

Dynamics of Methane

Subjects: Forestry | Soil Science | Environmental Sciences

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Mangrove forests sequester a significant amount of organic matter in their sediment and are recognized as an important carbon storage source (i.e., blue carbon, including in seagrass ecosystems and other coastal wetlands). The methane-producing archaea in anaerobic sediments releases methane, a greenhouse gas species. The contribution to total greenhouse gas emissions from mangrove ecosystems remains controversial. However, the intensity CH₄ emissions from anaerobic mangrove sediment is known to be sensitive to environmental changes, and the sediment is exposed to oxygen by methanotrophic (CH₄-oxidizing) bacteria as well as to anthropogenic impacts and climate change in mangrove forests. This review discusses the major factors decreasing the effect of mangroves on CH₄ emissions from sediment, the significance of ecosystem protection regarding forest biomass and the hydrosphere/soil environment, and how to evaluate emission status geospatially. An innovative “digital-twin” system overcoming the difficulty of field observation is required for suggesting sustainable mitigation in mangrove ecosystems, such as a locally/regionally/globally heterogeneous environment with various random factors.

Keywords: geospatial ; greenhouse gas ; carbon storage

1. Introduction

The carbon (C) sequestered in the biomass and deep sediment of vegetated coastal ecosystems, including mangroves, seagrass beds, and tidal marshes, has been called “blue carbon” [1][2]. Although vegetated coastal habitats cover a relatively small area (<2%) of the coastal ocean, they have C burial rates that are 40 times higher than tropical rainforests and account for more than half of the C burial in marine sediment [3].

Although the global area of vegetated ecosystems is one to two orders of magnitude smaller than that of terrestrial forests, the contribution of vegetated coastal habitats per unit area to long-term C sequestration is much greater, which is in part because of their efficiency in trapping suspended matter and associated organic C during tidal inundation [4]. Among vegetated ecosystems, mangroves have been well highlighted as among the major sources of organic matter in tropical areas because they occupy a large part of the tropical coastal area [4]. Additionally, organic C production is more rapid in these areas than for other estuarine and marine primary producers [3][5].

Mangrove forests have gained attention because of their high C productivity [6][7] and because they are among the most C-rich ecosystems in the world [8][9]. The total net primary production of mangroves is approximately 200 Tg C year⁻¹ [10][11], but most of this C is lost or recycled via CO₂ flux to the atmosphere (34.1 Tg C year⁻¹, ~20%) or is exported as particulate organic C, dissolved organic C, and dissolved inorganic C to the ocean (117.9 Tg C year⁻¹; ~60%) [11][12]. Of the remaining C, burial accounts for 18.4 to 34.4 Tg C year⁻¹ [1][9][10][13][14], and this blue C is considered to represent a significant long-term storage of atmospheric CO₂ [13][15]. The global C sequestration rate in mangrove wetlands is 174 g C m⁻² year⁻¹, on average, corresponding to about 10% to 15% of global coastal ocean C [9]. Organic-rich soils dominate in mangrove C storage, accounting for 49% to 98% of C stocks in mangrove wetlands [8][16].

Global mangroves are mainly distributed along tropical and subtropical coastlines, covering 137,760 km². The world's largest mangrove areas are in low latitudinal regions, such as Indonesia (22.6% of the global total), Australia (7.1%), and Brazil (7.0%) [17]. The world's best developed mangrove forests can be found in the Sundarbans, the Mekong Delta, the Amazon, Madagascar, and Southeast Asia [17]. Furthermore, Indonesia has the highest mangrove species diversity (48 species [15]) and exceptionally high C stocks in mangrove sediment [15]. Because the economic/population growth in those area is also substantial, the loss of mangrove forest due to anthropogenic impacts is substantial globally [15][18][19]. Loss rates vary greatly between countries, ranging from 1% to 20% of the total mangrove forest area, so predicting global mangrove forest changes in the future is difficult [20]. Loss of mangroves by clearing, conversion to industrial

estates/aquaculture, and changes in drainage patterns lead to striking changes in soil chemistry and usually result in rapid emission rates of greenhouse gases [21][22][23].

2. Methane Flux from Mangrove Forests

2.1. Significance of Methane Emission from Mangrove Forests

The magnitude of CH₄ flux in mangrove forests and its relative contribution to global warming compared to CO₂ flux remains controversial. The global scale practice of the mangrove C budget has shown that CH₄ emissions from soil are 2 Tg C year⁻¹ [9]. Considering its global warming potential, the contribution of CH₄ emissions is comparable to the above-mentioned rate of C burial (18–34 Tg C year⁻¹) and C emission by soil respiration (34 Tg C year⁻¹) [1][9][10][13][14]. Recent studies have reported a significant amount of CH₄ flux from mangrove sediment [24][25][26][27][28][29][30][31] and have claimed that the contribution of CH₄ flux to global warming was non-negligible in estuarine mangrove forests, which could account for 18% to 22% of blue C burial rates [13] and 9% to 33% of plant CO₂ sequestration [32]. However, observed CH₄ flux from mangrove soils is mostly negligible compared to CO₂ emissions from sediment but is highly variable [33][34][35][36][37][38][39][40][41][42][43][44], particularly for non-polluted mangrove sediment [21][31][45][46][47][48]. To confirm this observation, the authors compared incubation experiments with mangrove sediment collected from the Vietnamese Mekong delta and the Indian Sundarbans forest [34]. CH₄ production was equivalent to only 0.05 to 0.27% of the CO₂ production under aerobic incubation or 0.05 to 0.22% under anaerobic incubation conditions, even when considering the potential difference caused by ($n = 30$ in each incubation experiments).

