procedures are followed (Verhoeven, and Meuleman, 1999; Luederitz et al., 2001). Generally, TN removal is higher in surface flow (SF) CWs than subsurface flow (SSF) CWs (Table 1) but there is no straight forward relationship observed in previous studies. For example, Van der Zaag et al. (2010) showed higher N removal in SF CWs than SSF but Søvik et al. (2006) and Gui et al. (2007) showed the opposite. In SSF CWs, limited removal can be caused by the typically reduced environment that enhances NH_4^+ accumulation and limits NH_4^+ oxidation. In SF CWs, denitrification rates can be limited due to lack of NO_3^-. In vertical subsurface flow (VSSF) CWs, aeration can increase NH_4^+ oxidation to NO_3^- which can be denitrified or converted to NH_4^+ by DNRA. Plant species are important structural components of CWs, which affect C and N

removal. Species selection based on the published data is still complex because some species are efficient in removing one pollutant but not the other (Bachand and Horne, 2000; Bojcevska and Tonderski, 2007; da Motta Marques et al., 2000). In some studies there are no inter species differences at all (Calheiros et al., 2007). Impact of plant species on nutrient removal has not been reported clearly in the published literature, where species differences for one or more nutrient was observed, because of the problems with plant growth due to unpredicted environmental conditions, wastewater toxicity, and in many cases the reasons were not specified (Fraser et ., 2004; Solano et al., 2004; Haule et al., 2002). Plants regulate CWs hydrology (evaporation and transpiration), temperature (insulating CWs from seasonal temperature change, trapping falling and drifting snow and heat loss of wind). Mixed species perform better than monocultures to remove C and N pollutants because they increase microbial biomass and diversity. Some species can create large surface area for microbial attachment and enhance microbial diversity, but experimental evidences are still scarce. Future research should investigate the mechanisms of nutrient removal rather than the only empirical comparison in removal efficiency for the plant species (Brisson and Chazarenc, 2009).

Soil physicochemical properties are important factors controlling the purification capacity in CWs. Soils with high permeability enhance downward nutrient movement to groundwater. High cation exchange materials in soil enhance NH_4^+ fixation by soil matrix. Microbial activities and growth depend on substrate C quality and C: N ratios which consequently affect nutrient removal efficiency. Better growth of heterotrophic microorganisms is a function of the wastewater C: N stoichiometry (Makino et al., 2003). A specific level of C:N ratio can favour one group of microbes but disfavour some other groups. For example, higher C:N ratio can enhance denitrification by providing electron donor for denitrifiers but the opposite can increase nitrification. High C:N ratios can also encourage DNRA over denitrification. Yan et al. (2012) showed the highest TN removal at C:N ratio 2.5: 1 coupled with the lowest TOC removal which indicates that removal of one nutrient might lead to a problem to a different one. Future research should therefore orient to maintain an optimum C:N ratio for achieving better water quality standard. The uncertainty in removal efficiencies suggest that the rates of C and N transformations and the fate of the removed nutrients within the CWs should be investigated. However, to our knowledge, no study is available that provides a holistic evaluation of C and N attenuation and transformations.

The removal efficiency of pollutants in CWs depends on hydraulic loading rates (HLR) and hydraulic retention time (HRT) (Toet et al., 2005). The HLR and HRT are considered to be significant design parameters determining the nutrient removal efficiencies (Weerakoon et al., 2013). Longer residence of wastewater in CWs increases the removal of C and N (Wang et al., 2014) by increasing sedimentation and duration of contact between nutrients and the CW systems. The effects of HLR and HRT can vary with the nature of the use of CWs e.g. whether they are used for treating single or mixed pollutants. To reduce Nr delivery to the receiving waters or to the atmosphere, CWs need to be optimally designed with respect to HLR and HRT.

Fluctuating hydraulic loading influences all biotic and abiotic processes in CWs. For example, if the groundwater table is lowered through changes in hydraulic loading, soil aeration can increase or decrease. Ammonification and nitrification rates increase with increased soil aeration and this enhances C utilization by bacteria and therefore can stimulate the removal of TC and TN. Investigation into the effects of fluctuating hydraulic loadings (hydraulic pulsing) on C and N removal efficiencies and their transformation products will provide information about the fate of the added nutrients in terms of their environmental benefits and or pollution swapping. For example, if the dominant product is N_2, the system will be more benign to the environment but if it is NH_4^+, it can be fixed in the soils or transport to ground and surface waters connected to CWs if the cation exchange sites become saturated. However, contrasting results on the impacts of hydraulic loadings on nutrients removal efficiency are available in literature (Toet et al., 2005). Luo et al. (2005) reported that low HLR results in incomplete denitrification, whereas Zhang et al. (2006) argued that low HLR increases NH_4^+ and chemical oxygen demand (COD) oxidation. Song et al. (2011) reported an average HRT of 2.4 days during wetland operation periods for the removal of NO_3^- and NH_4^+ without documented design optima or understanding of their fate. The way in which a CW performance is assessed can lead to different conclusions regarding removal of Nr. For future studies, evaluation of systems in a holistic manner, which include pollution swapping at different HLR and HRT is important, particularly within the context of the changing hydrologic cycle in a changing climate. Also local legislative targets should be considered and weighting factors (are GHG more important than water quality targets?) developed for different species within the system to come up with overall balances for the CW. In addition to the estimation of nutrient removal rates, investigation of the effect of HLR and HRT on the different forms of nutrients in the final effluents and their fate in the natural environment can attempt to elucidate the pollution swapping potential of CWs.

3. Accumulation of C and N in CWs Soils

The soil in CWs is a major sink for C and N (Mustafa and Scholz, 2011). Data on the C and N accumulation (dissolved/particulate organic carbon - DOC/POC (labile or stable), dissolved organic N- DON, TN, NH_4^+ or NO_3^--N) within the soil profile of various CWs are scarce as most studies focus on N balances between influent and effluent N loads. A wide range of N accumulation has been reported in the literature (e.g. 30-40%, Shamir et al., 2001; 39%, Harrington et al., 2007; 9%, Mander et al., 2008; 2.5%, Obarska-Pempkowiak and Gajewska, 2003). This wide range could be due to the variations in CW types and management strategies. The accumulated species of N are reactive unless they have been transformed to N_2 by biogeochemical processes. However, there is a dearth of information on the extent of Nr accumulation in soils and discharge to surface waters and air (Shamir et al., 2001). Accumulated organic N could be mineralised to NH_4^+ and NO_3^- depending on the physico-chemical properties of soil. The Nr could be assimilated by plants and microbes which are recycled in a soil-plant-soil continuum. Nitrogen spiralling occurs from NH_4^+ to organic N and back to NH_4^+ within the CW (O'Luanaigh et al., 2010). Typically, N accumulation has been found to decrease with soil depth (Shamir et al., 2001). In terms of the conventional input-output balance, these are considered as removed N, but in fact in such a biogeochemically active system they still remain**.** In addition to N, organic C accumulation occurs in CW soils (Nguyen, 2000). As such, soils of CWs represent organic C and Nr rich systems where the products of the continuously occurring biogeochemical processes can be transported to fresh waters and to the atmosphere. Estimation of the rates of nutrient accumulation in soils in various types of CWs under different management systems is important. The stability of the accumulated C and N under changing climatic scenarios also needs to be addressed to consider the long term sustainability of CWs.

4. C and N Dynamics and Greenhouse Gas Emissions

Increased nutrients input to the CWs increases the productivity of wetland ecosystems and thus increase the production of GHG. As CWs are designed to remove pollutants in an anaerobic/suboxic environment, they change the C and N biogeochemistry and contribute significantly to CH_4 and N_2O emissions (Johansson et al., 2002, 2003; Mander et al., 2005, 2008; Stadmark and Leonardson, 2005; Liikanen et al., 2006). Søvic et al. (2006) measured N_2O, CH_4 and CO_2 emissions in various CWs in different European countries and suggested that the potential atmospheric impacts of CWs should be examined as their development is increasing globally. Management of CWs must consider the negative climatic aspects of increased emissions

of GHG in addition to their primary functions (Ström et al., 2007). Therefore, estimation of the contribution of CWs to global warming is required. In this regard, measurement of spatial and temporal variations (seasonal and diurnal) of GHG emissions is required to accurately estimate CWs derived GHG emissions. A holistic assessment of ecologically engineered systems has been thoroughly outlined in Healy et al. (2011, 2014) and developed further in Fenton et al. (2014). Such assessments can be applied in evaluating nutrient dynamics in CWs. Moreover, plant mediated GHG emissions could be an important component of total emissions but again research in this area is very limited. Effective modelling or up-scaling of GHG emissions from watershed to regional/ national scales is important for the improvement of global GHG budgets. Such up-scaling needs an accurate estimation of C and N inputs and outputs i.e. a balance coupled with net GHG emissions while considering all possible processes and pathways involved. So, studying the dynamics of C and N in CW systems is crucial as the forms of removed C and N are particularly pertinent to their potential for pollution swapping; global warming and water pollution.

