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Partitioning net ecosystem carbon exchange of native and invasive plant communities by vegetation cover in an urban tidal wetland in the New Jersey Meadowlands (USA)

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A B S T R A C T

Due to increased loss of wetland areas globally, restoration of wetlands has become common practice and is applied to increase wetland areas as well as improving ecological services of existing wetlands. Although the importance of wetlands in carbon sequestration has been recognized, the effects of restoration activities on CO_2 release and uptake are still unknown. We measured net-ecosystem exchange (NEE) in two sites, a natural site and a restored site, at the New Jersey Meadowlands, an urban tidal wetland, using the eddy-covariance method from 2014 to 2016. These data were used to compare the two sites and understand the effects of restoration on NEE. A procedure was developed to partition measured fluxes between vegetation types, using footprint modeling and light response curves. Following the partitioning, we compared CO_2 fluxes from invasive and native wetland vegetation communities. Separating flux data into vegetation communities and to seasonal and diurnal fluxes revealed patterns in CO_2 fluxes that allowed determining the nature of each vegetation cover as a source or sink for CO_2. Our results show that CO_2 emissions from the restored wetland were significantly higher than the natural wetland. During restoration, the invasive wetland species *Phragmites australis* was replaced by the native *Spartina alterniflora*, which was then found to have an increased CO_2 uptake during summer days, but also higher CO_2 emissions during nights and winters. Therefore, the wetland restoration practice might be responsible for the increased carbon dioxide release. Additionally, allochthonous carbon input from the river to the wetland was found to be larger at the restored site. Thus, during the study, *S. alterniflora* that was introduced to the restored wetland site was serving as a carbon source, releasing CO_2 to the atmosphere.

1. Introduction

Wetlands are one of the most productive ecosystems in the world, having the capacity to sequester and store hundreds or thousands of years' worth of carbon in their soil, often called 'blue carbon' (Chmura et al., 2003; Hansen, 2009; Howard et al., 2017; McLeod et al., 2011; Moreno-Mateos et al., 2012; Sutton-Grier and Moore, 2016; Woodward and Wui, 2001). With the increasing probability of these systems to be exposed to disturbances, either through human activity or with relation to climate change, there is a higher risk that they will not only lose their ability to continue sequestrating carbon each year, but possibly also shift from a net sink to a net carbon source, and start releasing the stored carbon (McLeod et al., 2011). Due to continuous loss of wetland areas, with a loss of 50% of all the world's wetlands in the twentieth century alone (Russi et al., 2013), construction, restoration and management of wetland areas have become common practices in the last decades, in an effort to save this valuable resource and the services that it provides (Erwin, 2009). Coastal wetlands, especially in urban areas, are more affected by human activities such as construction and development that lead to changes in wetland functions. In this context, restoration activities (also referred to many times as mitigation), are done by changing the wetland hydrology, eradicating invasive vegetation species and introducing native species to the wetland. These efforts are done to restore these ecosystems to function as previous natural wetlands (Zedler and Kercher, 2004). However, still much is unknown about the effects of coastal wetlands management on carbon fluxes (Zedler and Kercher, 2005).

In the last two decades the eddy-covariance (EC) method has become widely used to measure net ecosystem exchange (NEE) of CO_2 in wetlands, mostly in peatlands (e.g. Neumann et al., 1994; Aurela et al., 1998; Joiner et al., 1999; Aenrth et al., 2002; Lafleur et al., 2003; Corradi et al., 2005; Glenn et al., 2006; Kutzbach et al., 2007; van der

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Molen et al., 2007), but also in marshes (Bonneville et al., 2008; Guo et al., 2009; Schäfer et al., 2014). The study presented here was performed in the New Jersey Meadowlands that consist of predominantly tidal brackish marsh occurring along the Hackensack River in northeastern New Jersey. This area covers part of the Hudson-Raritan estuary ecosystem and is comprised of about 35,000 ha of wetlands. It is located within an urban environment and has a long history of disturbances and alterations of tidal flow via development, construction of dams, tide gates and the New Jersey Turnpike. Although some EC studies have been previously performed in restored wetlands, these wetlands were usually treated by restoring water level via flooding of areas that have been previously drained (Herbst et al., 2012; Knox et al., 2015; Koch et al., 2014; Matthes et al., 2014; Petrone et al., 2001). The restoration at the New Jersey Meadowlands is aimed at increasing the productivity of existing wetlands not only by changing their hydrology, but also by actively replacing invasive *Phragmites australis* that dominates the coastal wetlands of eastern North America, with the native wetland species *Spartina alterniflora*. Previous EC measurements in this area have shown large differences of NEE between years following restoration activities (Schäfer et al., 2014). In this study, we revisit the same restored site a few years later. To better understand the effects of these restoration activities on CO_2 fluxes, we compare it with new measurements in a natural untouched nearby site that is dominated by *P. australis* and experiences the same meteorological and environmental conditions as the restored site.

