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May 22, 2020

Steven Bouillon, Dr
Associate Editor
Biogeosiences

Dear Dr. Bouillon,

We would like to thank you for your positive decision for our manuscript.

We have modified the manuscript according to your suggestions. Please find a point-by-point response at the bottom of this letter noted in bold with the editor's original comments in blue.

Thank you for considering our manuscript for your journal.

Sincerely,

Dr. Xuefei Li

Comments to the Author:
“Dear dr Li,
Thank you for your detailed reply to both referee reports, and for your revised version. Review report #1 was favorable to start with, and I feel their suggestions and points of concern were well addressed. For the 2nd review report, most of the concerns have been incorporated to the extent possible, e.g. the issue of gap filling is addressed in the revised version with the additional information provided in the supplement; this is transparent and the reader can make their own appreciation of the merits and limitations. I do feel that some more notes on the gas transfer velocity parameterization. I understand your rationale to stick with the Cole Caraco model, but feel that a summary of the information you provide in your author replies would be worth including in the manuscript itself; this will strengthen the rationale to use their model and at the same time provide some more background to the reader what the limitations are and why these were not straightforward to address with your dataset”

**We now explained in more details on the rationale of using Cole & Caraco model in the manuscript (Line 225-239)**

“Two small additional notes from my side: -the total phosphorus data: you mention these were "calculated based on turbidity data" and refer to Valkama et al. (2017). I would suggest to replace ’calculated’ by ’estimated”’

**We changed ’calculated’ by ’estimated’ (Line 140).**

“I assume the data you use are in fact the same as those of the latter publication, and the same holds for the nitrate data ? If that is the case: avoid confusion and make it clear that these nitrate and TP data were published earlier and were taken from Valkama et al. (2017). ”

**While the method of estimating phosphorus using turbidity data has been explained in Valkama et al.(2017), the data was not taken from that paper. But these data was published previously in Wahlroos et al. (2015). It has been now clarified in the manuscript (Lines 138-140).**

“Figure 7 caption: subscript in ’FCH4”’

**’FCH4’ has been changed to $F_{CH4}$ in Figure 7 and its captions.**

# Carbon dioxide and methane fluxes from different surface types in a created urban wetland

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**Abstract.** Many wetlands have been drained due to urbanization, agriculture, forestry or other purposes, which has resulted in losing their ecosystem services. To protect receiving waters and to achieve services such as flood control and stormwater quality mitigation, new wetlands are created in urbanized areas. However, our knowledge of greenhouse gas exchange in newly created wetlands in urban areas is currently limited. In this paper we present measurements carried out at a created urban wetland in Southern Finland in the boreal climate.

We conducted measurements of ecosystem $CO_2$ flux (NEE) and $CH_4$ flux ($F_{CH4}$) at the ~~constructed~~ created stormwater wetland Gateway in Nummela, Vihti, Southern Finland using eddy covariance (EC) technique. The measurements were commenced the fourth year after construction and lasted for one full year and two subsequent growing seasons. Besides ecosystem scale fluxes measured by the EC tower, the diffusive $CO_2$ and $CH_4$ fluxes from the open-water ~~area~~areas ($F_w$_$CO_2$ and $F_w$_$CH_4$, respectively) were modelled based on measurements of $CO_2$ and $CH_4$ concentration in the water. Fluxes from the vegetated areas were estimated by applying a simple mixing model using the above-mentioned fluxes and the footprint-weighted fractional area. The half-hourly footprint-weighted contribution of diffusive fluxes from open water ranged from 0 to 25.5 % in ~~year~~ 2013.

The annual NEE of the studied wetland was 8.0 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ with the 95 % confidence interval between -18.9 and 34.9 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ and $F_{CH4}$ was 3.9 g C-$CH_4$ $m^{-2}$ $yr^{-1}$ with the 95 % confidence interval between 3.75 and 4.07 g C-$CH_4$ $m^{-2}$ $yr^{-1}$. The ecosystem sequestered $CO_2$ during summer months (June-August), while the rest of the year it was a $CO_2$ source. $CH_4$ displayed strong seasonal dynamics, higher in summer and lower in winter, with a sporadic emission episode in the end of May 2013. Both $CH_4$ and $CO_2$ fluxes, especially those obtained from vegetated areas, exhibited strong diurnal cycles during summer with synchronized peaks around noon. The annual $F_w$_$CO_2$ was 297.5 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ and $F_w$_$CH_4$ was 1.73 g C-$CH_4$ $m^{-2}$ $yr^{-1}$. The peak diffusive $CH_4$ flux was 137.6 nmol C-$CH_4$ $m^{-2}$ $s^{-1}$, which was synchronized with the $F_{CH4}$.

Overall, during the monitored time period, the established stormwater wetland had a climate warming effect with 0.263 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$ of which 89 % was contributed by $CH_4$. The radiative forcing of the open-water areas exceeded that of the~~e~~ vegetation areas (1.194 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$ and 0.111 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$, respectively), which implies that, when considering solely the climate impact of a created wetland over a 100-year horizon, it would be more beneficial to design and establish wetlands with large patches of emergent vegetation, and to limit the areas of open-water to the minimum necessitated by other desired ecosystem services.

## 1 Introduction

Wetlands provide many beneficial ecosystem services such as flood control and water quality mitigation, natural habitat for flora and fauna and recreational opportunities (Mitsch and Gosselink, 2015). Many wetlands have been drained globally for agriculture, forestry and other purposes including urbanization at the cost of losing wetland ecosystem services (Vasander et al., 2003). Migration from rural area to cities will increase in even greater numbers in the near future, and the United Nations report (United Nations, 2016)~~UN 2016 report~~ has predicted that 75 % of the world population will be living in cities by 2030. There is an urgent need for more sustainable urbanism and one effective measure is to create functional and connected wetland networks in cities (Lucas et al., 2015~~;~~; Mungasavalli and Viraraghavan, 2006).

Wetlands can take up carbon dioxide ($CO_2$) through emergent and submerged vegetation but they are also important sources of methane ($CH_4$), a greenhouse gas more potent than $CO_2$ when considered over a 100-year horizon (Stocker et

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al., 2014). The exchange of greenhouse gases (GHG) such as $CO_2$ and $CH_4$ between atmosphere and ecosystem have direct influence on the atmospheric concentration of these gases, thus besides the ecosystem services that wetlands provide, the GHG budget of constructed wetlands should be accounted for according to international agreements such as the ~~Kyoto~~ Paris Agreement~~protocol~~.

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Reports on boreal wetlands, such as peatlands, have shown that large carbon storage remains in the soil due to anaerobic conditions limiting microbial decomposition, and thus offering a global cooling effect (Frolking et al., 2006). However, in newly constructed urban wetlands on mineral soil the gas exchange may be very different from natural wetlands: 1) T~~t~~he cooling effect of a wetland may be reduced or it becomes a source of carbon due to the early successional stage of the wetland. When an urban wetland is newly created by ~~re~~wetting ~~the~~ a landscape, it takes time for ~~the~~ vegetation to establish itself in the new environment. The low coverage of vegetation at the initial phase of wetland establishment can lead to low $CO_2$ sequestration on an ecosystem scale.~~;~~ 2) W~~w~~etlands in close proximity to urban centers receive a significant amount of nutrients and dissolved organic carbon from runoff (Lu et al., 2009; Vohla et al., 2007; Valkama et al., 2017)~~ and 3) urban wetlands exhibit high spatial heterogeneity and hydrology where different processes of the production and transportation of GHG are involved~~. At the areas with emergent vegetation, $CO_2$ is absorbed by photosynthetic activity during daytime and growing season and is released through respiration~~al processes~~. At open-water surfaces, the net production of $CO_2$ is a result of photosynthesis by algae, cyanobacteria as well as submerged aquatic plants, respiration of organic carbon~~ and oxidation of $CH_4$ produced in the water~~. When the $CO_2$ concentration in the water exceeds atmospheric equilibrium, the surface becomes a source of $CO_2$. $CH_4$ can be produced through anaerobic metabolism in wetland soil and can be transported to the atmosphere by plant-mediated pathway through aerenchyma, sediment ebullition and diffusive fluxes at the water-atmosphere interface. In open water, the transport is dominated by diffusion whereas in the vegetated areas the plant-mediated transport is most prominent.

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Urban wetlands have received extensive attention globally and their societal and economical importance have been evaluated (Salminen et al., 2013), whereas their climate impact is still largely overlooked except for only a few studies (e.g. Morin et al., 2014a; Morin et al., 2014b). The thus far~~only~~ review of GHG emission in constructed wetlands for wastewater treatment reported that the average $CO_2$ emission was 92.3 mg $CO_2$-C $m^{-2}$ $h^{-1}$ and that the $CH_4$ emission ranged from 1.6 to 27 mg $CH_4$-C $m^{-2}$ $h^{-1}$ from free water surface (Mander et al., 2014). All of the studies were based on static chamber measurements during a short period so that the annual carbon balance of the ecosystem could not be assessed. In contrast to static chamber measurements, the eddy covariance (EC) method provides continuous measurements of GHG exchange at ecosystem scale, presenting the net result of fluxes as exchange in different source areas contributing simultaneously within the footprint extent (Baldocchi, 2003). It is worth noticing that one of the assumptions of the EC method is surface homogeneity, yet in many study sites the ~~situations are~~landscape setting is far from ideal. The change of source area due to changes in wind provides difficulties in estimating GHG emissions in spatially heterogeneous sites especially in short-term flux measurements (Baldocchi et al., 2012). Therefore, for heterogeneous sites such as urban wetlands, accurate footprint modelling and surface area map at high spatial resolution are important in identifying the source area, and a land-surface specific analysis is vital to reveal the diel pattern, sink/source strength of the wetland.

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The objective of this study is to investigate how $CO_2$ and $CH_4$ surface-atmosphere exchange vary with seasonality and spatial heterogeneity and what the annual radiative forcing of these gases are in a ~~constructed~~ created urban wetland ~~near~~ ~~attown~~ the Nummela suburb, Municipality of Vihti, Southern Finland. The studied Gateway wetland was designed and implemented to serve the purposes of stormwater quality treatment, creating an urban park, as well as supporting biodiversity. Besides taking advantage of ecosystem-scale EC measurements, we also parse the variability of gas exchange induced by surface heterogeneity (the open water and the vegetated areas) using diffusional flux modeling and footprint modelling overlapped on a high-resolution surface map. To illustrate how the urban wetland functions as a source or a sink of GHG equivalents, we calculate separately the sustained global warming potential (SGWP) of $CO_2$ and $CH_4$ over a hundred-year horizon in each surface type.