Processes involved in N removal and N transformations in wetlands include sedimentation of particulates (Koskiaho, 2003); nitrification, denitrification and DNRA (Poach et al., 2003; Burgin et al., 2013), microbial assimilation and plant uptake-release (Findlay et al., 2003), anammox and deamox (DEnitrifying AMmonium OXidation). Constructed wetlands are a complex system which facilitates aerobic and anaerobic microsites. This means that nitrification and denitrification can occur simultaneously. Nitrification, denitrification and nitrifier denitrification are the processes responsible for the production of N_2O. Depending on the environmental conditions or management practices prevailing, a certain process will dominate e.g. denitrification is the dominant process in SF CWs (Beaulieu et al., 2011) but nitrifier denitrification is dominant in VSSF CWs (Wunderlin et al., 2013). Generally CWs are anaerobic but aquatic macrophytes can transport oxygen from the atmosphere to the rooting zone where it can sustain nitrification. Nitrate produced in the rooting zone can be taken up by plants or denitrified and/or converted back to NH_4^+ by DNRA. The DNRA is an anaerobic process where NO_3^- is transformed to NH_4^+ which can remain in that form only until it has contact with an aerobic environment (Tesoriero et al., 2000), after which it is oxidized to NO_3^- (Thayalakumaran et al., 2008). The fermentative bacteria which carry out DNRA are obligate anaerobes (Hill, 1996) and so cannot occupy all the niches that denitrifiers can. Takaya (2002) stated that more reducing state favours DNRA over denitrification. However, the conditions that favour the occurrence of either denitrification or DNRA are still in debate (Rütting et al., 2011). The differences between denitrification and DNRA may be due to the availability of organic matter,

because DNRA is favoured at high C:N ratio. The N_2O produced in soil can be consumed by denitrifiers and/or emitted to the atmosphere by ebullition and diffusion processes or via vascular plants. Denitrification has been estimated to be a significant N removal process but actual quantification data are scarce. Limited studies have estimated N losses by denitrification e.g. 19% (Mander et al., 2008) and 86% (Obarska-Pempkowiak and Gajewska, 2003) of the total N input based on the mass balance study. To our knowledge, no data are available on denitrification measurements in soil/subsoils of surface flow CWs. While many of these pathways transfer Nr (mainly NH_4^+ and N_2O) to the environment, other pathways can convert Nr to N_2 (e.g. denitrification, anammox and deamox). Anammox can remove NO_2^- and NH_4^+ as N_2 when the existing environment is hypoxic. The deamox can remove NO_3^- and NH_4^+ as N_2 where NO_3^- is converted to NO_2^- by autotrophic denitrification with sulphide (Kalyuznyi et al., 2006). In CWs, anammox and DEAMOX are not well understood and so it is crucial to identify which of the processes are occurring in a specific type of CWs and at what rate they occur. Once a process that provides N_2 as the end product is determined in a specific system then the CW management could be directed towards enhancement of that process. Hence, quantifying the rates of these processes under various CW types is required for improved N management towards lowering Nr in the environment.

The various components of the C cycle include: fixation of C by photosysnthesis, respiration, fermentation, methanogenesis and CH_4 oxidation with reduction of S, Fe and NO_3^-. Anaerobic methane oxidation coupled with denitrification, a recently proposed pathway of the C cycle (á Norði and Thamdrup, 2014; Haroon et al., 2013; Islas-Lima et al., 2004), can reduce CH_4 emissions in CWs. The C removal processes are sedimentation, microbial assimilation, gaseous emissions, dissolved C losses through water to ground and surface water bodies and chemical fixation (bonding with chemical ions). Net primary productivity (NPP) of wetland hydrophytes varies across CW type, season, climatic region and local environmental conditions. For example taking the same plant species results can vary remarkably in different geographical regions (Brix et al., 2001). Carbon mineralization in sediments depends on the redox chemistry of soil, availability and quality of C (labile or recalcitrant) and temperature. In CWs, C cycling is very complex due to the changes in redox chemistry which regulates production and consumption of CO_2 and CH_4 (Brix et al., 2001). In low redox conditions with limited dissolved oxygen, methanogens can consume DOC and thus it is conducive to CH_4 production. The C:N ratios of wastewater affect microbial growth and development which in turn affect their response to C and N cycles and GHG emissions. Previous research on the effects of C:N ratios on nutrient removal

and GHG emissions are limited. A few examples include Yan et al. (2012) and Zhao et al. (2014), which showed lower CO_2 and CH_4 emissions at C:N ratios between 2.5:1 and 5:1, but this lower range of C:N ratios decreased TOC removal. Hence, investigation of the influence of C:N ratio on nutrient removal efficiencies and GHG emissions across CW and management types is crucial. In summer, oxygen diffusion to the topsoil can reduce methanogenesis and stimulate CH_4 oxidation (Grünfeld and Brix, 1999). However an increase in temperature can decrease dissolved oxygen in deeper subsoil layers, which can enhance CH_4 production. Only a limited number of studies have considered CH_4 and CO_2 efflux from CWs (e.g. Mander et al., 2008). Like for all biochemical reactions temperature, increases C and N turnover in CWs causing high variations in GHG emissions under same plant species in different regions (temperate/ tropical/ arctic). These variations need to be considered while extrapolating GHG emissions for different types of CWs under different management practices.

Emissions of GHG in CWs can vary across CW typologies e.g. surface flow or subsurface flow (Van der Zaag et al., 2010). A summary of N_2O, and CO_2 and CH_4 were presented in Table 3 and Table 4, respectively. Generally CH_4 emissions are higher in SF CWs than the SSF CWs (Table 3) but vary with seasons. Nitrous oxide and CO_2 emissions are higher in VSSF CWs than HSSF and SF CWs. The N_2O emissions factors (EF; N_2O/TN input*100) ranged from 0.61 ± 0.21% in SF CWs to 1.01 ± 0.48% in VSSF CWs. The EF for CH_4 emissions ranged from 1.27 ± 0.31% in VSSF CWs to 16.8 ± 3.8% in SF CWs. The GHG from CWs can vary between vegetated and non-vegetated systems (Table 5). Aquatic plants play important role on GHG production (provide C necessary for microbes) and transport to the atmosphere by releasing GHG through their interconnected internal gas lacunas (Laanbroek, 2010). Emergent plants can transport atmospheric oxygen to the rooting zone and contribute to increased N_2O and CO_2 production and CH_4 consumption (Brix, 1997). Vascular plants can exchange GHG between the rooting zone and atmosphere (Yavitt and Knapp, 1998). Vegetation and its composition affect the nutrient dynamics and the production, consumption and transport of greenhouse gases and hence their exchange between wetlands and atmosphere (Ström et al., 2003, 2005; Søvic et al., 2006; Johansson et al., 2003). They can also affect the biogeochemistry of CWs due to the differences in their growth and development, longevity, root systems, root density, root depth and microbial ecology in the rhizosphere. Some plant litter as it decomposes can release organic matter with lignocellulose and humic compounds that are more or less labile or stable in nature than others. Release of low molecular weight organic matter that is labile in nature is more likely to produce GHGs than stable forms. For example, *Z. latifolia* showed higher nutrient removal and CH_4

fluxes than *P. australis* (Inamori et al., 2007). *Z. lotifolia* root system is shallow and the activity of methanotrophs was primarily confined to the top soil. The root systems of *P. australis* are deeper which is more favourable to oxidise CH_4. Fluctuating water table in CWs has significant impacts on GHG dynamics. Pulsing hydrologic regimes decreases CH_4 but increases N_2O emissions. In aerobic and anaerobic conditions caused by pulsing hydrology, incomplete nitrification and denitrification increase N_2O emissions. However, the effects of pulsing hydrologic regimes on GHG emissions are contradictory. For example, intermittent hydrologic regimes decrease both N_2O (Sha et al., 2011) and CH_4 emissions (Song et al., 2010). Highly contrasting results on gas emissions with fluctuating water levels have been reported and the controlling mechanisms are unclear (Elberling et al., 2011). Therefore, the assessment of GHG emissions in various types of CW (surface flow, subsurface flow; vertical and horizontal), vegetation used (vegetated, nonvegetated, plant species composition) and under different management systems (HLR, HRT, soil used and water table) is necessary in light of the national and global GHG budgets and mitigation of GHG emissions. In addition, such measurements will help scientists, environmental managers and policy makers to adopt environmental friendly construction and management of CWs. The enhanced reduction of N_2O to N_2 needs further elucidation. Assessment of the reactive versus the benign forms of C and N transformation products in various types of CWs will give insights into their environmental friendly design and management.

5. Surface Emissions vs Subsurface Export of C and N

Dissolved GHG produced in soils and subsoils can be emitted to atmosphere by transpiration of vascular plants (from within the rooting zone), ebullition and diffusion from soils. Elberling et al. (2011) reported that in wetlands, the transport of gases through subsoil occurs both via diffusive transport in the pores and through the vascular plants. Surface emissions of GHG from CWs are well recognised and have been commonly measured by chamber methods. The GHG produced in CWs can also be transported to the groundwater with the percolating water and emit to atmosphere upon discharge to surface waters. It can also flow towards surface waters by advective transport and or by dispersion of groundwater. Dissolved nutrients can be preferentially leached down into deeper soil layers and groundwater via different pathways (e.g. root channels). The Nr delivered to groundwater can be transformed *in situ* to other reactive or benign forms. Hence, quantification of such Nr loadings to groundwater and their *in situ* consumption (e.g. N_2O to N_2 or CH_4 to CO_2) is required to understand their environmental consequences. In addition, DON, NO_3^- and NH_4^+ and DOC delivered to surface waters can undergo biochemical reactions

and produce N_2O, CO_2 and CH_4 in streams and estuaries. Ström et al. (2007) measured a considerable quantity of CH_4 in porewater and found a correlation between the surface emissions and porewater CH_4 concentrations in vegetated wetlands. Measuring only the surface emissions of GHG can miss substantial quantity of GHG emissions from CWs. For example, Riya et al. (2010) measured indirect emissions of CH_4 and N_2O accounting for 2.9 and 87% of total emissions. Measuring porewater GHG and linking these to the surface emissions and subsurface export to groundwater below CWs will help to estimate a better GHG balance from both a national and global context. Elberling et al. (2011) linked subsurface gas concentrations in wetlands to the surface fluxes using a diffusion model which has demonstrated the need for future studies on subsurface GHG production, consumption and net GHG emissions in CWs ecosystem in a climate change context. It is important to characterise soils and subsoils physical (e.g. texture, bulk density) and hydraulic (development of a soil water characteristic curve) properties and to assess their potential to percolate dissolved nutrients and gases in the solute phase to the underlying groundwater. To our knowledge, indirect pathway of GHG emissions from CWs has never been reported despite the fact that this would appear to have a high biogeochemical potential to produce and exchange GHG. The balance between N and C input and output flows between CWs and aquatic and atmospheric environments together with the direct and indirect emissions of C and N species could be an important input to global C and N budgets.