In most EC measurements over wetlands, it is challenging to fulfill all the theoretical requirements of eddy covariance or follow the guidelines provided by the eddy-flux community (Aubinet et al., 2012; Baldocchi et al., 2001) due the heterogeneous nature of wetlands and fetch limitations. Therefore, a compromise should be made between the flux tower height, the tower location (challenging by itself to construct over the wetland soil) and the coverage area of the flux footprint that includes different patches of vegetation species, mudflat and open water at different times. Previous studies have used a flux footprint model to interpret how spatial heterogeneity contributes to the variability in EC fluxes in a wetland (e.g. Sachs et al., 2010; Forbrich et al., 2011; Matthes et al., 2014; Morin et al., 2017). Since the source area that contributes to the measured eddy covariance flux changes with wind direction, EC measurements combined with footprint modeling can be used to partition NEE dynamics of different vegetation covers within the tower footprint area. The motivation of this study is therefore (i) to better understand the effects of the restoration activities at the New Jersey Meadowlands on NEE of CO_2 over different timescales of days, seasons and years, (ii) to assess how it is compared with eddy covariance fluxes over a natural site, and (iii) to partition measured fluxes by vegetation type using footprint analysis and compare carbon fluxes from invasive vs. native wetland vegetation communities.

2. Materials and methods

2.1. Study sites

Eddy covariance flux towers were placed at two sites. The first study site is in the Marsh Resource Meadowlands Mitigation Bank (MRMMB) located in Carlstadt, New Jersey (40.48N, 74.04W). This site (see Fig. 1) is 83 ha in size and was mitigated in two phases in 1999 and 2001. During the mitigation, standing *P. australis* was removed along with the top layer of the soil. It was then ground up, and mixed with site mud and compressed on top of a coconut mat so that *S. alterniflora* could be planted. The site has been actively managed following the mitigation campaigns by herbicide spraying to prevent invasive *P. australis* from returning. This has been done only in parts of the originally mitigated area, so that as of today, some of the area is repopulated with *P. australis* and a mix of other low marsh vegetation. The EC tower is located within the mitigated part at the interface between the two areas, so that the footprint of the tower captures both, an area that is comprised only of *S. alterniflora*, and an area that includes a mix of different types of low marsh vegetation (Fig. 1). The second study site, Hawk property (HP) is located in Secaucus, Hudson County, on the eastern bank of the Hackensack River (40.77N, 74.09W), about 6 Km downstream from the MRMMB site. This is an unmitigated, natural wetland area dominated by *P. australis* with some patches of native *Spartina patens*. The location of the eddy covariance flux tower was selected at the transition area between *P. australis* and the largest patch of *S. patens* so that the footprint captures both types of vegetation (Fig. 1).

2.2. Data collection and processing

Data were continuously measured in both sites for 3 years, from January 2014 to December 2016. The towers monitor turbulent fluxes by continuously measuring three-dimensional wind velocity and virtual temperature with an ultrasonic anemometer (CSAT3, Campbell Scientific Inc., Logan, UT) and CO_2 and H_2O concentrations with an open path infrared gas analyzer (LI-7500A, LI-COR Biosciences, Lincoln, NE) at a frequency of 20 Hz. The sensors are located on a tower 4.3 m and 3.3 m above the surface at MRMMB and HP respectively, as a compromise to include a large source area without the footprint extending outside of the wetland area for most the time.

Additionally, meteorological measurements were collected continuously and stored on a datalogger (CR3000, Campbell Scientific Inc., Logan, UT) once every 30 min, including net radiation (CNR1 in MRMMB and CNR4 in HP, Kipp & Zonen, Delft, Netherlands), air temperature, vapor pressure and air humidity (HMP45C, Vaisala, Helsinki, Finland). The meteorological half-hourly data were filtered by removing measurements above and below a plausible range (see Supplementary material).

The collected high-frequency data were filtered and despiked using standard EC processing methods. The wind velocity measurements were rotated using the planar-fit method (Wilczak et al., 2001) and the Webb-Pearman-Leuning correction for open-path instruments (Webb et al., 1980) was applied. Average wind velocity, variances, Reynolds stresses, friction velocity (u_*), and scalars fluxes were calculated in a 30 min window interval. A spectral correction was then used (Horst, 1997). The time-averaged data were despiked again according to Papale et al. (2006), and u_* filter for nighttime data was applied. The data processing is described in detail in the Supplementary material. The total available 30 min intervals remaining after despiking and filtering was similar between sites and years (about 6000–8000 half-hour intervals per year-site except for MRMMB 2016 for which ~4000 half-hour intervals remained, see Table 1).

2.3. Footprint analysis

To assess how much data needs to be filtered due to fetch limitations, a 2-D footprint model by Detto et al. (2006) was used, which is based on the 1-D model by Hsieh et al. (2000). The footprint model uses the friction velocity, the boundary layer stability (Obukhov length) and the tower and vegetation height, and estimates the contribution from all upwind emissions to a measured flux at the sensor height at each available half-hour measurement. This footprint function is then rotated to the direction of the wind and the integrated flux-footprint probability for each patch type within the wetland is calculated, using vegetation maps. These maps were generated using remote-sensing images (Google Earth) and represent 3 classes for HP (*P. australis*, *S. patens* and water/mudflat), and 4 classes for MRMMB (*S. alterniflora*, low-marsh mixed vegetation, water/mudflat and highway). The footprint analysis based on these maps is later used to partition the flux originating from each vegetation cover area. We used simplified vegetation maps rather than other available hyper spectral maps (e.g. from the Meadowlands Environmental Research Institute) to reduce the number of classes so that we aggregate as many measured half-hour intervals of each vegetation cover type to estimate its representative