## 2 Materials and Methods

### 2.1 Site description

Our study site is a created stormwater wetland Gateway, located by an eutrophicated Lake Enäjärvi in the District of Nummela, Municipality of Vihti, Southern Finland (60.3272°N, 24.3369°E). Southern Finland experiences a climate with a 30-year mean air temperature of 4.6 °C and an annual precipitation rate of 627 mm in the period of year 1981-2010 (Pirinen et al., 2012).

The wetland was constructed in 2010 at the mouth of a 550 hectare largely urbanized (35 % impervious) watershed of Stream Kilsoi. It was excavated over six weeks in early winter 2010 on an abandoned agricultural field growing meadow vegetation. All of the old drainage ditches were blocked as amphibian habitats, which also ensured only one inlet route receiving water from Stream Kilsoi and one outlet route discharging water to the nearby Lake Enäjärvi. Lake Enäjärvi is an eutrophicated lake. The internal phosphorus load from human activities and the run-off from its catchments have resulted in regular cyanobacterial blooms and fish kills in the lake (Varis et al., 1989; Salonen et al., 2000).

The wetland park has a total area of 7 hectares within which - during mean water flow conditions - a 0.5 hectare inundated wetland is located. This stormwater treatment wetland consists of an inlet stilling pond, a meandering shallow water area with three habitat islands, and an outlet pond. The average water depth in the ponds is 1.5 m; within emergent vegetation patches water depth ranges between 0.3 and 0.5 m. There are also submerged macrophytes in the open water as the water is shallow, thus in the paper we refer the "vegetated area" to the area with emergent vegetation and "open water" to the area covered by water in the absence of emergent macrophytes. The outlet bottom dam sets a low water level (WL) to 50.04 m above the Baltic Sea level (N60+ coordinate system). Herbaceous vegetation has been allowed to fully self-establish after the construction of the wetland. Annual monitoring of vegetation carried out in summers 2010, 2011 and 2012 indicated rapid self-establishment of vegetation which was rich in taxa and dominated by native species (Wahlroos et al., 2015). At frequently-inundated area (elevation levels of 50-50.35 m), vegetation was arranged in dense patches with different dominating wetland plant species: *Typha latifolia* L., *Iris pseudacorus* L., *Carex* spp. or *Juncus effuses* L. At the major less-frequently inundated area (elevation levels of 50.35-50.45m), the wet meadow species *Filipendula ulmaria* L. (Maxim.), *Lysimachia vulgaris* L., and *Lythrum salicaria* L. with the three species co-existing at 1:1:1 ratio formed the plant community. Drier areas (elevation levels of 50.45-50.60 m) were mostly colonized by dry meadow species such as *Poa* spp. and *Calamagrostis* spp., including patches dominated by *Cirsium* species (Fig. ~~S~~1). Note that the area with water level lower than 49.5 m is defined as the open water area while the rest is defined as the vegetated area in this study.

### 2.2 Water and micrometeorological measurements

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Water monitoring stations were set up at the inlet (60.3283° N, 24.3356° E) and at the outlet (60.3281° N, 24.3377° E) of the wetland. During the 2012-2013 and 2013-2014 monitoring periods, water temperature as well as water turbidity, oxygen concentration, conductivity and pH were measured at the inlet and outlet monitoring station with the YSI-6~~0~~600 series multiparameter sonde (YSI Inc., Yellow Springs, OH, USA). Measurements were conducted continuously with a 10-minute interval. Water level at the outlet was measured continuously with a pressure gauge (STS sensor, Sensor Technik Sirnach AG, Switzerland). At the outlet monitoring station, the concentration of dissolved carbon dioxide ([$CO_2$]) and dissolved methane ([$CH_4$]) were measured with Contros HydroC™ $CO_2$ and HydroC™ $CH_4$ sensors (CONTROS Systems & Solutions GmbH, Germany). In 2014, the same sensors were also installed at the inlet monitoring station to measure [$CO_2$] and [$CH_4$]. Dissolved $CO_2$ and $CH_4$ molecules diffuse from water column into the detection chamber through a thin-film composite membrane where the concentration of $CO_2$ and $CH_4$ is determined by means of IR absorption spectrometry and Tunable Diode Laser Absorption Spectroscopy, respectively. $NO_3$-N and total phosphorus (TP) ~~data~~ data have been previously published in Wahlroos et al. (2015). Briefly, $NO_3$-N was measured with Scan sensors (Scan gmbh, Austria) and TP was ~~calculated~~estimated based on turbidity data~~ which was me~~ measasured at 10-min intervals.~~ (Valkama et al., 2017).~~

Local weather conditions were recorded with a Vaisala WXT weather transmitter (WXT520, Vaisala Oyj, Finland) at the inlet monitoring station. Rainfall, wind speed and direction, temperature and relative humidity were recorded continuously at a 10-minute interval. Photosynthetic photon flux density (PPFD) was measured with a PQS1 PAR quantum sensor (Kipp & Zonen, the Netherlands). Due to instrument failure we obtained PPFD data only from 26 Jan to 7 April and from 22 July to 29 Dec 2013. The gaps were filled with PPFD data from another meteorological station nearby (60°38' N, 23°58' E) in Lettosuo, Finland. The prevailing wind directions were southwest and northeast, and the average of half-hourly average wind speed was 1.13 m s$^{-1}$ from January to December 2013 with higher wind speed in winter than in summer. The average daily air temperature was 5.9 ℃ with the minimum and maximum daily temperatures of -24.4 ℃ and 23.3 ℃ in ~~year 2013~~2013. During the winter 2012-2013, there was ice coverage from the beginning of December 2012 to the end of March 2013. In contrast, winter was mild and warm in 2014 and there was practically no snow cover during a winter period (December 2013-March 2014).

### 2.3 Greenhouse gas measurements by EC tower and gap-filling

To understand the whole-ecosystem exchange of $CO_2$ and $CH_4$ in the wetland, a 2.9 m eddy covariance tower was established in the autumn of 2012 on the southern side of the wetland. The operational period of the EC tower was the entire calendar year of 2013 (from 1 January to 31 December 2013) and the peak growing season in 2014 (from 1 June to 31 August 2014). The EC set-up included a 3D-sonic anemometer (uSonic-3, Metek, Elmshorn, Germany) to measure the three wind speed components and sonic temperature, a gas analyser (LI-7200, Li-Cor Inc., Lincoln, Nebraska, USA) which measures $CO_2$ and $H_2O$ mixing ratio and a TDL gas analyser (TGA100A, Campbell Scientific Inc., USA ) to measure $CH_4$ mixing ratio. Data from the analyzers were collected on a computer at the frequency of 10 Hz. The post-processing of the EC flux data has been done with EddyUH post-processing software (Mammarella et al., 2016). The fluxes were calculated as 30-min covariances between the vertical wind velocity and the gas mixing ratio using block averaging. The raw data was despiked according to standard methods (Vickers and Mahrt, 1997). Coordinate rotations were conducted by performing a two-step rotation to make the x-axis along the mean wind direction and the mean vertical wind velocity zero within each 30-min block. The time lag between the anemometer and gas analyzer signals, resulting from the transport through the inlet tube, were determined for each 30-min interval by maximizing the cross-correlation

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function between vertical wind speed and the scalar ($CO_2$ and $CH_4$). The fluxes were corrected for high-frequency loss due to the limited frequency response of the EC system and low-frequency loss due to the limited averaging time period used for calculating the fluxes. Theoretically and experimentally determined co-spectral transfer functions at low and high frequency were used in the correction (Mammarella et al., 2009).

After calculating the fluxes, data collected from periods when the sonic anemometer showed sign of freezing (mean temperature < 0.5 °C and standard deviation of temperature > 1.5 °C) were discarded. The data collected during weak turbulence with friction velocity below 0.1 m $s^{-1}$ have been removed. The measurement points with flux stationarity greater than 1 were omitted to ensure the quality of the co-variances. Fluxes were further filtered according to the wind direction. Since the patchy forest to the southeast of the EC tower (from 100° to 200°) and the highway to the west (from 200° to 280°) could potentially lead to flow distortion and additional source of $CO_2$ and $CH_4$, only fluxes from 280° to 100° were accepted for further analysis. The percentage of 30-min fluxes excluded from this analysis was 72 % for $CO_2$ and 73 % for $CH_4$ in 2013, whereas in 2014 the percentage for data exclusion was 54 % for $CO_2$ and 68 % for $CH_4$.

We used an artificial neural network (ANN) technique to gap-fill half-hourly flux data using meteorological variables (Moffat et al., 2007; Papale et al., 2006). Those variables included radiation, air temperature, water temperature, water level, wind speed, relative humidity, time of the day, season, and dissolved $CO_2$ and $CH_4$ concentration in the water. We tested the model performance with different ANN architectures, starting from the architecture with the most complexity, then reduced the variables to find the simplest ANN architecture with good performance (more than 5 % loss in model accuracy with additional variable reduction). For $CO_2$, water level and wind speed were found to have trivial contributions to the ANN model thus they were removed from the model input, while for $CH_4$, only wind speed was removed for the same reason. We found that dissolved gas concentration greatly improved the model prediction as they captured the variation of diffusive fluxes from the water (Fig. S1~~S2~~). Ancillary meteorological variables in general had good data coverage and short gaps (up to several hours) were gap-filled by linear interpolation. The only exception was dissolved gas concentration, which had a long measurement breakage in ~~year 2013~~2013 (day of year 214-254). Fluxes were therefore gap-filled with two separate ANNs, one with dissolved gas concentration and one without. During the ~~above mentioned~~above-mentioned period with long gaps, the ANN modeled without dissolved gas concentration were used to gap-fill.

Levenberg-Marquardt algorithm was used in the learning process of ANN. The optimized number of neurons in the hidden layer were determined by training the network 100 times with varying number of neurons (from 3 to 15), and 10 neurons was considered to be sufficient after evaluating the performance of the network using root-mean-square-error (RMSE) (data not shown). The entire dataset was divided into three parts, 2/3 of the data was used to train the networks, 1/6 for testing the networks and the remaining 1/6 was used for validating the networks. Since the training of the networks can be biased towards periods with greater data coverage (e.g. daytime conditions), the environmental variables were first divided into five natural clusters using a k-mean clustering algorithm in Matlab (MATLAB 2015a, The MathWorks, Inc., Natick, Massachusetts, United States), and then the data used for training, testing and validation was proportionally extracted from each cluster. After each data extraction, the network was reinitialized for 10 times to avoid local minima and the initialization with the lowest RMSE was selected and the resulting network was saved. We repeated the whole process of data extraction and initialization for 20 times, and we used the median of these 20 predictions to gap-fill the missing flux values. The uncertainty of the ANN gap-filling procedure was presented using a 95 % confidence interval of the 20 ANN predictions.