6. Hydrogeochemistry below CWs

The CWs can be designed with or without a clay liner or a compacted soil bed at the base, which can lead to large differences in permeability of the underlying layers. The variation in permeability of a CW soil bed will affect solute, nutrient and GHG flows and their interactions with the underlying groundwater (Dzakpasu et al., 2012; 2014). Groundwater hydrogeochemistry below CWs can therefore provide a unique insight into such interactions. An example of such interactions would be between nutrient rich water discharging from CW cells mixing with laterally moving regional groundwater. It should be noted that groundwater can also discharge into CWs depending on the hydraulic gradients. This necessitates that fully screened multiple multi-level piezometers or boreholes are installed at such sites to elucidate groundwater flow direction, hydraulic gradients and hydraulic permeability. Such monitoring networks allow water samples to be taken and the sources of nutrients in the groundwater body below CWs to be identified. The local site hydrology (precipitation, groundwater table fluctuations and evapotranspiration) has greater impacts on the pollutant removal by physical attenuation and by

biochemical transformations. Hydrogeochemical studies on an accurate spatial and temporal resolution should explain the effects of precipitation on nutrient removal by dilution as well in situ nutrient turnover. Effective CW management requires an understanding of the effects of wetland hydrology on the physical and biochemical attenuation of nutrients in order to assess their impacts on the surface emissions and subsurface export of nutrients and GHG. Data on the species of N in groundwater below the CWs are required to provide an in-depth understanding of wetland ecosystem services, particularly if CW systems have the potential to leak pollutants down into the groundwater (Dzakpasu et al., 2014). Higher NH_4^+ concentrations in groundwater below the CWs than the effluents are often reported (Harrington et al., 2007; Dzakpasu et al., 2012). Therefore, questions arise with respect to NH_4^+ concentrations in groundwater below the CWs if they have been transported from CWs. Linking geochemistry of groundwater below CWs to site hydrology, water table fluctuations and soil/subsoil physico-chemical properties is required to elucidate the major environmental drivers of C and N removal and or pollution swapping. The quality of groundwater underlying CWs with regards to the Nr species is largely unknown.

7. Methodological Developments

Quantification of the occurrence and magnitude of N transformations and hydrochemical properties is crucial to improve the assessment of CW ecosystem services and to minimize their potential for pollution swapping. Measurement of GHG using the closed chamber method is widely used but has large uncertainty in estimating the diurnal variability due to internal changes in temperature and physical access to the chambers over 24 h time period. Gas ebullition and diffusion measurements are quite challenging in CWs covered by vegetation because of the difficulties in estimation of gas transfer velocity. Application of eddy covariance method is not likely for most of the CWs as it requires larger surface area (> several ha) to avoid contribution of surrounding area and complication of GHG foot printing. In addition, heterogeneity in CWs surface increases uncertainty in better mixing of gas turbulence before it goes to the sensor. Combination of chamber, ebullition and diffusion methods in a single system could minimise the uncertainly in GHG estimation. Methane ebullition measurement was found to be similar to surface emissions by chamber method but N_2O and CO_2 ebullition measurements were lower than the surface emissions (Søvik et al., 2006). Use of *in situ* microcosm study and soil core incubation methods can give better estimation of N_2O, CO_2 and CH_4 production and consumption in light with the better GHG and C and N budgets. With the recent advancement of isotope pairing and dilution techniques, single or simultaneously occurring C and N transformation processes can be quantified in laboratory or *in situ* conditions (Huygens et al., 2013; Müller et al.,

2014). The isotope technique relies on the introduction of a known amount of ^{14}C and or ^{15}N into the CW system and then quantification of C and N concentrations and isotopic compositions through different C and N pools after incubation for a specific period. Laboratory methods involve collection of intact soil/sediment cores, with subsequent incubation in the laboratory. The *in situ* field techniques involve release of ^{14}C/ ^{15}N solution *in situ* in the CWs soils. Incubation of intact soil cores with differentially labelled $^{15}NH_4^{14}NO_3$ and $^{14}NH_4^{15}NO_3$ can be used to quantify the rates of different N transformation processes (Rütting and Müller, 2008). The quantification of simultaneously occuring N transformation rates rely on the analysis with appropriate ^{15}N tracing models. Development in the recent years in ^{15}N tracing techniques is now available, which are able to identify process specific NO_2^- pools (Rütting and Müller, 2008), pathways specific N_2O production and emission as well as N_2O/N_2 ratios (Müller et al., 2014). Traditional techniques for investigation of gross N dynamics in sediments (Blackburn, 1979) could be combined with the latest ^{15}N tracing techniques where all N transformation rates are included, that are important in wetlands and under anoxic condition (Huygens et al., 2013). Thus, current models should consider processes such as anammox and/or deamox and then be tested in CW environments under various conditions. Denitrification in porewater samples can be measured by analysing samples for dissolved N_2 in a membrane inlet mass spectrometer (MIMS; Kana et al., 1994) and N_2O in gas chromatograph (GC; Jahangir et al., 2012). The studies of natural abundance of ^{15}N and ^{18}O ($\delta^{15}N$ and $\delta^{18}O$) in NO_3^- is an insightful tool for the investigation of the sources, fate and transformational processes of N in a system (e.g. in shallow groundwater, Baily et al., 2011). The *in situ* NO_3^- push-pull method has been used to determine denitrification in shallow groundwater (<3 m) in riparian wetlands (Addy et al., 2002; Kellogg et al., 2005) and in deep groundwater in arable/grassland (Jahangir et al., 2013).

The isotope based techniques can also be extended to other elements e.g., a ^{33}P tracing model has been developed recently to study phosphorous cycle in soil (Müller and Bünemann, 2014). These techniques can be applied in the study of C, N and P biogeochemistry in aquatic environment. In addition measurements of DOC and gases (CO_2 and CH_4) will provide insights into the C consumption and transformation associated with the N transformations. Carbon and N dynamics are influenced by the interacting effects of soil conditions with microbial community structure and functioning. Microbial functioning involves transcription of genes, translation of messenger RNA and activity of enzymes (Firestone et al., 2012). As such, activities of microbial communities under various environmental conditions and how these contribute to C and N dynamics is a very important area of future research (Müller and Clough, 2014). Molecular

approaches can be important tools to identifying and quantifying the genes that code for enzymes mediating C and N cycles (Peterson et al., 2012). These tools help assess the relationships among genes, environmental controllers and the rates of C and N processes. The scientific tools and multidisciplinary techniques are now available to better understand C and N transformation rates, processes and factors controlling the unwanted emission of N and C products to the environment.

8. Conclusions and Recommendations

The transformational processes on a mixture of contaminants within and below CWs can cause pollution swapping. A holistic assessment of C and N dynamics in CWs is needed to fully understand their removal, transport and impact on water quality and emissions to atmosphere. Mixed contaminants entering CWs and those formed within and underneath CWs during transformational processes must be considered in future studies. The overall balance of these constituents will decide on whether a CW is a pollution source or a sink. This will necessitate a higher degree of multi-level spatial and temporal monitoring and use of multi-disciplinary techniques both *in* and *ex situ* to fully characterise all pathways of C and N loss. We cannot yet suggest any best engineering aspect of CW type with the design optima in terms of nutrient removal and GHG mitigation because the research based information are not yet abundant. For this, transformation kinetics of C and N and net GHG emissions through all possible pathways are required to provide a holistic assessment. However, combination of various types of CW (several parallel VSSF CWs followed by HSSF CWs) and a mixture of plant types could provide higher removal efficiency and lower GHG emissions. A conceptual model highlighting the state of knowledge in this area and the specific research need to be done was presented in Figure 1.

Subsurface export of nutrients and GHG to groundwater through leaching through convective/dispersion transport or by preferential flow paths created by dead roots and holes of burrowing invertebrates and subsequent transport to surface water bodies should be accounted in CW management. Reducing the saturated hydraulic conductivity below the wetland bed will help reduce nutrients leaching to groundwater below the CWs cells. The reactive versus the benign forms of the N transformation products should be evaluated. Data on when, where and at what rates denitrification, deamox and anammox occur in CWs are needed as well as what are the key factors that control such processes. The provenance of NH_4^+ in groundwater below CW cells and its impact on down gradient receptors needs further elucidation. This review shows that CWs have the potential to produce N_2O, DON, DOC, dissolved inorganic C (DIC), CO_2 and CH_4 and

that the GHG produced in CWs can be emitted to atmosphere. They can also be exported to fresh waters via groundwater and degassed upon discharge to surface waters. Moreover, the DOC and DIC transferred to the fresh water sediments (rivers and lakes) can produce GHG that in turn emit to atmosphere. The amount of C and N exported from terrestrial ecosystem via the subsurface pathway to fresh waters has been the missing piece of our understanding of global C and N budgets. It is clear that data on the various C and N species along with the GHG in various CWs systems are thus crucial to make a robust input-output balance of C and N in such a rising and engineered ecosystem. Spatial and temporal variation of GHG emissions in CWs under different management systems is critical to get much more rigorous estimates of emission factors. These data will reduce the existing uncertainties in global C and N budgets.

