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Soil greenhouse gas fluxes in tropical mangrove forests and in land uses on deforested mangrove lands

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A B S T R A C T

Mangrove forests are important carbon sinks in the tropics, yet tropical mangrove deforestation and land use conversion still persists. Reporting of greenhouse gas (GHG) emissions from natural and anthropogenic sources in wetlands are important in regional and national emissions inventories. However, very few studies have been conducted to measure on the GHG fluxes in coastal wetlands, particularly in mangrove forest and non-forest land uses in deforested mangroves. We investigated the soil fluxes of CO_2, CH_4 and N_2O in mangrove forest and non-forest land uses on deforested mangrove areas (i.e. abandoned aquaculture ponds, coconut plantations, abandoned salt ponds, and cleared mangroves) in the coasts of Honda Bay, Philippines. Results showed that the emissions of CO_2 and CH_4 were higher by 2.6 and 6.6 times in mangrove forests (110 and 0.6 kg CO_2e ha $^{-1}$ day $^{-1}$, respectively) while N_2O emissions were lower by 34 times compared to the average of non-forest land uses (1.3 kg CO_2e ha $^{-1}$ day $^{-1}$). CH_4 and N_2O emissions accounted for 0.59% and 0.04% of the total emissions in mangrove forest as compared to 0.23% and 3.07% for non-forest land uses, respectively. Site-scale soil GHG flux distribution could be mapped with 75% to 83% accuracy using Ordinary Kriging. Unlike mangroves that can offset all GHG emissions through CO_2 uptake from photosynthesis, the non-forest land uses cannot offset their emissions on-site as they are usually devoid of vegetation. Our results could be utilised in higher tier national GHG inventories, to refine regional and global estimates of GHG emissions in mangrove wetlands, and improve policy on coastal wetlands conservation.

1. Introduction

Mangrove forests are important ecosystems inhabiting the coastal wetland of tropical and subtropical countries. They provide timber and other construction materials, fuelwood, fishery products and performs critical ecological functions such as biodiversity conservation, storm protection, sediment regulation, and coastal stabilisation, among others (Koch et al., 2009; Barbier et al., 2011; Salmo et al., 2013). Recently, mangroves have been increasingly recognised as among the most carbon dense tropical forests (Murdiyarso et al., 2010; Kauffman et al., 2011; Adame et al., 2013; Murdiyarso et al., 2015; Bhomia et al., 2016). Southeast Asian countries hold > 30% of the remaining mangroves in the world, estimated to be between 13,776,000 and 15,236,000 ha (Spalding et al., 2010; Giri et al., 2011).

On a global scale, approximately 3.6 million ha of mangroves had been deforested and converted to other land uses since 1980. However, deforestation rate has declined to 102,000 ha yr^{-1} during the period 2000–2005 from 185,000 ha yr^{-1} in the 1980s (FAO, 2007). In Southeast Asia, recent estimates of mangrove deforestation and land use conversion were about 9535 ha per year during the period 2000–2012, mostly to make way mostly for aquaculture, rice farms and oil palm plantations (Richards and Friess, 2015).

Overexploitation, conversion of mangroves to aquaculture, agriculture, urban, tourism and industrial developments in the coastal area are considered to be among the drivers of global mangrove loss (Alongi, 2002; Giri et al., 2011; Murdiyarso et al., 2013; Richards and Friess, 2015). In the Philippines, for example, conversion of mangroves to aquaculture ponds is considered the main reason for mangrove loss. Other causes are overexploitation and conversion to other non-forest land uses such as agriculture, salt ponds and settlements (Primavera and Esteban, 2008). Mangrove clearing and land reclamation results in the emission of stored carbon (C) in the form of CO_2 and other GHG through oxidation (Lovelock et al., 2011). Also, aquaculture and agriculture add nutrients to the system which can enhance the metabolism

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of soil microorganisms resulting in the emissions of N_2O and CH_4 (Chen et al., 2010).

Carbon dioxide, CH_4 and N_2O are the three main GHGs being monitored in the land use sector by the International Governmental Panel on Climate Change since 1990 (IPCC, 2013). Accurate reporting of GHG emissions from wetlands is, therefore, essential for regional and national emissions inventories. Compared to tropical peatlands (Hadi et al., 2005; Hergoualc'h and Verchot, 2011; Hergoualc'h and Verchot, 2012), the effects of mangrove land use conversion on soil fluxes of the three GHGs are not yet fully understood. For mangroves, the soil GHG fluxes could be minimal to substantial depending on anthropogenic influences (Chen et al., 2010; Chen et al., 2014). Several soil and vegetation characteristics (referred to in this paper as environmental site variables) are reported to influence the GHG fluxes in mangrove soil. For instance, CO_2 fluxes are related to soil organic matter content, soil moisture content, redox potential, salinity and porosity (Chen et al., 2010; Chen et al., 2014), and Leaf Area Index (Lovelock, 2008). CH_4 fluxes in wetlands are related to salinity (Purvaja and Ramesh, 2001; Allen et al., 2011; Poffenbarger et al., 2011) whilst N_2O flux is influenced by nitrate loading, salinity and porosity (Allen et al., 2007; Chen et al., 2010; Howard et al., 2014). The soil fluxes of the three GHGs, however, have not been measured simultaneously in non-forest land uses in deforested mangroves along with the mangrove forest that they replaced. Fluxes of CH_4 and N_2O have not been assessed for aquaculture ponds, salt ponds and coconut plantations that are formerly occupied by mangroves as well as in mangrove areas that were cleared of vegetation. Simultaneous measurements of the three GHG fluxes are necessary to evaluate their relative importance (Cobb et al., 2012).