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In order to be confident in our gap-filling results, we also applied alternative gap-filling methods to EC fluxes using parameterization based on biological principles (see Supplement Material). The results on annual cumulative fluxes were not significantly different from the ones gap-filled using ANN, thus we only report the results from ANN in the following text.

~~The gap-filled net ecosystem exchange (NEE) can be further partitioned into two components gross ecosystem production (GEP) and ecosystem respiration ($R_{eco}$) according to the following equation:~~

$$\cancel{\mathrm{NEE} = \mathrm{GEP} + R_{eco},} \qquad (1)$$

~~where positive $R_{eco}$ represents a net carbon flux from the ecosystem to the atmosphere and negative GEP represents a net carbon input from the atmosphere to the ecosystem. Thus the negative NEE indicates that the ecosystem is a carbon sink and the positive NEE means the ecosystem is a carbon source. $R_{eco}$ was estimated using a model describing the temperature dependence of $R_{eco}$~~

$$\cancel{R_{eco} = R_{10} e^{\left[E\left(\frac{1}{T_0} - \frac{1}{T_{air}+T_1}\right)\right]}} \qquad (2)$$

~~where $E$ =346.37 K is an activation-energy-related physiological parameter, $T_{air}$ is the air temperature, $T_0$ = 56.02 K and $T_1$ = 227.13 K (Lloyd and Taylor, 1994;Aurela et al., 2009). $R_0$ is the rate of ecosystem respiration at 10 °C. We first fitted the model with nighttime NEE (which represents the nighttime ecosystem respiration since photosynthesis is assumed to be zero at night) and determined $E$. We then calculated $R_0$ for each of the bi-weekly periods (Aurela et al., 2009). This model was then extrapolated to daytime periods so that $R_{eco}$ in the daytime was obtained. GEP was estimated as the difference between NEE and $R_{eco}$.~~

**2.4 Diffusive gas exchange**

We calculated diffusive gas exchange $F$ from open water according to the boundary layer model

$$F = k(c_{aq} - c_{eq}), \qquad (3)$$

where $k$ is the gas transfer velocity (cm h$^{-1}$), $c_{aq}$ is the gas concentration in surface water (mol m$^{-3}$) and $c_{eq}$ is the gas concentration that surface water would have when it reaches equilibrium with the air (mol m$^{-3}$). $c_{aq}$ and $c_{eq}$ can be obtained according to the solubility of the gas

$$c_{aq} = 10^{-3} k_H p \chi_{water} \qquad (4)$$

$$c_{eq} = 10^{-3} k_H p \chi_{air} \qquad (5)$$

where $k_H$ is Henry's law constant for the respective gas (mol L$^{-1}$ atm$^{-1}$), $p$ is air pressure (atm), $\chi_{water}$ is the gas mixing ratio in surface water (ppm) and $\chi_{air}$ is the gas mixing ratio in the air (ppm). In this study, $\chi_{water}$ was obtained from the outlet monitoring station as it was located most of the time in the flux footprint area and it had longer data coverage than from the inlet monitoring station. The gas transfer velocity $k$ can be calculated as the formula below (Cole and Caraco, 1998):

$$k = (2.07 + 0.215 U_{10}^{1.7}) \left(\frac{S_c}{600}\right)^{-0.5}, \qquad (6)$$

where $U_{10}$ is the horizontal wind speed extrapolated to 10 m using the theoretical log wind profile equation (m s$^{-1}$, approximately $U_{10}$ = 1.15$U$ where $U$ is the measured wind speed at 2.9 m height in the study site) and $S_c$ is the

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temperature-dependent Schmidt number of the respective gas. When gas concentration measurement was not available, linear interpolation was applied to obtain monthly and annual diffusive GHG fluxes from the open water.

Although the above-mentioned Cole-Caraco (CC) method is the most simple and most often used model for gas transfer velocity, the limitation of CC method is that it considers wind as the sole factor to cause the water turbulence and to drive the gas exchange. More complicated models were suggested to include the effect of buoyancy flux driven turbulence (Heiskanen et al., 2014; Tedford et al., 2014), which could not be applied in the current study as we did not have the measurements of some of the required model inputs, e.g. the net shortwave and longwave radiation from the water body.~~More complicated models were suggested to include the effect of buoyancy flux driven turbulence (Heiskanen et al., 2014; Tedford et al., 2014). It is important to note that we should apply with caution the model parameterization concluded from other sites with different meteorological and environmental condition. In the present study, the open water is connected shallow open-water pools with a maximum depth of 2 m while other studies are for deeper waters. Meanwhile, recent study showed good agreement between the diffusive fluxes calculated using CC method and measurements based on floating chamber (Cole et al., 2010).~~ However, we believe that the CC method is suitable to be applied here. First, the water body in our study is located in an open area where the contribution of wind shear to the turbulence in the surface mixed layer is relatively high. During the study period in 2013, the average wind speed was 1.57 m $s^{-1}$ with a maximum of 7.1 m $s^{-1}$. Secondly, the $k_{600}$ estimated in our study was on average 0.66 m/day, well situated within the range of the $k_{600}$ which were directly measured by floating chamber or gas tracer for small lakes and ponds (Holgerson et al., 2017). Thirdly, the estimated air-water fluxes of $CH_4$ and $CO_2$ based on the CC model were also well within the range of the diffusive gas fluxes over small lakes from other studies (Erkkila et al., 2018; Mammarella et al., 2015). Finally, the parameterization of Cole and Caraco has been similarly applied to connected small open-water pools and good agreement between the model estimation and the measurements were found (Cole et al., 2010; McNicol et al., 2017). Therefore, we decided to use the CC model to estimate diffusive fluxes from the water, bearing in mind that the calculated fluxes could potentially be underestimated.

### 2.5 Estimating zone fluxes and radiative forcing

By combining EC tower and diffusive flux from the open-water, the following model can be derived

$$F_{EC} = F_{water} \times f_{water} + F_{veg} \times f_{veg} \qquad (7)$$

where $F_{EC}$ is the flux measured by EC tower, $F_{water}$ and $F_{veg}$ stand~~s~~ for the fluxes from open-water and vegetated area, respectively. ~~,~~ $f_{water}$ and $f_{veg}$ are the footprint-weighted spatial fraction of open-water and vegetated area. In this study, ebullition was neither measured nor calculated, so the flux from water was only represented by the diffusive flux.

Specifically, we first modelled the half-hourly flux footprint with a parameterization of a three-dimensional backward Lagrangian footprint model (Kljun et al., 2015) in Matlab (MATLAB 2015a, The MathWorks, Inc., Natick, Massachusetts, United States). Periods in which the wind came from the patchy forest to the southeast of the EC tower (between 100° and 200°) and the highway to the west (between 200° and 280°) were eliminated in the footprint analysis. Secondly, a land cover classification map of vegetated and open-water zones was delineated manually using a high-resolution aerial image ~~acquired from National Land Survey of Finland during the growing season of 2013~~ (data from the National Land Survey of Finland Topographic Database 06/2013~~open source: https://www.maanmittauslaitos.fi/~~) with an image manipulation software (Gimp 2.10.6, www.gimp.org). Thirdly, the flux footprints were aligned and combined with the land cover classification map to calculate half-hourly $f_{veg}$ and $f_{water}$ within 90 % footprint contour lines. Specifically, we assigned each footprint pixel within the 90 % footprint area to either open-water or vegetated area on the land cover classification map while the footprint of the pixels outside the 90 % footprint area were regarded as zero. $f_{water}$ was calculated as the sum of the footprint within the open-water area to the total footprints while $f_{veg}$ was calculated as the sum of footprint within vegetated area to the total footprints. ~~Noting that n~~None of the 90 % footprint contour lines exceeded the map area and ~~,~~ the sum of $f_{veg}$ and $f_{water}$ equaled to 1. In order to obtain the long-term aggregated footprint of carbon fluxes, we calculated also the monthly and annual aggregated footprint climatology during the study period.

The uncertainty of the vegetation and water fraction come from two sources. Firstly, the delineation of the distinct surface types was conducted based on a land surface map of the growing season in 2013, which neglected the change in the spatial extent of the vegetation throughout the presented GHG monitoring time. Secondly, ~~,~~although the footprint model used here ~~although Kljun model (Kljun, Calanca, Rotach, & Schmid, 2015)~~ is provened to be robust and general, there are uncertainties in the model predictions. To be more confident in the footprint estimation, it would be good to compare our results with footprint estimates based on large eddy simulations, however that was~~it is~~ out of the scope of the current study. With only one EC tower we could not cross check the results as was done in another study (Matthes et al., ~~,~~ ~~Sturtevant, Verfaillie, Knox, & Baldocchi,~~ 2014). However, we chose to follow a simple approach dividing the landscape into vegetation and open water because we did not observe significant vegetation expansion to cover more of the open water during the ~~2013~~ growing season in 2013 and thus the area of open water wa~~i~~s relatively constant during the monitored time periods. Vegetation establishment at the Gateway wetland was very rapid the first growing seasons and by the summer 2012 emergent vegetation had densely established at the intended shallow wetland areas. Furthermore, the clear effect of the footprint-weighted fraction of open water on the synchronization between EC $CH_4$ measurements and diffusive $CH_4$ ~~CH4~~ flux from water ~~(Line 471-477, Fig.S6 in the supplement material)~~ was nicely demonstrated ~~presented~~ in our analysis (Fig. S4~~S6~~)~~, in our analysis~~

~~, so that we think the simple method used is sufficient to capture the major pattern in vegetation and water fraction in our study.~~

To better understand the influence of greenhouse gas fluxes in this urban wetland, we calculated the sustained global warming potential (SGWP) for $CO_2$ and $CH_4$ over a hundred-year horizon in each surface type. The difference between SGWP and global warming potential (GWP) is that SGWP accounts for the effect of GHG remain~~s~~ing in the atmosphere during the period. Since $CH_4$ is a more potent greenhouse gas, we multiply the emission of $CH_4$ by a factor of 45 to convert it to kg $CO_2$-eq $m^{-2}$ $yr^{-1}$ (Neubauer and Megonigal, 2015). However, for an easy comparison between our results and those from other studies using the conventional method, we calculated also $CH_4$ fluxes as $CO_2$ equivalents using a GWP of 34 following the 5th Assessment Report of IPCC (Myhre et al., 2013). ~~The GWP of CH4 fluxes from ecosystem, water and vegetation are 0.177, 0.077 and 0.195 kg CO2-eq m−2, and they will be added to the result section.~~

**2.6 Statistical analysis**

~~The Pearson correlations (r) were determined between fluxes and environmental variables.~~ Differences in the fluxes and environmental variables between the two peak growing seasons (summer 2013 and 2014) were evaluated using the *t*-test. Cumulative annual GHG fluxes measured by the EC tower are reported as the median of the 20 ANN predictions and uncertainty are presented as 95 % confidence interval of the 20 ANN predictions. As diffusive GHG fluxes were calculated from gas concentration meteorological parameters, no standard error is reported for the cumulative annual

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fluxes from the open water. All statistical analys~~i~~es were performed in Matlab (MATLAB 2015a, The MathWorks, Inc., Natick, Massachusetts, United States).