Managing wetting and drying spells (pulsing hydrology) in CWs can enhance NH_4^+ removal. Similarly, oxidation of organic C will increase CO_2 production and in anaerobic condition it can be reduced to CH_4. This requires more research into the C and N cycle processes over the wetting drying spells which is now possible with the advancement in $^{14}C/^{15}N$ tracing and modelling techniques. The selection of plant species is important to increase nutrients removal, sequester more C and decrease GHG emissions which are an area that still requires more research across types of CWs and countries. Further research is also still needed to be done on the impacts of hydraulic retention time on nutrients dynamics and removal. Rates of nutrient accumulation or fixation in soils and their *in situ* transformation in CWs need to be quantified to know their contribution to C sequestration and GHG emissions.

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Author's Contributions

The first author, M.M.R. Jahangir has reviewed articles in the relevant area, analysed results and identified knowledge gaps and prepared the draft paper. All co-authors were directly involved in preparation of the paper and edited the paper for its improvement.

References

á Norði, K. and Thamdrup, B.: Nitrate-dependent anaerobic methane oxidation in a freshwater sediment, Geochimica et Cosmochimica Acta, 132, 141-150, 2014.

571 Addy, K., Kellogg, D.Q., Gold, A.J., Groffman, P.M., Ferendo, G., and Sawyer, C.: *In situ* push-
572 pull method to determine groundwater denitrification in riparian zones, J. Environ. Qual.,
573 31, 1017-1024, 2002.
574 Bachand, P.A.M, and Horne, A.J.: Denitrification in constructed free-water surface wetlands: II.
575 Effects of vegetation and temperature, Ecol. Eng.,14,17–32, 2000.
576 Baily, A., Rock, L., Watson, C.J., and Fenton, O.: Spatial and temporal variations in groundwater
577 nitrate at an intensive dairy farm in south-east Ireland: Insights from stable isotope data,
578 Agril. Ecosysts. Environ., 308-318, 2011.
579 Beaulieu, J.J., Tank, J.L., Hamilton, S.K., Wollheim, W.M., Hall, R.O., and Mulholland, P.J.:
580 Nitrous oxide emission from denitrification in stream and river networks, Proc. Natl.
581 Acad. Sci., 108, 214–219, 2011.
582 Blackburn, T.H.: Methods for measuring rates of NH_4^+ turnover in anoxic marine sediments,
583 using a ^{15}N-NH_4^+ dilution technique, Appl. Environ. Microbiol., 37(4), 760-765, 1979.
584 Bojcevska, H., and Tonderski, K.: Impact of loads, season, and plant species on the performance
585 of a tropical constructed wetland polishing effluent from sugar factory stabilization ponds,
586 Ecol. Eng., 29, 66–76, 2007.
587 Brisson, J., and Chazarenc, F.: Maximising pollutant removal in constructed wetlands: should we
588 pay more attention to macrophytes species selection?, Sci. Total Environ., 407, 3923-
589 3930, 2009.
590 Brix, H.: Do macrophytes play a role in constructed treatment wetlands, Water Sci. Technol., 35,
591 11-17, 1997.
592 Burgin, A.J., Hamilton, S.K., Gardner, W.S., and McCarthy, M.J.: Nitrate reduction,
593 denitrification, and dissimilatory nitrate reduction to ammonium in wetland sediments, in:
594 R.D. DeLaune, K.R. Reddy, C.J. Richardson, and J.P. Megonigal, (Eds.), Methods in
595 Biogeochemistry of Wetlands, SSSA Book Series, no. 10, Madison, USA, 307-325, 2013.
596 Calheiros, C.S.C., Rangel, A.O.S.S., and Castro, P.M.L.: Constructed wetland systems vegetated
597 with different plants applied to the treatment of tannery wastewater, Water Res. 41, 1790–
598 98, 2007.
599 Clair, T.A., Arp, P., Moore, T.R., Dalva, M., and Meng, F.R.: Gaseous carbon dioxide and
600 methane, as well as dissolved organic carbon losses from a small temperate wetland under
601 a changing climate, Environ. Pollut., 116(1), S143-S148, 2002.
602 Conley, D.J., Paerl, H.W., Howarth, R.W. Boesch, D.F., Seitzinger, S.P., Havens, K.E., Lancelot,
603 C., and Likens, G.E.: Controlling eutrophication: Nitrogen and phosphorus, Science, 323,
604 1014–1015, 2009.

da Motta Marques D.M.L., Leite, G.R., and Giovannini, S.G.T.: Performance of two macrophyte species in experimental wetlands receiving variable loads of anaerobically treated municipal wastewater, Water Sci. Technol., 44, 311–6, 2001.

Dunne, E.J., Culleton, N., O'Donovan, G., Harrington, R., and Olsen, A.E.: An integrated constructed wetland to treat contaminants and nutrients from dairy farmyard dirty water, Ecol. Eng., 24(3), 219-232, 2005.

Dzakpasu, M., Scholz, M., Harrington, R., McCarthy, V. and Jordan, S.: Groundwater Quality Impacts from a Full-Scale Integrated Constructed Wetland, Groundwater Monit. Rem., 34(3), 51-64, 2014.

Dzakpasua, M., Scholz, M., Harrington, R., Jordan. S.N., and McCarthy, V.: Characterising infiltration and contaminant migration beneath earthen-lined integrated constructed wetlands, Ecol. Eng., 41, 41-51, 2012.

Elberling, B., Louise A., Christian, J.J., Hans, P.J., Michael K., Ronnie N.G., and Frants, R.L.: Linking soil O_2, CO_2, and CH_4 concentrations in a wetland soil: implications for CO_2 and CH_4 fluxes, Environ. Sci. Technol., 45, 3393–3399, 2011.

Fenton, O., Healy, M.G., Brennan, F., Jahangir, M.M.R., Lanigan, G.J., Richards, K.G., Thornton, S.F., and Ibrahim, T.G.: Permeable reactive interceptors – blocking diffuse nutrient and greenhouse gas losses in key areas of the farming landscape, J. Agric. Sci., 152, S71-S81, 2014.

Fey, A., Benckiser, G., and Ottow, J.C.G.: Emissions of nitrous oxide from a constructed wetland using a ground filter and macrophytes in waste-water purification of a dairy farm, Biol. Fertil. Soils, 29(4), 354–359, 1999.

Findlay, S., Groffman, P., and Dye, S.: Effects of *Phragmites australis* removal on marsh nutrient cycling, Wetlands Ecol. Manage., 11, 157-165, 2003.

Firestone, M., Blazewicz, S., Peterson, D.G., and Placella, S.: Can molecular microbial ecology provide new understanding of soil nitrogen dynamics, in: Richards, K.G., Fenton, O., Watson, C. J. (Eds.), Proceedings of the 17th Nitrogen Workshop, 11-14, 2012.

Fraser, L.F., Carty, S.M., and Steer, D.: A test of four plant species to reduce total nitrogen and total phosphorus from soil leachate in subsurface wetland microcosms, Bioresour. Technol., 94, 185–92, 2004.

Galloway, J.N., Aber, J.D., Erisman, J.W., Seitzinger, S,P., Howarth, R.W., Cowling, E.B., and Cosby, B.J.: The nitrogen cascade, BioSci., 53(4), 341-356, 2003.

637 Garcia, J., Capel, V., Castro, A., Ruiz, I., and Soto, M.: Anaerobic biodegradation tests and gas
638 emissions from subsurface flow constructed wetlands, Bioresour. Technol., 98(16), 3044–
639 3052, 2007.
640 Gold, A.J., Eddy, K., David, M.B., Schpper L.A., and Needelman, B.A.: Artificial sinks:
641 opportunities and challenges for managing offsite nitrogen losses, J. Contemp. Water Res.
642 Edu., 151, 9-19, 2013.
643 Gray, N.F.: Drinking water quality, 2nd edn., Cambridge Univ. Press, Cambridge, UK, 2008.
644 Grünfeld, S., and Brix, H.: Methanogenesis and methane emissions: effects of water table,
645 substrate type and presence of *Phragmites australis*, Aquatic Bot., 64, 63-75, 1999.
646 Gui, P., Inamori, R., Matsumura, M., and Inamori, Y.: Evaluation of constructedwetlands by
647 waste water purification ability and greenhouse gas emissions, Water Sci. Technol., 56(3),
648 49–55, 2007.
649 Haroon, M. F., Hu, S., Shi, Y., Imelfort, M., Keller, J., Hugenholtz, P., Yuan, Z., and Tyson,
650 G.W.: Anaerobic oxidation of methane coupled to nitrate reduction in a novel archaeal
651 lineage, Nature, 500, 567-570, 2013.
652 Harrington, R., Carroll, P., Carty, A.H., Keohane, J., and Ryder, C.: Integrated constructed
653 wetlands: concept, design, site evaluation and performance, Int. J. Water, 3(3), 243-255,
654 2007.
655 Haule, A.T., Pratap, H.B., Katima, H.J.Y., Bwette, M., and Njau, K.: Nitrogen removal from
656 domestic wastewater in subsurface flow constructed wetlands by indigenous macrophytes
657 in the tropics: a comparative study for six potential macrophytes in Tanzania, in: Bwette,
658 M, (ed.), Eighth IWA international conference on the use of constructed wetlands in water
659 pollution control, 16–19 September, Tanzania, p. 938–51, 2002.
660 Healy, M.G., Ibrahim, T.G., Lanigan, G., Serrenho, A.J., and Fenton, O.: Nitrate removal rate,
661 efficiency and pollution swapping potential of different organic carbon media in
662 laboratory denitrification bioreactors, Ecol. Eng., 40, 198-209, 2011.
663 Healy, M.G., Barrett, M., Lanigan, G.J., João Serrenho, A., Ibrahim, T.G., Thornton, S.F., Rolfe,
664 S.A., Huang, W.E., Fenton, O.: Optimizing nitrate removal and evaluating pollution
665 swapping trade-offs from laboratory denitrification bioreactors, Ecol. Eng., 74, 290-301,
666 2014
667 Hill, A.R.: Nitrate removal in stream riparian zones, J. Environ. Qual., 25, 743-755, 2010.
668 Huygens, D., Trimmer, M., Rütting, T., Müller, C., Heppell, C.M., Lansdown, K., and Boeckx,
669 P.: Biogeochemical N cycling in wetland ecosystems: ^{15}N isotope techniques, Methods in

biogeochemistry of wetlands, K.R. Reddy, J.P. Megonigal and R.D. Delaune, (Eds.), Soil Sci. Soc. Amer., 30, 553-591, 2013.