2.2. Factors Associated with Methane Emission

2.2.1. Soil Conditions

Low CH₄ production and emission in mangrove sediment compared to in interior wetland soils is mainly due to the high presence of sulphate in mangrove sediment, which allows sulphate-reducing bacteria to outcompete CH₄-producing archaea (i.e., methanogens) [35][41][42][43]. However, soil salinity and sulphate concentration show a low negative relationship with methane-producing activities, which suggests that both forms of methanogenesis are not completely inhibited by sulphate reducers with increasing sulphate concentrations [41]. A significant increase in CH₄ production activity caused by the dilution of seawater was also reported [34]. Therefore, mangrove sediment CH₄ production activity is highly and non-linearly sensitive to its specific soil pH/electrical conductivity by being affected by different freshwater intrusion intensities. Despite few studies on the impact of freshwater intrusion on CH₄ emission, it is still important to evaluate because rice paddies/agricultural fields are often found adjacent to protected mangrove zones (**Figure 1**). Regardless of a significant correlation between salinity/sulphate concentration in sediment and CH₄ emissions [34][41][49] CH₄ production activity can be significantly increased by the dilution of seawater concentration [34].



Figure 1. Reforestation zone in Sundarbans mangrove area in India (a), protected mangrove forest adjacent to rice paddies (b), and vegetable-growing field adjacent to mangrove forests (c) in Soc Trang, Vietnam.

Another reason for decreased CH₄ production is that compared to herbaceous organic matter, woody organic matter derived from mangrove trees is relatively recalcitrant to methanogens using it as a substrate [44]. Additionally, mangrove ecosystems are inundated by irregular periodic tides affected by the tidally mediated exchange of porewater between sediment and surface water via the ebb and flow of tides [10][45][46]. The tidally mediated exchange of porewater between sediment and surface water occurs via the ebb and flow of tides (i.e., tidal pumping [50][51][52][53]). Tidal pumping is a potential source of solutes to mangrove water and causes ebullition, but the process has only recently been quantified and directly linked to the export of C and nutrients [53][54]. In addition to the spatio-temporal heterogeneity caused by irregular tidal pumping, CH₄ flux is spatio-temporally heterogeneous and is highly variable because of the heterogenic spatial distribution of aerial mangrove tree roots [55] and burrows created by crabs/goby fish. Such activity enhances hydraulic connectivity and increases the surface area of the sediment–water interface [56] where the exchange of the by-products of subterranean respiration can occur during tidal inundation [57][58][59][60][61][62]. Owing to the difficulty of

observing CH₄ flux precisely, CH₄ data for mangrove forests are limited compared to data observing interior wetlands and underestimate the global emission [13][46].

2.2.2. Methanogenic and Methanotrophic Communities

Although CH₄ flux micrometeorological observation data are limited [62], recent studies on the community structures of methanogens and methanotrophs have revealed the biological processes common to interior wetlands and as unique characteristics in coastal wetlands. Previous studies on CH₄ metabolism have indicated that CH₄ emission in natural ecosystems is largely driven by microorganisms, especially methanogens and methanotrophs [63][64][65][66][67]. Highly diverse methanogenic and methanotrophic communities can promote CH₄ production and oxidation [68][69]. However, different types of methanogens and methanotrophs have preferable growing conditions, which further affect CH₄ emissions in natural ecosystems [64][70][71]. Methanogens include hydrogenotrophic, acetoclastic, and methylotrophic methanogens [71][72]. Methanotrophs exist under both aerobic and anaerobic conditions. Aerobic methanotrophs are phylogenetically divided into two main groups: type I (*Gammaproteobacteria*, e.g., *Methylococcaceae*) and type II (*Alphaproteobacteria*, e.g., *Methylocystaceae*) [73][74][75][76], nitrate- or nitrite-dependent [77][78] and metal-dependent [79] CH₄ oxidizers, respectively. Type I methanotrophs tend to be dominant in natural environments with sufficient nutrients and substrates (i.e., relatively high O₂ concentration, low CH₄ concentration) [34][44][80], whereas type II methanotrophs tend to be abundant in resource-limited environments with a high affinity for their nutrients and substrates (i.e., relatively low O₂ concentration, high CH₄ concentration) [71][74][81][82]. Of note, methanogens and methanotrophs in coastal wetland soils have unique characteristics that are rarely found in interior wetland soils. Hydrogenotrophic and acetoclastic methanogens are considered dominant in natural freshwater wetland soils. However, methylotrophic methanogens are dominant in hypersaline and sulphate-rich environments including coastal wetlands, and they make different contributions to CH₄ production [69][83][84]. In coastal wetlands, anaerobic methanotrophs include sulphate-dependent methanotrophs [34][85], which might have an important role in controlling low coastal CH₄ fluxes. Furthermore, several reports have described the possibility of active CH₄ production under aerobic conditions in mangrove forests based on laboratory incubation experiments and field observations [34][86].

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