We also conducted wavelet coherence analysis to explore the temporal correlations between fluxes and environmental variables on the multi-temporal scales (Grinsted et al., 2004; Torrence and Webster, 1998). Since the fluxes are gap-filled using some of the environmental variables, simply applying the wavelet coherence analysis to all the variables can overstate the correlations. Therefore, we only conducted wavelet coherence analysis between gap-filled ecosystem flux time series and those independent environmental variables which were not used in the gap-filling procedure (concentration of $NO_3$-N and TP) while ~~The~~ Pearson correlations ($r$) were determined between non-gapfilled fluxes and the other environmental variables.

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## 3 Results

### 3.1 Ecosystem seasonality and environmental variables

Daily average PPFD ranged from 0.9 to 691.5 µmol $m^{-2}$ $s^{-1}$ in ~~year 2013~~2013 with the highest value ~~appeared~~ appearing in July. June had the highest monthly average PPFD with 486.1 µmol $m^{-2}$ $s^{-1}$ followed by July and August with 470.2 and 430.6 µmol $m^{-2}$ $s^{-1}$, respectively. The PPFD during the peak growing season in~~ year~~ 2014 was on average 361.8 µmol $m^{-2}$ $s^{-1}$, lower than that during the same period in 2013 (Fig. ~~1~~2a).

Mean daily water temperature ($T_{water}$) ranged from 0 °C in March to 23.7 °C in June with an annual average of 7.9 °C in 2013 and from 0 °C in February to 21.4°C in July in 2014. Mean daily air temperature ($T_{air}$) had more fluctuation and ranged from -15.6 °C in January to 23.3 °C in June 2013 and from -19.0 °C in January to 23.4 °C in July 2014 (~~Fig. 1~~Fig. 2b). The open-water area experienced an ice-covered period between 1 January and 31 March 2013, while the winter 2013-2014 was so mild and warm that there was practically no snow cover during December 2013 – March 2014. Comparing the temperature between the two peak growing seasons, both $T_{water}$ and $T_{air}$ were higher in June 2013 while $T_{air}$ was lower in July 2013 than in 2014. In August, there was no significant temperature difference between the two years. Four seasons were classified for the ecosystem based on the trend in $T_{air}$ and $T_{water}$. In spring (April and May), the daily temperature started to increase, the vegetation showed a sign of early growing season and the warm temperature unfroze the lake. In summer, the peak growing season (June - August), vegetation exhibited the maximum~~-~~ growth~~growth which was reflected in the large negative GEP value~~, and the temperatures reached the annual maxima. In autumn (September and October), daily temperatures began to drop and the vegetation showed signs of early senescence. In winter (January to March and November, December), temperatures reached the annual ~~minima~~ minimum and vegetation was inactive in carbon sequestration. Precipitation was higher in August 2014 than in the preceding August, almost twice as high as that of 2013.

WL was higher in the winter and lower in the summer in 2013. The daily average of WL varied between 50.06 m in July 2013 and 50.4 m in April 2013. There was a spring peak in 2013 when the highest WL was observed due to snow melt while in 2014 no such event appeared due to the mild winter 2013-2014 without ice-covered period (~~Fig. 1~~Fig. 2c). The average daily WL from January to August was similar (50.13 cm and 50.15 cm for 2013 and 2014, respectively). However, during the peak growing season, it was on average 5.7 cm higher in 2014 than in 2013.

The annual rainfall in 2013 (snowfall not included) was 363.6 mm which happened mostly during summer and autumn (~~Fig. 1~~Fig. 2d). The maximum daily-averaged rainfall was in August (26.7 mm day$^{-1}$) while monthly-averaged rainfall was highest in November with 73.8 mm month$^{-1}$ followed by August with 68.3 mm month$^{-1}$. In 2014, an exceptionally high amount of rainfall was observed in August (125.7 mm month$^{-1}$), while the amount of rainfall in the other months were similar to 2013.

The daily-averaged $CO_2$ concentration in the water ([$CO_2$]) in 2013 had large variation with the maximum (461 µmol L$^{-1}$~~9324 ppm~~) and the minimum (~~353 ppm~~21.6 µmol L$^{-1}$) ~~both happening in October~~ (~~Fig. 1~~Fig. 2e). ~~[$CO_2$] was higher in summer months (5457 ppm) and lower in winter months (3345 ppm).~~ [$CO_2$] was higher in 2014 with an average of 262.6 µmol L$^{-1}$ ~~4924 ppm~~ from January to August than in 2013 with an average of 211.5 µmol L$^{-1}$~~3781 ppm~~. It also exhibited seasonal variation with high concentration in summer (~~8084 ppm)~~360.3 5 µmol L$^{-1}$~~-~~) and low concentration in winter (223.4 µmol L$^{-1}$~~3513 ppm~~). The [$CO_2$] measured in the inflow was generally lower than that in the outflow and they were well correlated ($r$=0.84). [$CH_4$] in the outflow was on average five times higher in 2014 than in 2013. The average annual concentration was 0.81 µmol L$^{-1}$ in ~~year 2013~~2013 and 2.25 µmol L$^{-1}$ in 2014. There were peak [$CH_4$] episodes in the outflow in May 2013 with a maximum of 5.43 µmol L$^{-1}$. During the summer months in 2014 there were even higher outflow [$CH_4$] peaks with a maximum of 16.83 µmol L$^{-1}$. The [$CH_4$] had a mean of 0.42 µmol L$^{-1}$ in the inflow which was lower than that in the outflow, and there ~~was~~ were no prominent [$CH_4$] peaks observed in the inflow. [$CH_4$] in the inflow and outflow were weakly correlated ($r$ = 0.2) (~~Fig. 1~~Fig. 2f).

The median concentration of total phosphorus~~TP~~ (TP) ~~concentration~~ measured at the outflow monitoring station was 56 µg L$^{-1}$ and the median $NO_3$-N concentration was 0.69 mg L$^{-1}$~~L−1.~~ in ~~year 2013~~2013 (Fig. 2g, 2~~, and~~ h). In the annual perspective, TP and ~~NO3-N~~ $NO_3$-N concentration consisted of several runoff peaks occurring after rain or snow melting events. This wetland serves as a nutrient removal measure as it improved water quality by retaining P and N from runoff before the release to the receiving lake, where the annual TP reduction was 1~~3~~0 % and $NO_3$-N ~~NO3-N~~ reduction was 3~~1~~4% from the original concentration in ~~3~~2013 ~~4 (Valkama et al., 2017)~~(Wahlroos, 2019).

### 3.2 Flux footprint mapping

A footprint distribution was modeled for each half hour when an eddy flux measurement was collected at the EC tower. The open-water area accounted for 10 % to 16 % of the total wetland area within the footprint while the rest was comprised of wetland vegetation. When weighted with footprint distribution, $f_{water}$ ranged from 0 to 25.5 % and $f_{veg}$ from 74.5 % to 100 %. The 1st quar~~rn~~tile, median and 3rd quar~~rn~~tile of $f_{water}$ and $f_{veg}$ were 0.09 %, 14.1 %, 17.9 % and 82.1 %, 85.9 %, 91.3 %, respectively.

The monthly cumulative footprint was slightly different for $CO_2$ and for $CH_4$ due to the different missing flux values. However, the difference on average was so small (7 %) and the footprint of $CO_2$ was used in further analysis. The flux footprints were shown to be northeast to the EC mast due to the wind direction filtering meaning only half-hourly data with wind directions from the wetland area were considered in the analysis (Fig. ~~S~~S2~~3~~). The monthly-average of the 90 % footprint area covered a minimum of 0.69 ha to a maximum of 2.28 ha with a mean of 1.3 ha. The mean extent of the 90 % flux footprints was 128 m. After applying the flux footprint function, the monthly-average of the footprint-weighted spatial fraction of open water showed lower value in summer and higher value in winter ranging from 11.3 % to 21.4 %

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with a mean of 13.3 % in 2013. In 2014 during the peak growing season, on average 13.8 % of the wetland area was ~~comprised of~~ open water and the mean $f_{water}$ was 10 %.

### 3.3 $CO_2$ and $CH_4$ fluxes

#### 3.3.1 Ecosystem $CO_2$ and $CH_4$ fluxes

Ecosystem $CO_2$ and $CH_4$ fluxes measured by the EC tower showed the ecosystem was nearly $CO_2$ neutral and it was a small $CH_4$ source in ~~year 2013~~2013.

Daily average of NEE was near zero during winter time (January to March, on average 0.37 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$), slightly positive in spring and it became negative from the end of May till the end of August indicating the ecosystem was a $CO_2$ sink during this period, with a maximum negative value of -5.14 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$ in June. Daily-average NEE was highest in September with a maximum of 3.29 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$~~, possibly due to the suppressed GEP and high $R_{eco}$~~. In October, November and December, NEE remained low but still positive (on average 0.77 g µmol C-$CO_2$ $m^{-2}$ $s^{-1}$), demonstrating the milder winter between 2013 and 2014 (Fig. 3~~2 and Fig. S4~~). NEE~~, GEP and $R_{eco}$~~ exhibited strong seasonality in 2013, which ~~NEE~~ was negative during June, July and August meaning the ecosystem was a $CO_2$ sink while the rest of year it was a $CO_2$ source. NEE was lowest in June and highest in September. ~~Both GEP and $R_{eco}$ achieved their highest values in July (Fig. S4).~~ The cumulative NEE in 2013 was 8 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ with the 95% confidence interval between -18.9 and 34.9 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ (Fig. 3~~2~~).

Daily-averaged $CH_4$ was low but not negligible from January to April (on average 5.1 nmol C-$CH_4$ $m^{-2}$ $s^{-1}$), with a sudden rise in the end of May reaching a maximum of 48.9 nmol C-$CH_4$ $m^{-2}$ $s^{-1}$. During summer months the ecosystem exhibited relatively high $CH_4$ emission (on average 15.4 nmol C-$CH_4$ $m^{-2}$ $s^{-1}$), not comparable with the emission episode in May but higher than winter months. In autumne (September and October) the daily-average CH4 was 8.8 nmol C-$CH_4$ $m^{-2}$ $s^{-1}$ and after that it gradually decreased throughout the rest of the year with an average of 5.5 nmol C-$CH_4$ $m^{-2}$ $s^{-1}$. The cumulative $CH_4$ for 2013 was 3.9 g C-$CH_4$ $m^{-2}$ $yr^{-1}$ with the 95% confidence interval between 3.75 and 4.07 g C-$CH_4$ $m^{-2}$ $yr^{-1}$ (Fig. 4~~3~~).