Inamori, R., Gui, P., Dass, P., Matsumura, M., Xu, K.Q., Kondo, K., Ebie, Y., and Inamori, Y.: Investigating CH_4 and N_2O emissions from eco-engineering wastewater treatment processes using constructed wetland microcosms, Proc. Biochem, 42, 363-373, 2007.

IPCC: 2013 Supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: wetlands, edited by: Hirashi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M., Troxler, T.G., Switzerland, IPCC Task Force on National Greenhouse Gas Inventories, 354, 2014.

Islas-Lima, S., Thalasso, F., and Gomez-Hernandez, J.: Evidence of anoxic methane oxidation coupled to denitrification, Water Res., 38(1), 13-16, 2004.

Jahangir, M.M.R., Johnston, P., Addy, K., Khalil, M.I., Groffman, P., and Richards, K.G.: Quantification of in situ denitrification rates in groundwater below an arable and a grassland system, Water Air Soil Pollut., 224, 1693, doi:10.1007/s11270-013-1693-z, 2013.

Jahangir, M.M.R., Johnston, P., Grant, J., Somers, C., Khalil, M.I., and Richards, K.G.: Evaluation of headspace equilibration methods for measuring greenhouse gases in groundwater, J. Environ. Manage., 111, 208-212, 2012.

Johansson, A.E., Gustavsson, A.M., Öquist, M.G., and Svensson, B.H.: Methane emissions from a constructed wetland treating wastewater – seasonal and spatial distribution and dependence on edaphic factors, Water Res., 38 (18), 3960–3970, 2004.

Johansson, A.E., Kasimir-Klemedtsson, A., Klemedtsson, L., and Svensson, B.H.: Nitrous oxide exchanges with the atmosphere of a constructed wetland treating wastewater, Tellus, 55B, 737–750, 2003.

Johansson, E.: Constructed wetlands and deconstructed discorses–greenhouse gas fluxes and discorses on purifying capacities, Dept. Water Environ. Studies, Linköping Univ., Sweden, 2002.

Kalyuzhnyi, S., Gladchenko, M., Mulder, A., and Versprille, B.: DEAMOX-new biological nitrogen, Water Res., 40, 3637-3645, 2006.

Kana, T.M., Darkangelo, C., Hunt, M.D., Oldham, J.B., Bennett, G.E, and Cornwell, J.C.: Membrane inlet mass spectrometer for rapid high precision determination N_2, O_2 and Ar in environmental water samples, Anal. Chem., 66, 4166-4170, 1994.

Kato, K., Koba, T., Ietsugu, H., Saigusa, T., Nozoe, T., Kobayashi, S., Kitagawa, K., and Yanagiya, S.: Early performance of hybrid reed bed system to treat milking parlour

wastewater in cold climate in Japan, in: 10th international conference wetland systems for water pollution control, MAOTDR, Lisbon, Portugal, pp. 1111-1118, 2006.

Kellogg, D.Q., Gold, A.J., Groffman, P.M., Addy, K., Stolt, M.H., and Blazejewski, G.: *In situ* groundwater denitrification in stratified, permeable soils underlying riparian wetlands, J. Environ. Qual., 34, 524-533, 2005.

Koskiaho, J., Ekholm, P., Räty, M., Riihimäki, J., and Puustinen, M.: Retaining agricultural nutrients in constructed wetlands- experiences under boreal conditions, Ecol. Eng., 20(1), 89–103, 2003.

Laanbroek, H.J.: Methane emission from natural wetlands: interplay between emergent macrophytes and soil microbial processes, A mini-review, Annals Bot., 105, 141–153, 2010.

Langergraber, G.: Modeling of processes in subsurface flow constructed wetlands: A review. Vadose Zone, J. 7, 830-842, 2008.

Lee, C., Fletcher, T.D., and Sun, G.: Nitrogen removal in constructed wetland systems, Eng. Life Sci., 9(1), 11-22, 2009.

Liikanen, A., Huttunen, J.T., Kaijalainen, S.M., Heikkinen, K. Vdisdinen, T.S., Nykiinen,H., and Martikainen, P.J.: Temporal and seasonal changes in greenhouse gas emissions from a constructed wetland purifying peat mining runoff water, Ecol. Eng., 26, 241-251, 2006.

Liu, C., Xu, K., Inamori, R., Ebie, Y., Liao, J., and Inamori, Y.: Pilot-scale studies of domestic wastewater treatment by typical constructed wetlands and their greenhouse gas emissions, Front. Environ. Sci. Eng. China, 3(4), 477–482, 2009.

Luederitz, V., Eckert, E., Lange-Weber, M., Lange, A., and Gersberg, R.M.: Nutrient removal efficiency and resource economics of vertical flow and horizontal flow constructed wetlands, Ecol. Eng., 18, 157–171, 2001.

Luo, W.G., Wang, S.H., Huang, J., and Qian, W.Y.: Denitrification by using subsurface constructed wetland in low temperature. China Water Wastewater (in Chinese), 21, 37–40, 2005.

Makino, W., Cotner, J.B., Sterner, R.W., and Elser, J.J.: Are bacteria more like plants or animals? Growth rate and resource dependence of bacterial C:N:P stoichiometry, Funct. Ecol., 17, 121–130, 2003.

Maltais-Landry, G., Maranger, R., Brisson, J., and Chazarenc, F.: Greenhouse gas production and efficiency of planted and artificially aerated constructed wetlands, Environ. Pollut., 157, 748-754, 2009.

737 Mander, U., Lõhmus, K., Teiter, S., Mauring, T., Nurk, K., and Augustin, J.: Gaseous fluxes in
738 the nitrogen and carbon budgets of subsurface flow constructed wetlands, Sci. Total
739 Environ., 404, 343-353, 2008.
740 Mander, Ü., Lõhmus, K., Teiter, S., Nurk, K., Mauring, T., and Augustin, J.: Gaseous fluxes from
741 subsurface flow constructed wetlands for wastewater treatment, J. Environ. Sci. Health A
742 40(6–7), 1215–26, 2005.
743 Mander, Ü., Maddison, M., Soosaar, K., and Karabelnik, K.: The impact of intermit-tent
744 hydrology and fluctuating water table on greenhouse gas emissions from subsurface flow
745 constructed wetlands for wastewater treatment, Wetlands, 31(6), 1023–1032, 2011.
746 Mitsch, W.J., and Gosselink, J.G.: Wetlands, 3rd Edn. New York, John Wiley & Sons, p. 936,
747 2000.
748 Müller, C. and Bünemann, E.K.: A ^{33}P tracing model for quantifying gross P transformation rates
749 in soil, Soil Biol. Biochem., 76, 218-226, 2014.
750 Müller, C. and Clough, T.J.: Advances in understanding nitrogen flows and transformations: gaps
751 and research pathways, J. Agril. Sci., 152, S1, 34-44, 2014.
752 Müller, C., Laughlin, R.J., Spott, O., and Rütting, T.: Quantification of N_2O emission pathways
753 via a ^{15}N tracing model, Soil Biol. Biochem., 72, 44-54, 2014.
754 Mustafa, A., and Scholz, M.: Nutrient accumulation in *Typha latifolia* L. and sediment of a
755 representative integrated constructed wetland, Water Air Soil Pollut., 219, 329-341, 2011.
756 Nguyen, L.M.: Organic matter composition, microbial biomass and microbial activity in gravel-
757 bed constructed wetlands treating farm dairy wastewaters, Ecol. Eng., 16, 199-221, 2000.
758 O'Luanaigh, N.D., Goodhue, R., and Gill, L.W.: Nutrient removal from on-site domestic
759 wastewater in horizontal subsurface flow reed beds in Ireland, Ecol. Eng., 36, 1266-1276,
760 2010.
761 Obarska-Pempkowiak, H., and Gajewska, M.: The dynamics of processes responsible for
762 transformation of nitrogen compounds in hybrid wetlands systems in a temperate climate,
763 in: J. Vymazal (Eds.) Wetlands- Nutrients, Metals and Mass Cycling. Backhuys
764 Publishers, Leiden, The Netherlands, p.129–42. 2003.
765 Pangala, S.R., Reay, D.S., and Heal, K.V.: Mitigation of methane emissions from constructed
766 farm wetlands, Chemosphere, 78, 493-499, 2010.
767 Peterson, D.G., Blazewicz, S., Herman, D.J., Firestone, M., Turetsky, M. and Waldrop, M.:
768 Abundance of microbial genes associated with nitrogen cycling as indices of
769 biogeochemical process rates across a vegetation gradient in Alaska, Environ. Microbiol.,
770 14, 993-1008, 2012.