In addition, site-scale information on the spatial variation of soil GHG fluxes is needed in order to identify priority areas for management intervention. Available studies on GHG fluxes in coastal wetlands are based on few plots from which the spatial variation in the entire study site is inferred. Maps of soil GHG fluxes produced from modelling the spatial variation of the soil GHG fluxes are, therefore, essential for designing effective programs aimed at reducing GHG emissions from soil.

The broad aim of our study was to evaluate how deforestation and land use change in forested coastal wetland affect the soil fluxes of CO_2, CH_4 and N_2O. We investigated the fluxes of these three GHGs under mangrove forests and compared with four non-forest land uses that replaced mangroves (i.e. abandoned aquaculture ponds, coconut plantations, abandoned salt ponds and cleared mangroves). Specifically, the study aims to: 1) quantify the fluxes of CO_2, N_2O and CH_4 in mangrove forests and non-forest land uses that replaced mangroves; 2) determine the relationship of these GHGs with selected environmental site variables; and 3) evaluate the accuracy of the output from GIS-based spatial modelling/interpolation of site-scale variation of GHG fluxes.

2. Methods

2.1. Site description

The study was conducted in July 2015 in the southern coast of Honda Bay, eastern coast of Puerto Princesa City in the island-province of Palawan, Philippines (between latitude 9.8028°N to 9.9612°N and longitude 118.725°E to 118.805°E; Fig. 1). The city is about 567 km south-west of Manila, the Philippines' capital. It has a tropical climate, wet (> 115 mm) from May to December and dry (< 55 mm) from January to April. The site receives a mean annual rainfall of 1527 mm. February and March are the driest months with < 40 mm rainfall while October and November are the wettest, with 216 and 211 mm of rain, respectively. The city's annual mean temperature is 27.4 °C. The lowest temperature (26.8–26.9 °C) is during the months of January and February while the highest (28.5–28.6 °C) is during April and May (PAGASA, 2016).

The southern coast of Honda Bay is lined with a contiguous mangrove forests, ca. 40 km in length, running north to south of the study site, along the coast (fringing mangrove), and along the mouths and upstream (estuarine mangrove) of the three rivers in the northern part of the study area. The fringing mangroves are interspersed with non-forest land uses such as agriculture, aquaculture, and built-up areas/settlements especially in the central and southern portion of the study site. The non-forest land uses mostly encroached the landward and middle portions of the fringing mangroves in the case of aquaculture, agriculture and built-up areas. However, the encroachment is occasionally within the whole band (landward to seaward) of mangroves, disrupting the north-south forest lining the shore. Such is the case of local boat stations, clearings for access and some human settlements. The width of the remaining fringing mangrove forests varies from 10 m to about 500 m while the width of the estuarine mangroves is from 250 m to 560 m at the river mouth to some 90 m to 200 m upstream. The estuarine mangroves extend ca. 1 km to about 6 km upstream from the river mouth. The mangroves in the northern portion have closed canopies (here termed as 'closed canopy forest'), while the mangroves in the central and southern portions, where the non-forest replacement land uses of mangroves are all located, mostly have open canopies (here termed as 'open canopy forest').

For the measurement of soil GHG fluxes, selected sites were selected under mangrove forests and four other land uses on deforested mangrove lands (i.e. competing land uses of mangroves) such as coconut plantations, abandoned aquaculture ponds, abandoned salt ponds and cleared mangrove areas. The study site has an area of 2750 ha, of which some 1216 ha is covered by mangrove forests, both under closed and open canopies (Fig. 1).

The mangroves in the site are dominated mostly by *Rhizophora* tree species. They are classified into two canopy types: closed canopy mangrove forest and open canopy mangrove forest. Closed canopy mangroves are dense, intact mangrove vegetation, with no significant open spaces or gaps. These forests are located mostly in areas far from built-up areas. They have a mean Leaf Area Index (LAI) of 2.25 and canopy gap fraction of 14% or about ~86% canopy foliage cover. On the other hand, open canopy mangroves have open spaces, with fewer trees, and mostly near built-up areas, with mean LAI of 0.62 and canopy gap fraction of 64% (~36% canopy foliage cover).

The non-forest land uses were historically occupied by mangroves prior to their conversion. The aquaculture ponds and salt ponds were mangrove forests until they were cleared in the early 1990s. These aquaculture and salt ponds were in operation until the early 2000s. The coconut plantation is a 20-year-old stand planted at the back of an open canopy mangrove forest and seemed not actively managed as there were thick growth of the mangrove fern *Acrostichum* sp. (see Fig. 2d). The cleared mangrove is a deforested mangrove area that was gradually cleared from 2005 to 2008 as a resettlement site but remained unutilised. A summary of the characteristics of the land uses of the study sites is shown in Table 1.

2.2. Field sampling design

Mangrove forests (represented by closed canopy mangrove and open canopy mangrove forests), along with non-forest land uses in deforested mangrove lands (represented by abandoned aquaculture ponds, coconut plantations, abandoned salt ponds and cleared mangroves), were used in this study. Each of these land uses (i.e. mangrove forests and replacement land uses) comprised six sites which were randomly selected. For mangrove forests, three sites were selected representing the closed canopy forests and another three sites representing the open canopy forests. For the replacement land uses, three sites were selected representing the abandoned aquaculture ponds, and another three sites representing the non-aquaculture land uses. Due to access restriction to other coconut plantations in the site, and absence of other abandoned salt ponds and cleared mangroves in