Comparing the peak growing season between 2013 and 2014, the 30-min ~~s~~ NEE ranged from -20.0 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$ in June to 18.5 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$ in September 2013. ~~GEP reached maximum negative value in July 2013 with -30.5 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$ and $R_{eco}$ in June with 13.9 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$.~~ During the peak growing season 2014, NEE had the lowest value of ~~-~~-22.6 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$ in June~~, GEP -28.6 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$ and $R_{eco}$ had its maximum in the beginning of August 2014 with 11.3 µmol C-$CO_2$ $m^{-2}$ $s^{-1}$~~. The monthly NEEs of peak growing season were -84.1, -76,1 and -22.2 g C-$CO_2$ $m^{-2}$ $month^{-1}$ in June, July and August 2013, and -97.6, -47,5 and -19.6 g C-$CO_2$ $m^{-2}$ $month^{-1}$ in 2014. ~~In both years, daily averaged GEP had its maximum negative value in July (-13.4 and -12.8 g C-$CO_2$ $m^{-2}$ $d^{-1}$). Daily-averaged $R_{eco}$ was highest in June 2013 with 12.1 g C-$CO_2$ $m^{-2}$ $d^{-1}$ while in 2014 $R_{eco}$ was low in June and the peak was in the end of July with 10.5 g C-$CO_2$ $m^{-2}$ $d^{-1}$ (Fig. S5a).~~ The average $CH_4$ emissions in June, July and August were 24.4, 10.8 and 11 nmol $m^{-2}$ $s^{-1}$ in 2013, and 15.5, 21.3 and 21.3 nmol $m^{-2}$ $s^{-1}$ in 2014, respectively~~ (Fig. S5b)~~.

#### 3.3.2 Diffusive $CO_2$ and $CH_4$ fluxes from open-water area

Diffusive $CO_2$ and $CH_4$ fluxes from the open water were estimated based on wind speed, [$CO_2$] and [$CH_4$] (See Sect. 2.4). The variation of diffusive fluxes demonstrated a pattern driven by both wind speed in short term and gas concentration dynamics in the water in long term. Diffusive $CO_2$ fluxes ranged from -0.07 to 4.09 µmol $CO_2$ $m^{-2}$ $s^{-1}$ with a mean of 1.04

µmol $CO_2$ m$^{-2}$ s$^{-1}$ in 2013 indicating $CO_2$ oversaturation in the water. From June to September the averaged flux (1.27 µmol $CO_2$ m$^{-2}$ s$^{-1}$) was higher than that of the other months (Fig. 5~~4~~a), corresponding to the higher [$CO_2$] in the water during summer months (~~Fig. 1~~Fig. 2d~~e~~). The monthly-averaged diffusive $CO_2$ flux during peak growing season in 2014 was 2.34, 2.71 and 1.99 µmol $CO_2$ m$^{-2}$ s$^{-1}$ for June, July and August, significantly higher than during the same period in 2013 due to the high [$CO_2$] in the open water (~~Fig. 1~~Fig. 2e~~d~~).

The average diffusive $CH_4$ emissions in 2013 was 4.9 nmol C-$CH_4$ m$^{-2}$ s$^{-1}$, where a peak emission appeared in late May with the highest flux of 137.6 nmol C-$CH_4$ m$^{-2}$ s$^{-1}$. Monthly-averaged $CH_4$ diffusive fluxes showed an increasing trend towards the end of the year with large variation in May due to the peak concentration episode. This phenomenon was mainly driven by the increasing dissolved $CH_4$ concentration in the outflow in 2013. The monthly-averaged diffusive $CH_4$ flux during peak growing season in 2014 was 20.9, 18.9 and 13.5 nmol $CH_4$ m$^{-2}$ s$^{-1}$ for June, July and August, respectively and they were significantly higher than the same period in 2013 due to the high [$CH_4$] in the open water (~~Fig. 1~~Fig. 2f~~e~~).

### 3.3.3 Diel patterns in $CO_2$ and $CH_4$ fluxes

Only non-gapfilled data were used for determination of diel patterns in both gas fluxes. $CO_2$ and $CH_4$ fluxes from the vegetated areas ($F_{veg}$) were~~as~~ calculated for each 30-min interval according to formula (7~~5~~). As expected, $CO_2$ flux showed a strong diel pattern in summer with $CO_2$ uptake during daytime and release in the night, which was controlled by photosynthetic activity (Fig. ~~5~~6a). The summer peak $CO_2$ uptake reached 11.5 µmol m$^{-2}$ s$^{-1}$ for the whole constructed wetland ecosystem and 15.2 µmol m$^{-2}$ s$^{-1}$ for the vegetated areas. The $CO_2$ flux from the vegetated areas had higher maximum uptake than the EC measurements carried out over the whole constructed wetland. In the winter, the $CO_2$ fluxes from both tower and vegetation were similar, being on average 0.46 and 0.55 µmol m$^{-2}$ s$^{-1}$ respectively (Fig. 6~~5~~b).

$CH_4$ flux also showed diel patterns in the summer with much larger variability than those from $CO_2$ flux. $CH_4$ emission in general was higher in daytime than in nighttime. In the daytime in summer, $CH_4$ flux from the vegetated area was higher than the flux measured from the tower while there was no difference during the nighttime (Fig. 6~~5~~c, 6~~5~~d). The summer peak daytime flux from the tower (18.9 nmol m$^{-2}$ s$^{-1}$) and vegetated area (24.7 nmol m$^{-2}$ s$^{-1}$) was 2.4 times and 3.3 times higher than the nighttime flux (7.5 nmol m$^{-2}$ s$^{-1}$), respectively. This can be understood as daytime $CH_4$ flux is linked with photosynthesis while nighttime $CH_4$ flux is controlled by other processes like diffusion, ebullition and convection between the soil, water and atmosphere. In winter there was small (on average 4.6 nmol m$^{-2}$ s$^{-1}$) but constantly positive $CH_4$ flux without obvious diel pattern.

## 3.4 Environmental variables with fluxes

Only non-gapfilled flux data were used in the Pearson correlation analysis between environmental variables and flux pairs. Radiation, $T_{air}$ and $T_{water}$ all had high negative correlation coefficient (*r*) with NEE and high positive *r* with $CH_4$ flux in 2013, corresponding to the results of ANN model parameter selection. Radiation was best correlated with NEE and $T_{water}$ was best correlated with $CH_4$ (Table 1). The correlations were rather weak (small *r* or even the opposite sign of *r*) during 2014 due to the short measuring period and narrow ranges of the variables. Water level was positively correlated with NEE and negatively correlated with $CH_4$, which was counter intuition, possibly because it was masked by temperature variation as the water level was in general higher in winter and lower in summer. [$CO_2$] and [$CH_4$] were not correlated with either NEE or $CH_4$ although they were shown to be important parameters in ANN model selection.

$NO_3$-N did not show consistent correlation with any of the fluxes (Fig. 7a, 7b). The variation of TP was negatively leading the change in NEE at 1-day scale (more TP leads to more $CO_2$ uptake; Fig. 7c) where the time lag varies between 1 to 5

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### 3.5 Estimating radiative forcing from different zones

To obtain the climate forcings from each land surface type, we calculated the ~~half-hourly and~~ annual ~~gas~~ cumulative fluxes from the vegetated area based on eq. (7) using footprint-weighted spatial fraction, ecosystem fluxes and diffusive fluxes from the open water (See Sect. 2.5). The annual median value of footprint-weighted spatial extent was used to calculate the annual fluxes, which showed open-water area was a $CO_2$ source (297.5 g C-$CO_2$ $m^{-2}$ $yr^{-1}$) and vegetated area was a $CO_2$ sink (-39.5 C-$CO_2$ $m^{-2}$ $yr^{-1}$). Both open-water and vegetated area were $CH_4$ sources but the $CH_4$ emission from vegetated area was higher than open-water area, being 4.26 and 1.73 g C-$CO_2$ $m^{-2}$ $yr^{-1}$, respectively (Table 2).

Open water has contributed a large amount of $CO_2$ emission into the atmosphere through diffusion (1.09 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$) whereas the $CH_4$ emission was relatively small (0.104 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$). Vegetated area was a small sink of $CO_2$ but the cooling effect of vegetation by $CO_2$ uptake was relatively small (-0.145 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$) compared to its $CH_4$ emission (0.256 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$). Overall, the ecosystem had a small warming effect with 0. 263 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$ of which 89% was contributed by $CH_4$ (Table 2). ~~The GWP of CH4 fluxes from ecosystem, water and vegetation are 0.177, 0.077 and 0.195 kg CO2-eq m-2, **and they will be added to the result section.**~~

## 4 Discussion

### 4.1 The GHG fluxes from an urban stormwater wetland ecosystem

The studied urban wetland ecosystem was a small carbon source over the full-year studied period in ~~year 2013~~2013. Due to the scarcity of studies on urban wetlands using the EC method, we compare our results to restored wetlands which can be considered to be proxy ecosystems to urban wetlands with both including ~~re~~wetting practice in an ecosystem which has been drained or dry previously. The annual $CO_2$ balance of 8 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ from the ecosystem, or -39.5 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ from the vegetated area (Table 2), were small compared to a restored wetland in western Denmark where the annual $CO_2$ balance ranged from -286 to -53 g C-$CO_2$ $m^{-2}$ $yr^{-1}$ (Herbst et al., 2013), and the annual $CH_4$ balance of 3.9 g C-$CH_4$ $m^{-2}$ $yr^{-1}$ was less than half of the annual $CH_4$ emission (between 9 and 13 g C-$CH_4$ $m^{-2}$ $yr^{-1}$) in that study. Over a network of restored freshwater wetlands in ~~the~~ California, the $CO_2$ sequestration can be up to nearly 700 g C $m^{-2}$ $yr^{-1}$ and $CH_4$ emission up to 63 g C $m^{-2}$ $yr^{-1}$ (Hemes et al., 2018). It is not surprising that the studied ecosystem appeared to $CO_2$ neutral as it was recently constructed. The herbaceous vegetation has been allowed to fully self-establish without human intervention and at the early successional stage, plant diversity and biomass were still increasing each year (Wahlroos, 2019). With the vegetation being more developed, a greater $CO_2$ uptake from the vegetated area can be expected in the following years. The low $CH_4$ emission observed in this study may be due to the depletion of organic matters in the bottom soil from agricultural uses thus it provided little substrate for anaerobic microbial activity to produce $CH_4$. With the accumulation of organic matters in the anoxic wetland sediment, $CH_4$ production may increase in the future. Certain chemical compounds like Fe in mineral soils can also inhibit $CH_4$ production leading to much lower ecosystem-scale $CH_4$ flux (Chamberlain et al., 2018). In the meanwhile, methane-oxidizing bacteria (methanotroph) regulates $CH_4$ consumption at the soil-water interface. With the ecosystem being used previously as cropland, the physical disturbance of soil may have greatly reduced the methanotroph communities so that the $CH_4$ oxidation may also be low in the soil (Smith et al., 2000; Saggar et al., 2008). Furthermore, after the initial establishing phase, the ecosystem productivity can also be reduced at a water treatment wetland due to the standing litter that inhibits the generation of new vegetation growth. It was shown that in a restored freshwater wetland the ecosystem was a net $CO_2$ sink (-804 ± 131 g C-$CO_2$ $m^{-2}$ $yr^{-1}$) in 2002-2003, six years after the restoration but near $CO_2$ neutral in 2010-2011 due to the reduced photosynthetic plants (Anderson et al., 2016). Thus, given the urban wetland is sustained for a sufficiently long period, it is still unclear whether the $CO_2$ uptake from the vegetated zone would compensate for its $CH_4$ emission, not considering the large GHG emission from the open-water zone. Thus, similar studies as the present one should be conducted at a later stage after the construction of the wetland to fully reveal the GHG balance of the ecosystem along time.