771 Poach, M.E., Hunt, P.G., Vanotti, M.B., Stone, K.C., Matheny, T.A., Johnson, M.H., and Sadler
772 E.J.: Improved nitrogen treatment by constructed wetlands receiving partially nitrified
773 liquid swine manure, Ecol. Eng., 20, 183-197, 2003.
774 Rabalais, N.N., Diaz, R.J. Levin, L.A. Turner, R.E., Gilbert, D., and Zhang, J.: Dynamics and
775 distribution of natural and human-caused coastal hypoxia, Biogeosci., 7, 585–619, 2010.
776 Reay, D.S.: Title: Fertilizer 'solution' could turn local problem global - Protecting soil and water
777 from pollution may mean releasing more greenhouse gas, Nature, 427, 485-485, 2004.
778 Riya, S., Zhou, S., Nakashima, Y., Terada, A., Hosomi, M.: Direct and indirect greenhouse gas
779 emissions from vertical flow constructed wetland planted with forage rice, Kagaku
780 Kogaku Ronbunshu, 36, 229–236, 2010.
781 Rütting, T. and Müller, C.: Process-specific analysis of nitrite dynamics in a permanent grassland
782 soil by using a Monte Carlo sampling technique, Euro. J. Soil Sci., 59: 208-215, 2008.
783 Rütting, T., Boeckx, P., Müller, C., and Klemedtsson, L.: Assessment of the importance of
784 dissimilatory nitrate reduction to ammonium for the terrestrial nitrogen cycle, Biogeosci.,
785 8, 1169-1196, 2011.
786 Seitzinger, S.P., Sanders, R.W., and Styles, R.: Bioavailability of DON from natural and
787 anthropogenic sources to estuarine plankton, Limnol. Oceanogr., 47, 353–366, 2002.
788 Sha, C.Y., Mitsch, W.J., Mander, Ü., Lu, J.J., Batson, J., Zhang, L., and He, W.S.: Methane
789 emissions from freshwater riverine wetlands, Ecol. Eng., 37:16–24, 2011.
790 Shamir, E., Thompson, T.L, Karpisak, M.M., Freitas, R.J., and Zauderer, J.: Nitrogen
791 accumulation in a constructed wetland for dairy wastewater treatment, J. Amer. Water
792 Resour. Assoc., 37(2), 315-325, 2001.
793 Solano, M.L., Soriano, P., Ciria, M.P.: Constructed wetlands as a sustainable solution for
794 wastewater treatment in small villages, Biosyst. Eng., 87, 109–18, 2004.
795 Song, K., Lee, S.H., and Kang, H.: Denitrification rates and community structure of denitrifying
796 bacteria in newly constructed wetland, Eur. J. Soil Biol., 47(1), 24-29, 2011.
797 Song, K., Lee, S.H., Mitsch, W.J., and Kang, H.: Different responses of denitrification rates and
798 denitrifying bacterial communities to hydrological pulses in created wetlands, Soil Biol.
799 Biochem., 42, 1721–1727, 2010.
800 Søvik, A.K., Augustin, J., Heikkinen, K., Huttunen, J.T., Necki, J.M., Karjalainen, S.M., Klove,
801 B., Liikanen, A., Mander, U., Puustinen, M., Teiter, S., and Wachniew, P.: Emission of
802 the greenhouse gases nitrous oxide and methane from constructed wetlands in Europe. J.
803 Environ, Qual., 35, 2360-2373, 2006.

804 Stadmark, J., and Leonardson, L.: Emissions of greenhouse gases from ponds constructed for
805 nitrogen removal, Ecol. Eng., 25, 542-551, 2005.
806 Ström, L., Ekber,g A., Mastepanov M., and Christensen, T.R.: The effect of vascular plants on
807 carbon turnover and methane emissions from a tundra wetland, Global Change Biol. 9,
808 1185–1192, 2003.
809 Ström, L., Lamppa, A., and Christensen, T.R.: Greenhouse gas emissions from a constructed
810 wetland in southern Sweden, Wetlands Ecol. Manage., 15, 43–50, 2007.
811 Ström, L., Mastepanov, M., and Christensen, T.R.: Species specific effects of vascular plants on
812 carbon turnover and methane emissions from wetlands, Biogeochem., 75, 65–82, 2005.
813 Tai, P.D., Li, P.J., Sun, T.H., He, Y.W., Zhou, Q.X., Gong, Z.Q., Mizuochi, M., and Inamori, Y.:
814 Greenhouse gas emissions from a constructed wetland for municipal sewage treatment, J.
815 Environ. Sci. China, 14(1), 27–33, 2002.
816 Takaya, N.: Dissimilatory nitrate reduction metabolisms and their control in fungi, J. Biosci.
817 Bioeng., 94, 506–510, 2002.
818 Tanner, C.C., and Kadlec, R.H.: Influence of hydrological regime on wetland attenuation of
819 diffuse agricultural nitrate losses, Ecol. Eng., 56, 79-88, 2013.
820 Tanner, C.C., and Sukias, J.P.S.: Multi-year nutrient removal performance of three constructed
821 wetlands intercepting drainage flows from intensively grazed pastures, J. Environ. Qual.,
822 40, 620-633, 2011.
823 Tanner, C.C., Nguyen, M.L., and Sukias, J.P.S.: Nutrient removal by a constructed wetland
824 treating subsurface drainage from grazed dairy pasture, Agric. Ecosysts. Environ., 105,
825 145-162, 2005.
826 Teiter, S., and Mander, Ü.: Emission of N_2O, N_2, CH_4 and CO_2 from constructed wetlands for
827 wastewater treatment and from riparian buffer zones, Ecol. Eng., 25(5), 528–541, 2005.
828 Thayalakumaran, T., Bristow, K.L., Charlesworth, P.B., and Fass, T.: Geochemical conditions in
829 groundwater systems: Implications for the attenuation of agricultural nitrate, Agril. Water
830 Manage., 95, 103-115, 2008.
831 Toet, S., Richards, S.P., van Logtestijn, Kamp, R., Schreijer, M., and Verhoeven, J.T.A.: The
832 effect of hydraulic retention time on the performance of pollutants from sewage treatment
833 plant effluent in a surface flow constructed wetland system, Wetalnds, 25, 375-391, 2005.
834 Van der Zaag, A.C., Gordon, R.J., Burton, D.L., Jamieson, R.C., and Stratton, G.W.: Greenhouse
835 gas emissions from surface flow and subsurface flow constructed wetlands treating dairy
836 wastewater, J. Environ. Qual., 39, 460-471, 2010.

837 Verhoeven, J.T.A., and Meuleman, A.F.M.: Wetlands for wastewater treatment: Opportunities
838 and limitations, Ecol. Eng., 12, 5–12, 1999.
839 Vymazal, J., and Kröpfelova, L.: Growth of *Phragmites australis* and *Phalaris arundinacea* in
840 constructed wetlands for wastewater treatment in the Czech Republic, Ecol. Eng., 25,
841 606-621, 2005.
842 Vymazal, J.: Constructed Wetlands for Wastewater Treatment. Water, 2, 530-549, 2010.
843 Vymazal, J.: Removal of nutrients in various types of constructed wetlands, Sci. Total Environ.,
844 380, 48-65, 2007.
845 Wang, H., Huang, C., Ge, Y., zhi Wu, J., and Chang, J.: The Performance of Species Mixtures in
846 Nitrogen and Phosphorus Removal at Different Hydraulic Retention Times, Pol. J.
847 Environ. Stud., 23, 917-922, 2014.
848 Weerakoon, G.M.P.R., Jinadasa, K.B.S.N., Herath, G.B.B., Mowjood, M.I.M., and vanBruggen,
849 J.J.A.: Impact of the hydraulic loading rate on pollutants removal in tropical horizontal
850 subsurface flow constructed wetlands, Ecol. Eng., 61, 154-160, 2013.
851 Wild, U., Lenz, A., Kamp, T., Heinz, S., and Pfadenhauer, J.: Vegetation development, nutrient
852 removal and trace gas fluxes in constructed Typha wetlands, In: Mander Ü, Jenssen PD,
853 (eds.). Natural wetlands for wastewater treatment in cold climates, Adv. Ecol. Sci.,
854 Southampton, Boston, WIT Press, p. 101–26, 2002.
855 Wunderlin, P., Lehmann, M.F., Siegrist, H., Tuzson, B., Joss, A., Emmenegger, L., Mohn, J.:
856 Isotope signatures of N_2O in a mixed microbial population system: constraints on N_2O
857 producing pathways in wastewater treatment. Environ. Sci. Technol., 47, 1339–1348,
858 2013.
859 Xue, Y., Kovacic, D.A., David, M.B., Gentry, L.E., Mulvaney, R.L., and Lindau, C.W.: *In situ*
860 measurements of denitrification in constructed wetlands, J. Environ. Qual., 28(1), 263–
861 269, 1999.
862 Yan, C., Zhang, H., Li, B., Wang, D., Zhao, Y., and Zheng, Z.: Effects of influent C/N ratios on
863 CO_2 and CH_4 emissions from vertical subsurface flow constructed wetlands treating
864 synthetic municipal wastewater, J. Hazardous Materials, 203-204, 188-194, 2012.
865 Yavitt, J.B., and Knapp, A.K.: Aspects of methane flow from sediment through emergent cattail
866 (*Typha latifolia)* plants, New Phytol., 139, 495-503, 1998.
867 Zhang, J., Shao, W.S., He, M., Hu, H.Y., and Gao, B.Y.: Treatment performance and
868 enhancement of subsurface constructed wetland treating polluted river water in winter,
869 Environ. Sci. (in Chinese), 27, 1560–1564, 2006.