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Overall, the ecosystem $CO_2$ and $CH_4$ fluxes measured by the EC tower ranged from -5.33 to 3.4 g C-$CO_2$ $m^{-2}$ $day^{-1}$ and from 1.0 to 55.2 mg C-$CH_4$ $m^{-2}$ $day^{-1}$, respectively. They are consistent with the flux ranges provided by other studies on GHG fluxes in restored wetlands (Anderson et al., 2016; Knox et al., 2015; Matthes et al., 2014; Morin et al., 2014b; Herbst et al., 2013), although for both gases they tend to be on the lower end. NEE~~, GEP and $R_{eco}$~~ exhibited seasonal variation so that the ecosystem was a $CO_2$ sink between June and August. The highest NEE appeared in September possibly because ~~GEP~~ photosynthesis has greatly reduced due to plant senescence~~t~~ while ecosystem respiration ~~$R_{eco}$~~ remained relatively high because of the warm temperature ~~(Fig. S4)~~. Previous studies have found good agreement between $CH_4$ emission and ~~GEP~~ photosynthesis as plants provide substrates for methanogenesis (Rinne et al., 2018), which was not observed in the daily-average of gas fluxes in this study (Figs 2 and 3) as the peak $CH_4$ flux appeared in May and peak gross primary productivity~~GEP~~ appeared in July (data now shown). Nonetheless, both $CH_4$ and $CO_2$ fluxes, especially those obtained from the vegetated areas, exhibited strong diurnal cycles during summer with synchronized peaks around noon (Fig. 6~~5~~a, 6~~5~~c). This finding reflects that short-term $CH_4$ emission from vegetation is linked with photosynthesis by providing labile carbon from root exudate and by gas transport through aerenchyma and open stomata while long-term $CH_4$ emission may be determined by complex processes related to environmental variables e.g. temperature and redox potential (Linden et al., 2014).

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### 4.2 Parsing GHG fluxes from heterogeneous land surfaces

We found that the open-water area was constantly a source of $CO_2$ and $CH_4$ to the atmosphere during the studied period as the [$CO_2$] and [$CH_4$] in the water generally exceeded the atmosphere equilibrium except during the ice-covered period (Fig. 4~~5~~). The annual average of [$CO_2$] in the surface water in 2013 was 0.3% in our study, comparable to 0.4% in another temperate restored wetland (McNicol et al., 2017), while the seasonal pattern (higher in summer and fall) was the opposite as they have found. We also found that both [$CO_2$] and [$CH_4$] were higher in 2014 than 2013 (~~Fig. 1~~Fig. 2d, 1e). The $O_2$ concentration ([$O_2$]) and $O_2$ balance ([$O_2$]$_{outlet}$ - [$O_2$]$_{inlet}$) measured by another study on the same wetland (Wahlroos, 2019) could partially explain the observed phenomenon. The relatively high water temperature and oxic conditions in the water in fall 2013 have allowed high decomposition of detritus leading to high [$CO_2$] (Wahlroos, 2019). The long period of hypoxia during summer 2014 could explain the three-fold increase in [$CH_4$] as the condition was more favorable for $CH_4$ production. The negative $O_2$ balance in summer 2014 indicated strong $O_2$ consumption by microbial decomposition producing $CO_2$ in the water. As the long-term diffusive fluxes (daily and monthly) ~~was~~ were mainly driven by gas concentration in the water, it was straight-forward to understand high diffusive $CO_2$ and $CH_4$ fluxes in 2014 ~~comparing~~ compared to 2013. Interestingly, the ecosystem $CH_4$ emission in 2013 was well synchronized with the diffusive $CH_4$ flux by capturing sporadic emission episodes from the water (Fig. S4~~S6~~a, S4~~S6~~c) while they were not synchronized in summer 2014 although several stronger diffusive peaks happened (Fig. S4~~S6~~b, S4~~S6~~d). When footprint-weighted contribution was

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accounted for, it clearly revealed that the synchronization of $CH_4$ emission from ecosystem and water was closely related to the flux footprint distribution. When there was high flux contribution from the open water (20-25 %), high diffusive $CH_4$ was also reflected in ecosystem flux measured by EC. This has further proved the application of footprint analysis is essential in explaining gas exchange from heterogeneous surfaces using EC data.

It is worth noticing that in our study we only classified the surface landscapes into "open water" and "vegetation" but neglected the difference in sink/source strength from different plant types within the vegetation zone (Fig. ~~S~~1). We did not account for the dissimilarity between vegetation types because the characteristics in gas exchange are much more distinct between open water and vegetation, which was the focus of this study. For the same reason, ebullition was not considered in this study neither, as ebullition was shown to have only minor significance in a restored wetland accounting for less than 0.1% of ecosystem $CO_2$ flux and 4.1% of ecosystem $CH_4$ flux (McNicol et al., 2017). However, for a proper downscaling analysis of EC data, the subareas of different plant types and ebullition should also be taken into account.

**4.3 Climate impact of urban wetland and implications for management**

In the present study, the urban boreal wetland had an overall SGWP of 0.263 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$ which was comparable or higher than other restored wetlands in boreal region (Herbst et al., 2013), and within the range of inter-annual variation or lower than restored wetlands in temperate zone (McNicol et al., 2017; Anderson et al., 2016). Different from other studies, the urban wetland was $CO_2$ neutral and a $CH_4$ source. It is worth noting that the paramount contribution of $CH_4$ in ecosystem SGWP was mainly driven by the large footprint-weighted spatial area of the vegetation (See Sect. 3.2). In fact, The SGWP of GHG emission from open water (1.194 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$) was 10 times as large as that from vegetation (0.111 kg $CO_2$-eq $m^{-2}$ $yr^{-1}$) (Table 2). The implication of this result is that during wetland restoration, it would be more beneficial to have large patches of emergent vegetation ~~area~~ at least from the GHG emission point of view. Similar results have been obtained by other studies as well that open water has more climate-warming impact than emergent vegetation due to the large diffusive fluxes from open water (Stefanik and Mitsch, 2014~~;~~; McNicol et al., 2017). The climate impact of natural wetland depends on the net balance between the cooling effect of $CO_2$ uptake by vegetation and the warming effect of other GHG emissions, mainly $CH_4$ (Bridgham et al., 2013). In wetlands constructed in urban areas, the ~~large fraction~~ extent of open water, which is a significant emitter of $CO_2$, should also be taken into consideration when evaluating the role of urban wetland in global climate change.

~~"Firstly, in our study we found that the radiative forcing effect of the open-water area exceeded the vegetation area in an urban wetland in Finland. Thus, if considering only the climate impact, it would be advisable to have lower water/vegetation fraction which means limiting open-water surfaces and setting a design preference for areas of emergent vegetation in the establishment of urban wetlands. Secondly, our~~Our results also showed that total phosphorus enhanced both $CO_2$ ~~CO2~~ uptake and $CH_4$ ~~CH4~~ emission which have contradictory climate impacts to the ecosystem (Fig. 7b, 7d). Although it is out of the scope of our study, it would be very interesting to understand the mechanisms, to quantify the magnitude and the duration of these enhancements induced by nutrient input. Previous studies have found that ~~nutrien~~nutrient ~~i~~inputs can influence the identity of the key primary producer (submerged plants versus phytoplankton) in the water, which is crucial in shaping the $CH_4$ ~~CH4~~ emission from shallow water (West et al.~~, Creamer, & Jones,~~ 2016~~;~~; Davidson et al., 2018). Submerged plants may decrease $CH_4$ ~~CH4~~ production in the lake by ~~producing alleochemicals,~~ transporting oxygen to the sediment and providing good habitat for $CH_4$ ~~CH4~~ oxidizing bacteria (Heilman & Carlton, 2001), while phytoplankton was shown to significantly increase $CH_4$ ~~CH4~~ ebullition by changing the quality of the dissolved organic carbon which promotes methanogenesis (West et al., 2016) or/and by altering the sediment texture and redox conditions favoring the release of bubbles. As a result, we suggest to control the nutrient input to the water of the newly established wetland to limit the abundance of phytoplankton as well as to support the existence of submerged plants.

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## 5 Conclusions

Urban wetlands have received global attention as a nature-based urban runoff management solution for sustainable cities, as they provide cost efficient flood control and water quality mitigation as well as many ecological and cultural services. In the meantime, the climate impact of urban wetlands should also be considered. Wetting a landscape may enhance the $CO_2$ sequestration in the ecosystem, whereas $CH_4$ can be emitted due to the anaerobic conditions in the soil after wetting. Furthermore, heterogeneity induced in newly created urban wetlands may contribute differently to the overall climate impact.

In the present study, for the first time a full annual carbon balance of an urban stormwater wetland in the boreal region was evaluated and the radiative forcing from heterogeneous landscapes were presented. We found that, during the monitored period at the study wetland, both the open water area and the vegetated area within the created wetland were carbon sources, and thus the urban wetland had a net climate warming effect, the monitored fourth year after the wetland establishment. However, if the same carbon content from the contributing watershed would have reached the receiving lake without treatment at the studied in-stream created wetland, conversion to $CH_4$ would likely have exceeded emissions observed at the wetland. The radiative forcing effect of the open-water area exceeded the vegetated area, which indicated that limiting open-water surfaces and setting a design preference for areas of emergent vegetation in the establishment of urban wetlands can be a beneficial practice when considering only the climate impact of a created urban wetland. In the meanwhile, we also emphasize that the value of urban wetlands should not be determined solely by GHG radiative forcing. The values of urban wetlands in other areas e.g. flood control, pollutant removal, biodiversity, recreation and education are as well of paramount importance to human society.

**Data availability**

Eddy covariance, gas concentration and meteorological data are available from the DRYAD database at https://datadryad.org/stash/share/WrtTNnpIt6FgLoMSZ_Wlr0lK22IcxqjGZAStuuKdHLs

**Author contribution**

IM, OW, HV, AO and TV designed the field study. SH, IM and JP carried out eddy covariance measurements, automatic gas concentration measurements in the open water and manual field measurements. XL and IM participated in eddy covariance data processing and analysis. XL analysed the results and prepared the manuscript with contributions from all co-authors.

**Competing interests**

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The authors declare that they have no conflict of interest.

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## Tables

Table 1. Pearson correlation coefficient (*r*) between the daily averages of environmental variables and fluxes in ~~year~~ ~~2013~~2013 and 2014. NEE – net ecosystem exchange; $T_{air}$ – air temperature; $T_{water}$ – water temperature; PPFD – photosynthetic photon flux density; WL – water level; [$CO_2$] and [$CH_4$] – $CO_2$ and $CH_4$ concentration measured in the outlet; * indicates only peak growing seasons (June, July and August) are included in the analysis.

| Flux | Year | $T_{air}$ | $T_{water}$ | PPFD | WL | [$CO_2$] | [$CH_4$] |
|---|---|---|---|---|---|---|---|
| $CO_2$ | 2013 | -0.45 | -0.61 | -0.62 | 0.46 | -0.34 | 0.18 |
| | 2014 | 0.43 | 0.54 | -0.12 | 0.12 | -0.12 | -0.05 |
| $CH_4$ | 2013 | 0.61 | 0.65 | 0.56 | -0.3 | 0.17 | -0.09 |
| | 2014* | 0.37 | 0.26 | 0.27 | -0.24 | 0.28 | 0.25 |

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Table 2. Annual $CO_2$ and $CH_4$ exchange ~~from different surface zones~~, ~~and~~ their sustained global warming potential (SGWP) and global warming potential (GWP) from different surface zones in Nummela wetland in 2013. "Ecosystem", "W~~w~~ater" and "~~v~~Vegetation" represent flux, ~~and~~ SGWP and GWP measured or calculated from the ecosystem by EC tower, from the open water and from the vegetated areas, respectively. The numbers in the square bracket represent the 95% confidence interval of the average. No error bounds are reported for flux, ~~and~~ SGWP and GWP from open water as they are modelled using gas concentration in the water and meteorological measurements.

| | | Ecosystem | Water | Vegetation |
|---|---|---|---|---|
| Flux (g C m$^{-2}$) | $CO_2$ | 8 [-18.9, 34.9] | 297.5 | -39.5 [-70.8, -8.1] |
| | $CH_4$ | 3.9 [3.8, 4.1] | 1.7 | 4.3 [4.1, 4.5] |
| SGWP (kg $CO_2$-eq m$^{-2}$) | $CO_2$ | 0.029 [-0.069, 0.128] | 1.090 | -0.145 [-0.260, -0.030] |
| | $CH_4$ | 0.234 [0.225, 0.244] | 0.104 | 0.256 [0.246, 0.268] |
| GWP (kg $CO_2$-eq m$^{-2}$) | $CH_4$ | 0.177 [0.170, 0.185] | 0.077 | 0.195 [0.187, 0.204] |

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**Figures**

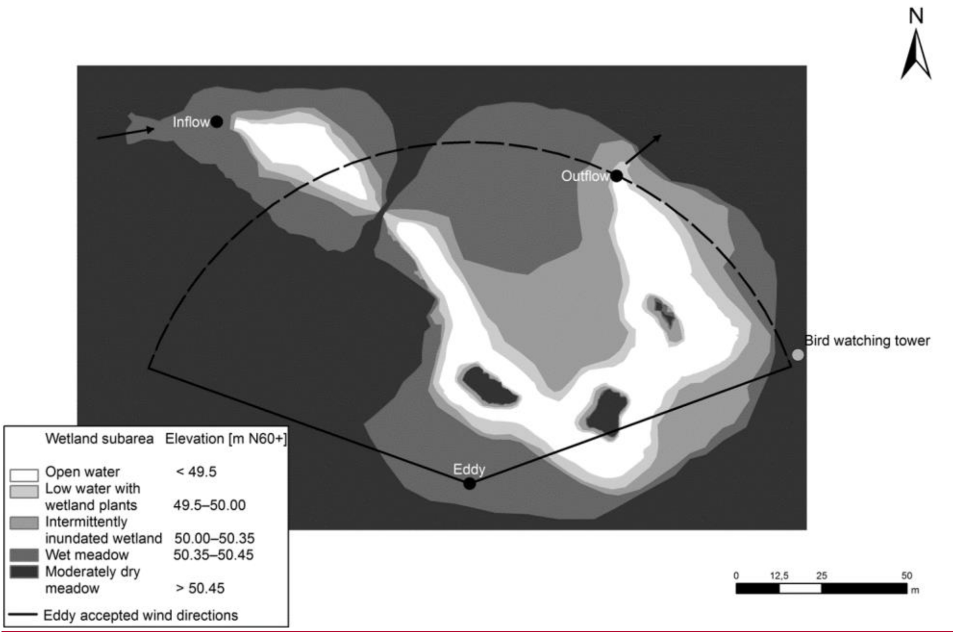


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Figure 1:- The landscape classification of Nummela wetland. Wetland subareas specified according to mean water level are shown with different colors. The arrows indicate the direction of the water flow. The black dots indicate the inflow and outflow measuring stations and the location of the EC tower.

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~~The aggregated footprint climatology of Nummela wetland in August 2013. White contour lines show 10% - 90% flux footprint climatology. The blue cross indicates the location of eddy covariance tower.~~

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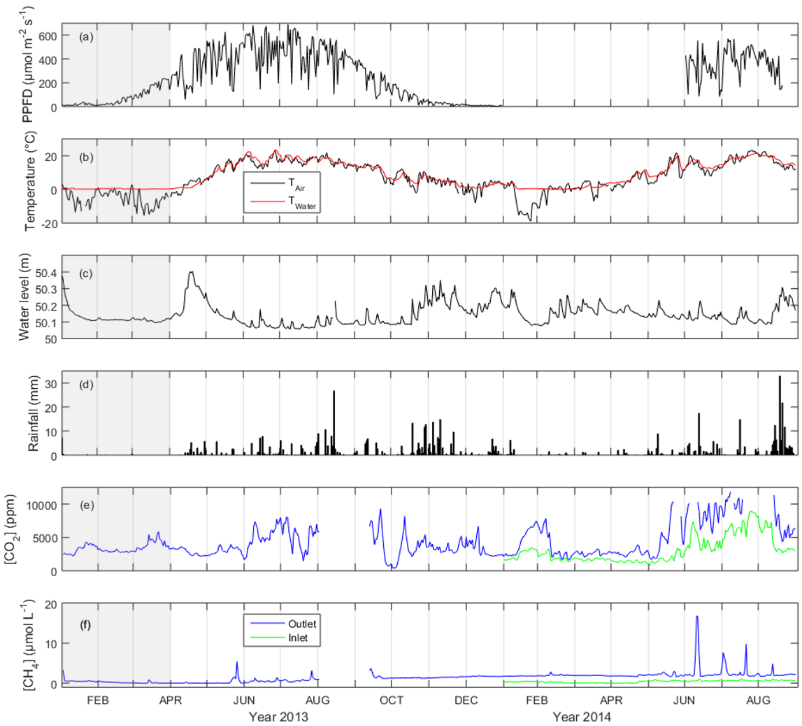


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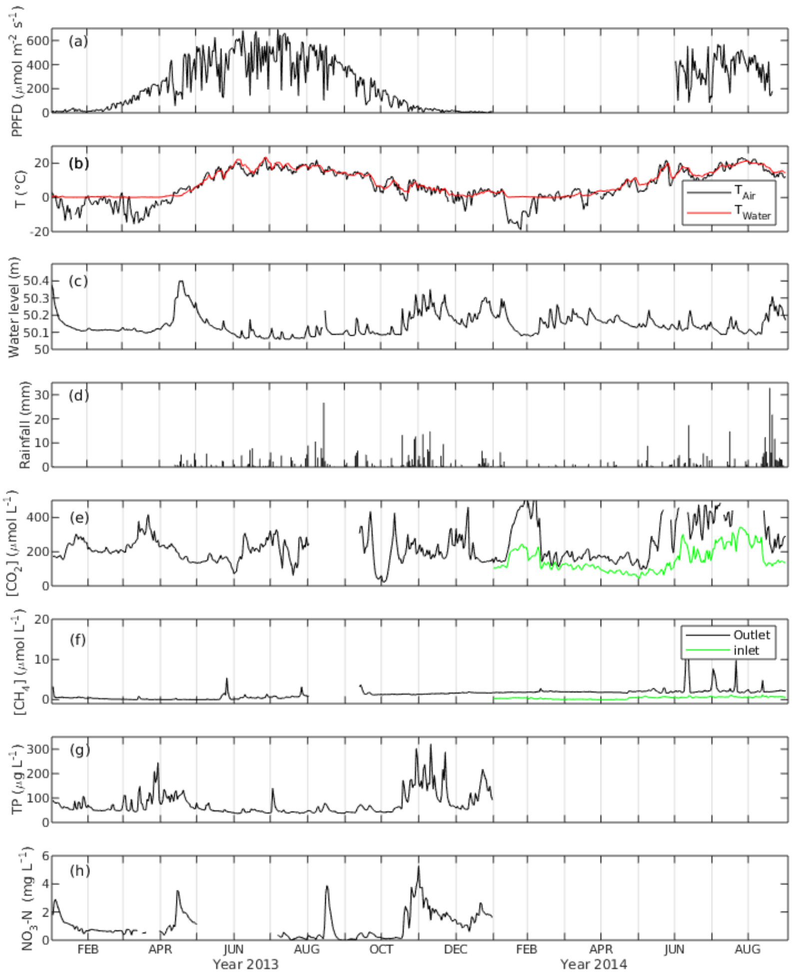


Figure ~~1~~2: The daily-average of (a) photosynthetic photon flux density (PPFD), (b) air and water temperature ($T_{air}$ and $T_{water}$), (c) water level, (d) rainfall, (e) $CO_2$ ~~$CO_2$~~ concentration ([$CO_2$]), ~~and~~ (f) $CH_4$ concentration ([$CH_4$]), (g) concentration of total phosphorus (TP) and (h) concentration of $NO_3$-N of ~~from inlet and outlet of~~ Nummela wetland from January 2013 to August 2014. ~~The grey zone indicates the ice-covered period.~~