870 Zhao, Y., Zhang, Y., Ge, Z., Hu, C., and Zhang, H.: Effects of influent C/N ratios on wastewater
871 nutrient removal and simultaneous greenhouse gas emission from the combinations of
872 vertical subsurface flow constructed wetlands and earthworm eco-filters for treating
873 synthetic wastewater, Environ. Sci. Proc. Impacts, 16, 567-575, 2014.
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Table 1 Mineral N (mg N L^{-1}), total nitrogen (TN, mg N L^{-1}) and TN removal (%) in various CWs treating wastewater; average (± standard error, SE) are presented for TN removal; n.a – data not available

CW type	Treatment	N input (mg N L^{-1})			N output (mg N L^{-1})			N removal (%)			References
		TN	NH_4^+	NO_3^-	TN	NH_4^+	NO_3^-	TN	NH_4^+	NO_3^-	
SF_Finland	Municipal	1.4 ± 150	0.03 ± 5.8	0.3 ± 95	1.1 ± 48	0.01 ± 3.0	0.02 ± 6.7	21.4	66.7	93.3	Søvik et al., 2006
SF_Finland	Agril. runoff	66.1 ± 1.9	63.5 ± 1.3	0.7 ± 0.13	64.7 ± 1.7	61.2 ± 1.7	0.3 ± 0.09	2.1	3.6	57.1	Søvik et al., 2006
SF_Norway	Municipal	43.4 ± 3.6	41.5 ± 3.0	0.0 ± 0.0	36.7 ± 2.7	32.6 ± 1.9	0.9 ± 0.4	15.4	21.4	-800	Søvik et al., 2006
SF	Municipal	n.a	4.5	15.5	n.a	n.a	n.a	61	n.a	n.a	Song et al., 2011
SF	Domestic	n.a	40	5	n.a	n.a	n.a	97-98	n.a	n.a	Dzakpasu et al., 2011
SF	Various	n.a	39	4.4	n.a	n.a	n.a	39-48	n.a	n.a	Vymazal, 2007
SF	Municipal	n.a	36		n.a	n.a	n.a	39	n.a	n.a	Vymazal, 2010
SF	Municipal	n.a	196	<2	n.a	n.a	n.a	35	n.a	n.a	Shamir et al., 2001
SF	various	n.a	80	<1	n.a	n.a	n.a	>90	n.a	n.a	Harrington et al., 2007
SF	Municipal	n.a	0.95	1.54	n.a	n.a	n.a	45	n.a	n.a	Toet et al., 2005
SF	Dairy washout	227	n.a	n.a	n.a	n.a	n.a	40	n.a	n.a	Van der Zaag et al., 2010
All SF								31.3 ± 6.3			
HSSF_Estonia	Municipal	96.5 ± 3.0	83.9± 2.7	0.2 ± 0.02	46.2 ±1.5	36.2 ± 1.4	5.9 ± 0.65	52.1	56.9	-2850	Søvik et al., 2006
HSSF_Norway	Municipal	53.4 ± 4.3	38.4 ± 7.7	14.1 ± 7.5	45.0 ± 4.1	43.1 ± 4.7	1.0 ± 0.8	15.7	-12.2	92.9	Søvik et al., 2006
HSSF	Dairy washout	306 ±101*	n.a	n.a	177 ± 58*	n.a	n.a	42.2	n.a	n.a	Van der Zaag et al., 2010
HSSF	Domestic	n.a	74.9	3.9	n.a	n.a	n.a	29	n.a	n.a	O'Luanaigh et al., 2010
HSSF	Domestic	87						46-48			Mander et al., 2008
HSSF	Dairy washout	227						28			Van der Zaag et al., 2010
HSSF	Milk parlour	112	22	n.a	24	11	n.a	78	50	n.a	Kato et al., 2006
HSSF	Agriculture	67	40	0.85	27	11	1.1	47	39	-29	Vymazal and Kröpfelova, 2009
HSSF	Industry	124	65	8.5	103	31	7.4	20	20	8	Vymazal and Kröpfelova, 2009
HSSF	Landfill	157	149	1.5	147	98	1.3	30	33	31	Vymazal and Kröpfelova, 2009
HSSF	Municipal	43	24	2	24	14	1.2	40	30	33	Vymazal and Kröpfelova, 2009
All HSSF								40.4 ± 4.4			
VSSF_Estonia	Municipal	50.9 ± 9.2	35.7 ± 6.2	1.1 ± 0.32	43.1 ± 7.6	31.7 ± 5.5	1.7 ± 0.84	15.3	11.2	-54.5	Søvik et al., 2006
VSSF_Norway	Municipal	52.6 ± 5.2	49.6 ± 4.0	0.0 ± 0.0	47.8 ± 6.9	21.4 ± 6.9	25.5 ± 1.3	9.1	56.9	-25400	Søvik et al., 2006
VSSF	Municipal	41.0 ± 0.5	n.a	n.a	20.7 ± 0.8	n.a	n.a	49.3 ± 1.8	n.a	n.a	Yan et al., 2012
VSSF	Municipal	46 ± 13	n.a	n.a	n.a	n.a	n.a	74 ± 3	n.a	n.a	Zhao et al., 2014
All VSSF								37.0 ± 10.9			

SF- surface flow; HSSF – horizontal subsurface flow; VSSF- vertical subsurface flow; *mg N m^{-2} h^{-1}

Table 2 Total organic C (TOC) removal (%) in various CWs treating wastewater; average (± standard error, SE) are presented for TOC removal; n.a - data not available

CWs type	Treatment	C input (TOC; mg C L^{-1})	C outflow (TOC; mg C L^{-1})	TOC Removal (%)	References
SF_Finland	Municipal	13.0 ± 0.3	14.0 ± 0.5	-7.7	Søvik et al., 2006
SF_Finland	Agril runoff	25.0 ± 3.4	20.0 ± 3.4	20.0	Søvik et al., 2006
SF_Norway	Municipal	26.7 ± 2.9	17.1 ± 1.8	36.0	Søvik et al., 2006
SF	Dairy wash out	186†	136†	27	Van der Zaag et al., 2010
All SF				18.8 ± 9.4	
HSSF	Domestic	150*	n.a	n.a	Garcia et al., 2007
HSSF	Dairy wash out	186†	107.9†	42	Van der Zaag et al., 2010
HSSF_Estonia	Municipal	62.8 ± 16.6†	41.0 ± 11.3 †	34.7	Søvik et al., 2006
HSSF_Norway	Municipal	40.5 ± 11.3	15.0 ± 2.4	63.0	Søvik et al., 2006
All HSSF				46.6 ± 7.3	
VSSF_Estonia	Municipal	132.2 ± 32.2†	62.8 ± 16.6 †	52.5	Søvik et al., 2006
VSSF_Norway	Municipal	40.5 ± 11.3	15.0 ± 2.4	63.0	Søvik et al., 2006
VSSF	Municipal	106 ± 35	74 ± 21	26 ± 4.6	Yan et al., 2012
VSSF	Municipal	249 ± 49	n.a	83 ± 1.0	Zhao et al., 2014
All VSSF				56.2 ± 9.5	

SF- surface flow; HSSF – horizontal subsurface flow; VSSF- vertical subsurface flow; †BOD; *mg m^{-2} h^{-1}

Table 3 Nitrous oxide (N_2O) emissions (mg N m^{-2} d^{-1}); N_2 emissions (mg N m^{-2} d^{-1}) and N_2O emission factor (N_2O/TN input*100) in various type of CWs; mean (± SE) was presented for N_2O emission factor; n.a - data not available

CW type	Treatment	Denitrification		N_2O-N/TN (%)	N_2-N/TN (%)	References
		N_2O emissions (mg N m^{-2} d^{-1})	N_2 emissions (mg N m^{-2} d^{-1})			
HSF	Agril. tile drainage	0.01-0.12	n.a	0.19-1.4	n.a	Xue et al., 1999
HSF	Treated municipal	2.0±3.3	n.a	0.02-0.27	n.a	Johansson et al., 2003
HSF	Agril. drainage	-0.2-1.9	n.a	-0.14-0.52	n.a	Wild et al., 2002
HSF	Dairy wash out	16.8±7.0	n.a	0.33±0.12	n.a	Van der Zaag et al., 2010
HSF_Finland	Municipal	0.01±0.01	n.a	1.6±1.3	n.a	Søvik et al., 2006
HSF_Finland	Agril. runoff	0.40±0.25	n.a	0.37±0.18	n.a	Søvik et al., 2006
HSF_Norway	Municipal	4.0±1.6	n.a	1.5±4.4	n.a	Søvik et al., 2006
All SF		2.78±1.72		0.61±0.21		
HSSF	Domestic	0.2-17.0	n.a	0.06-3.8	n.a	Mander et al., 2005
HSSF_Estonia	Municipal	7.1±1.2	n.a	0.05±0.31	n.a	Søvik et al., 2006
HSSF_Norway	Municipal	6.9±4.3	n.a	0.24±0.53	n.a	Søvik et al., 2006
HSSF	Domestic	1.3-1.4	160-170	0.37-0.60	15.2-22.7	Mander et al., 2008
HSSF	Domestic	0.003-0.001	0.01-5.42	n.a	n.a	Teiter and Mander, 2005
HSSF	Domestic	0.13	n.a	0.008	n.a	Fey et al., 1999
HSSF	Dairy wash out	9.5±1.5	n.a	0.18±0.12	n.a	Van der Zaag et al., 2010
HSSF	Domestic	0.17	n.a	0.23	n.a	Liu et al., 2009
VSSF	Domestic	0.17	n.a	0.01		Mander et al., 2011
All HSSF		4.23±1.87		0.62±0.38		
VSSF	Domestic	0.001-0.002	0.01-5.0	n.a	n.a	Teiter and Mander, 2005
VSSF	Domestic	4.6	150	0.45-0.50	n.a	Mander et al., 2008
VSSF	Domestic	11.0	n.a	0.29	n.a	Mander et al., 2005
VSSF	Domestic	1.44	n.a	0.03		Mander et al., 2011
VSSF	Domestic	0.005	n.a	0.09	n.a	Gui et al., 2007
VSSF	Domestic	0.003	n.a	0.04	n.a	Liu et al., 2009
VSSF_Estonia	Municipal	15±3.9	n.a	04.3±0.95	n.a	Søvik et al., 2006
VSSF_Norway	Municipal	960±40	n.a	1.4±0.72	n.a	Søvik et al., 2006
All VSSF		123.8±106		1.01±0.48		