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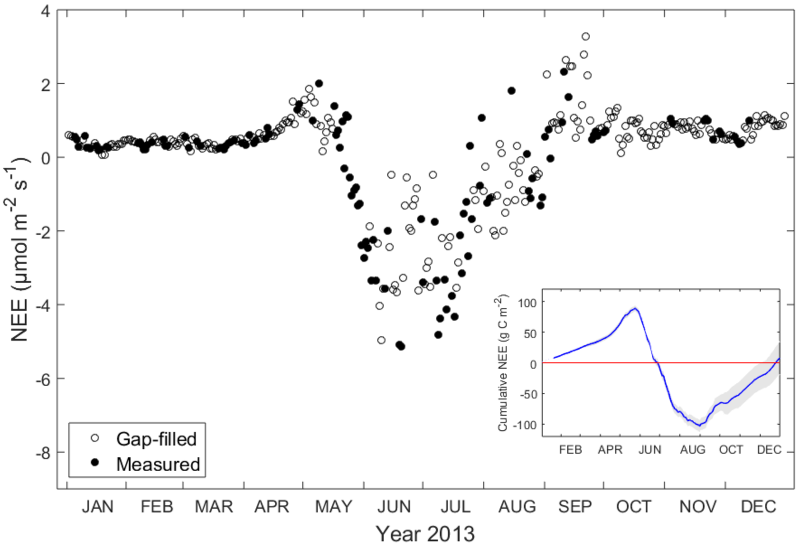


Figure 3~~2~~: Daily average of net ecosystem exchange of $CO_2$ (NEE, μmol $m^{-2}$ $s^{-1}$) in ~~year 2013~~2013. Filled dots indicate measurement (when available half-hourly measurement data ⩾ 10) and circles indicate gap-filled data (when available half-hourly measurement data < 10). The insert shows cumulative NEE (g C $m^{-2}$) in the ecosystem and the red line indicates the zero reference line. Error bounds (marked in grey) on cumulative NEE reflect the 95 % confidence interval for the gap-filling procedure.

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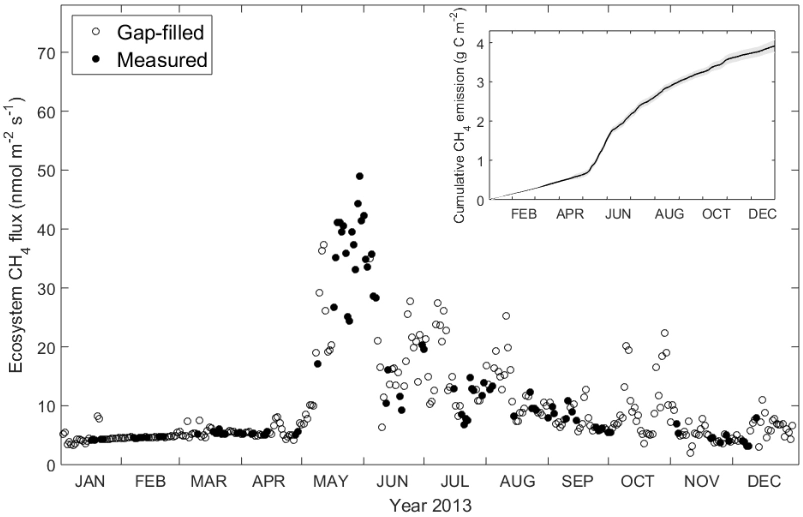


Figure ~~3~~4: Daily average of ecosystem $CH_4$ flux measured by EC tower and cumulative $CH_4$ emission in ~~year 2013~~2013. Filled dots indicate measurement (when available half-hourly measurement data ⩾ 10) and circles indicate gap-filled data (when available half-hourly measurement data < 10). The insert shows cumulative $CH_4$ emission with the error bounds in grey reflecting the 95 % confidence interval for the gap-filling procedure.

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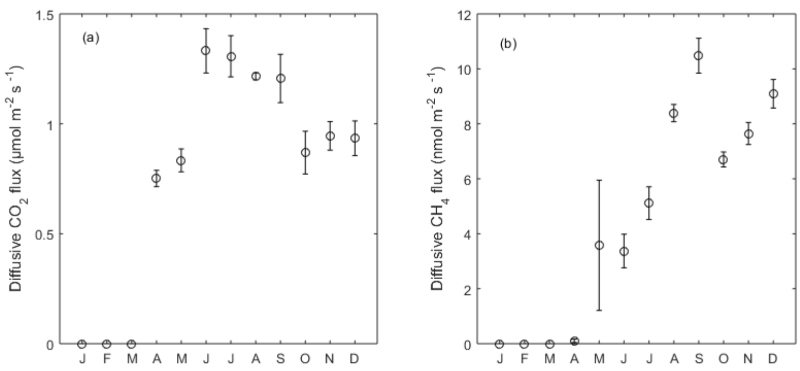


Figure 5~~-4~~: Monthly-average of (a) diffusive $CO_2$ and (b) $CH_4$ flux from the open-water in ~~year~~ 2013. Error bar indicates the standard error of the mean. From January to March there was an ice-covered period.

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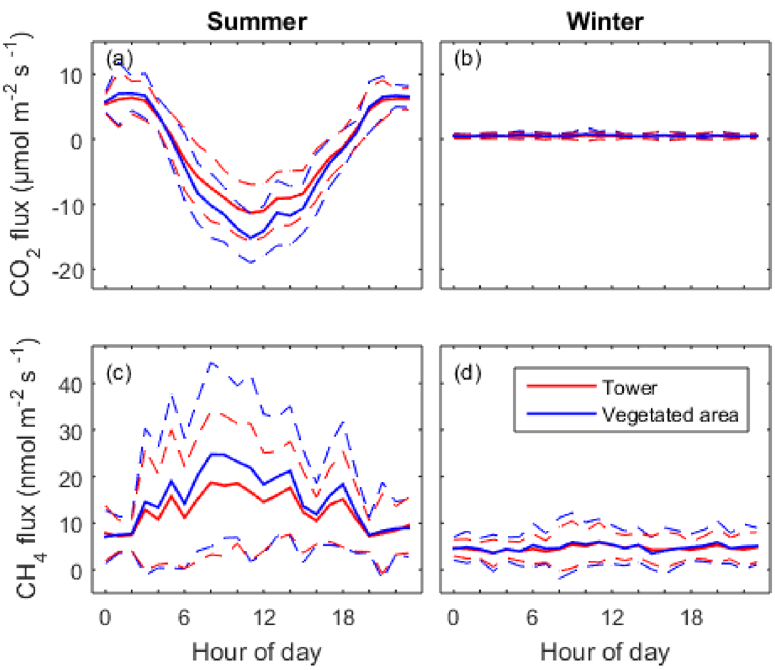


Figure 6~~5~~: Mean diel pattern of the half-hourly net $CO_2$ and $CH_4$ fluxes in summer ((a) and (c)) and in winter ((b) and (d)). The dashed lines represent the standard deviation. Red lines indicate measurement from the EC tower and the blue lines show the fluxes modelled for the vegetated area.

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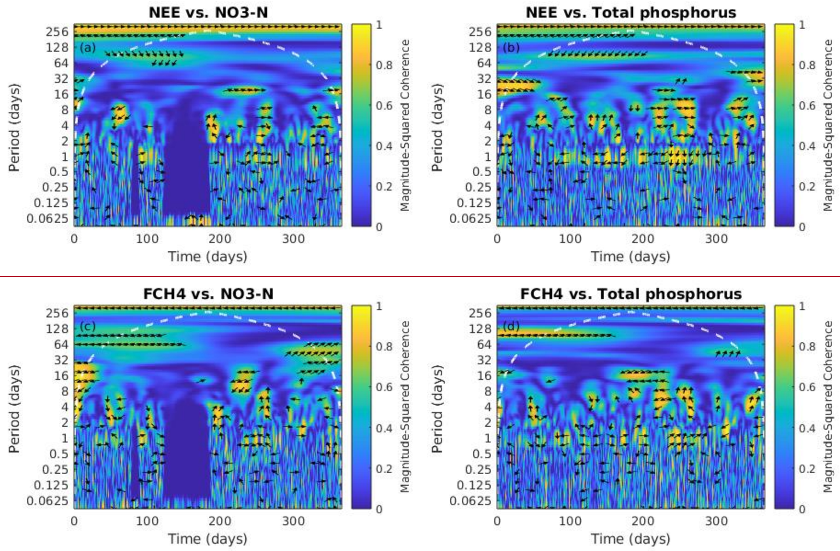


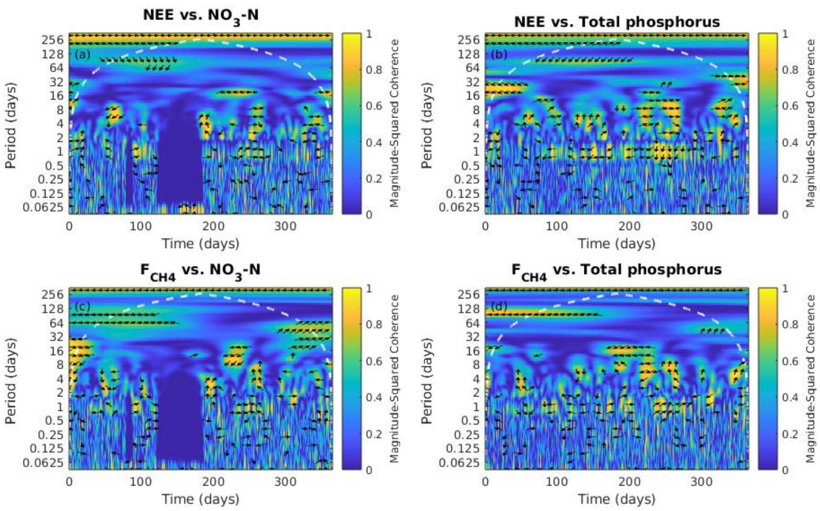


Figure 7: Wavelet coherence analysis and the phase difference between ecosystem fluxes, the net ecosystem exchange (NEE) and the $CH_4$ flux ($F_{CH4}$), and nutrient concentration in the water, $NO_3$-N and total phosphorus from January to December 2013. The color represents the power of the coherence from 0 to 1. The phase difference is indicated by black arrows which only show up where the coherence is greater than or equal to 0.5. → indicates in-phase (two time series in synchrony) and arrows in other direction indicate out of phase (representing lags between time series), i.e. ← indicates

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anti-phase, ↓ indicates the 1st series (fluxes) leads by quarter-cycle and ↑ indicates 2nd series ($NO_3$-N and total phosphorus) leads by quarter-cycle. White dash contour lines indicate the cone of influence.

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