SF- surface flow; HSSF – horizontal subsurface flow; VSSF- vertical subsurface flow

Table 4 Carbon dioxide (CO_2, mg C m^{-2} d^{-1}), CH_4 (mg C m^{-2} d^{-1}) and CH_4 emission factor (CH_4-C/TOC input*100) in various type of CWs; mean (± SE) was presented for CH_4 emission factor; n.a - data not available

CWs type	Treatment	CO_2 emissions (mg C m^{-2} d^{-1})	CH_4 emissions (mg C m^{-2} d^{-1})	CH_4/ TC (%)	References
SF	Municipal	n.a	5.4	n.a	Tai et al., 2002
SF	Domestic	0.19	n.a	26	Gui et al., 2007
SF	Domestic	1.13	n.a	16	Liu et al., 2009
SF	Agril. drainage	n.a	0.88	31	Wild et al., 2002
SF	Dairy wash out	4250±550	223±35	9.45	Van der Zaag et al., 2010
SF_Finland	Municipal	1200±420	29±6.4	19±4.3	Søvik et al., 2006
SF_Finland	Agril runoff	3200±560	350±180	11±5.5	Søvik et al., 2006
SF_Norway	Municipal	1400±250	72±28	4.8±2.2	Søvik et al., 2006
All SF		1675±703	113±58	16.8±3.8	
HSSF	Domestic	n.a	1.7-528	n.a	Mander et al., 2005a
HSSF	Domestic	2.54-5.83	0.03-0.40	n.a	Teiter and Mander, 2005
HSSF	Domestic	5.33	0.001	0.03	Garcia et al., 2007
HSSF	Domestic	n.a	0.03	4.3	Gui et al., 2007
HSSF	Domestic	n.a	0.29	4.0	Liu et al., 2009
HSSF	Dairy wash out	3475±375	118±9.0	4.4	Van der Zaag et al., 2010
HSSF	Domestic	0.6-1.7	1.4- 4.1	0.12-0.23	Søvik et al., 2006
HSSF	Domestic	600	0.48	0.02	Mander et al., 2011
HSSF_Estonia	Municipal	3800±210	340±240	n.a	Søvik et al., 2006
HSSF_Norway	Municipal	790±170	130±43	9.5±3.3	Søvik et al., 2006
All HSSF		1010±672	112±74	3.23±1.4	
VSSF	Domestic	5.83-12.13	0.60-5.70		Teiter and Mander, 2005
VSSF	Domestic	n.a	16.4	n.a	Mander et al., 2005
VSSF	Domestic	n.a	0.013	1.68	Gui et al., 2007
VSSF	Domestic	n.a	0.13	1.73	Liu et al., 2009
VSSF	Municipal	2662 ± 175	33.5 ± 3.2	n.a	Mander et al., 2008
VSSF	Domestic	1080	3.36	0.05	Mander et al., 2011
VSSF_Estonia	Municipal	8400±2100	110±35	n.a	Søvik et al., 2006
VSSF_Norway	Municipal	22000±5000	140±160	0.39±0.27	Søvik et al., 2006
All VSSF		6616 ± 3779	42.9 ± 23.7	1.27±0.31	

SF- surface flow; HSSF – horizontal subsurface flow; VSSF- vertical subsurface flow

Table 5 Nitrous oxide (N_2O, mg N m^{-2} d^{-1}), CO_2 and CH_4 emissions (mg C m^{-2} d^{-1}) in various type of CWs under different plant types; n.a - data not available

CW type	Wastewater type	Plant type	N_2O (mg N m^{-2} d^{-1})	CH_4 (mg C m^{-2} d^{-1})	CO_2 (mg m^{-2} d^{-1})	Reference
HSF	Secondary treated municipal	No plant	3.79±2.64	163±209		Johansson et al., 2003; Johansson et al., 2004
		Typha lotifolia	2.64±4.09	109±185	n.a	
		Phalaris arundinacea	3.79±3.44	212±151	n.a	
		Glyceria maxima	0.76±1.01	112±178	n.a	
		Lemna minor	1.45±1.18	450±182	n.a	
		Spirogyra sp.	0.98±1.25	107±135	n.a	
HSF	Sewage Treatment water	No plant	-0.26±2.53	-4.76±61.8	4.32±0.73	Ström et al., 2007
		Typha atifolia	4.94±2.00	225±47.7	25.3±4.08	
		Phragmites australis	7.80±2.53	333±76.6	25.1±4.74	
		Juncus effusus	3.87±1.86	489±46.3	26.1±3.00	
HSSF	Domestic	No plant	0.04±0.02	87±6.3	80±6.3	Maltais-Landry et al., 2009
		Phragmites	0.06±0.03	50±7.5	200±35	
		Typha	0.03±0.01	28±3.0	235±32	
		Phalaris	0.01±0.01	45±6.0	195±31	
VSSF	Municipal	*Phragmites australis*	15±3.9	110±35	8400±2100	Søvik et al., 2006
VSSF	Municipal	*Phragmites australis*	264	384		Mander et al., 2005

SF- surface flow; HSSF – horizontal subsurface flow; VSSF- vertical subsurface flow

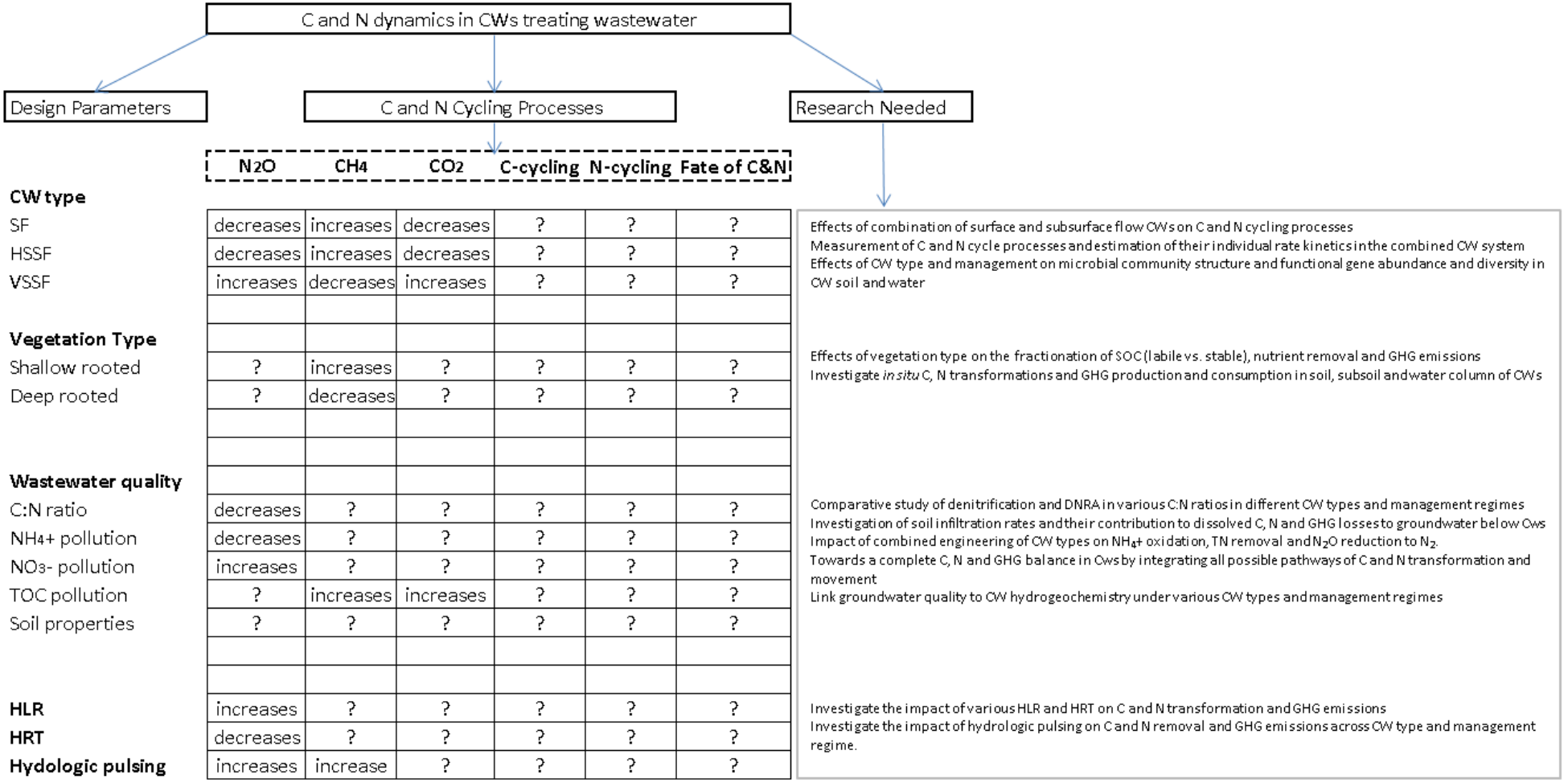

Figure 1 Conceptual model showing the current state of knowledge of C and N dynamics in constructed wetlands treating wastewater and the specific experimental work that needs to be undertaken in the future; SF- surface flow; HSSF – horizontal subsurface flow; VSSF- vertical subsurface flow; HLR – hydraulic loading rate; HTR – hydraulic retention time; ? – not known or very little known