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Carbon sequestration in wetlands, from science to practice: An overview of the biogeochemical process, measurement methods, and policy framework

Jorge A. Villa[a], Blanca Bernal[b,*]

[a] Facultad de Ingeniería, Grupo de Investigación GAMA, Corporación Universitaria Lasallista, Carrera 51 no. 118 sur—57, Caldas, Antioquia, Colombia
[b] Smithsonian Environmental Research Center, 647 Contees Wharf Rd, Edgewater, MD 21037, USA

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ABSTRACT

Ecosystem services are becoming increasingly important and a reason to promote the sustainable use of natural resources. Wetlands provide many valuable ecosystem services, including carbon (C) sequestration. Wetlands are an important C sink, playing a key role in climate regulation. As such, their ability to sequester C is being considered in national GHG emissions assessments and private initiatives as a potential source of revenue to manage carbon-balanced landscapes and pay for ecosystem services. To be able to implement these initiatives widely some aspects of wetland carbon science and practice still need to be formalized and standardized. Here we synthesize the scientific basis of the biogeochemical processes that drive C sequestration in wetlands and assess the methods available for its measurement. We have reviewed data in 110 peer-reviewed studies form wetlands around the world and provide an overview of the current policies and guidelines in which C sequestration in wetlands is framed as an ecosystem management practice. The intention of this review is to provide a wide and comprehensive summary of C sequestration in wetlands, from science to practice. This analysis can help inform practitioners and landscape managers in future considerations regarding project design and policy implementation, improve current climate mitigation schemes and payment for ecosystem services frameworks, and foster the worldwide implementation of wetland restoration, creation, and conservation projects for sustainable development and climate change mitigation and adaptation.

1. Introduction

Since the Millennium Ecosystem Assessment (2001–2005) and The Economics of Ecosystems and Biodiversity Initiative (2008–2010) evaluated the consequences of ecosystem changes on human well-being, ecosystem services have been considered increasingly valuable and a reason to promote the preservation and sustainable use of natural resources. Before that, Costanza et al. (1997), Pimentel et al. (1997), Daily (1997), de Groot et al. (2002), and Patterson (2002), among others, analyzed the value of the services provided to society by several ecosystems using environmental valuation techniques and ecological pricing methods (Hansen, 2009; Ninan and Inoue, 2013). These studies pointed out the highly valuable benefits that ecosystems provide to humans, and the costs that replacing those services by anthropogenic means would entail. In addition, Day et al. (2009) emphasized the increasing importance of ecosystem services in supporting human economy as energy becomes scarcer and, consequently, the pressing need to manage ecosystems sustainably. About 10 years after the Millennium Ecosystem Assessment, countries agreed in the UN Sustainable Development Summit (2015) to adopt 17 Sustainable Development Goals to promote a global action plant for sustainable development through ecosystem conservation.

International and non-governmental organizations have produced guidelines to account for these services. However, defining these services and assigning them a tradeable value remains challenging (Vo et al., 2012; Muradian et al., 2013). This challenge exists because (1) a value for an ecosystem function is significantly more difficult to assign than a value for its structural attributes (Alongi, 2011); (2) the appropriate indicators or metrics to evaluate functions are as challenging to select as the appropriate methodologies to quantify them (Meyerson et al., 2005; Alongi, 2011; Robinson et al., 2012); (3) there are no sufficient data, nor is there a proper framework to analyze it (Ringold et al., 2013); (4) the value of the

* Corresponding author.
E-mail address: bernalb@si.edu (B. Bernal).

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service depends on its scale and magnitude in each specific geographical location (Meyerson et al., 2005); and (5) uncertainties rise as one projects further into the future since many ecosystem services reflect non-linear processes (Barbier, 2009; Chisholm, 2010).

The determination of ecosystem services in wetlands can be further challenging, as these are dynamic ecosystems that have characteristics of both mature and immature ecosystems and thus the current state of a wetland may not reflect its future trends or conditions (Odum, 1969; Mitsch and Gosselink, 2015). Moreover, scarce adequate and up to date records of wetlands around the world make it difficult to track the changes in their quality and quantity (Zedler and Kercher, 2005). The loss and the degradation of wetlands imply that they are no longer able to perform their ecosystem functions, and the services they provide are thereby lost. This is not a small loss; despite of covering a small percentage of the land's surface (between 3.2%–9.7%; Melton et al., 2012), several studies report that wetlands are significant carbon sinks that store about one-third of the organic soil C pool of the world (Gorham, 1991; Lal, 2008), and one of the most valuable ecosystems in terms of the services they provide (Costanza et al., 1997; Millennium Ecosystem Assessment, 2005; de Groot et al., 2006).

C sequestration is the wetland service that has received most attention in recent years in the climate scenario, due to scientific evidence on the role that wetlands play in the radiative balance of earth's atmosphere (e.g. Roulet et al., 2007; Sjögersten et al., 2014), and to the emergence of policies introducing mechanisms to integrate C sequestration into climate change adaptation schemes and ecosystem conservation initiatives (Lovelock and McAllister, 2013; Aziz et al., 2016). However, the role that wetlands play as C sinks varies widely depending on the wetland hydrogeomorphic type and landscape position (e.g. Bernal and Mitsch, 2012; Bernal and Mitsch, 2013b). Furthermore, even though wetland ecosystems sequester C in their soil, their actual C sink capacity depends on the net balance of its C fluxes. Methane (CH_4) emissions particularly, hinder the ability of many freshwater wetlands to function as a net C sink (Bastviken et al., 2011). Coastal wetlands such as salt marshes and mangroves, on the other hand, are able to build a significant soil C pool with little- to no-CH_4 emissions (Poffenbarger et al., 2011), resulting in a consistent C sink. Fig. 1a gives an overview of the C pools of biomes around the world, based on global estimates (see Zedler and Kercher, 2005). The abundance of tropical forests and their high biomass production yields a high biome C stock. Wetlands, on the other hand, represent a vast underground C stock despite of their little acreage. Global estimates of wetland pools (tons C per ha) indicate that boreal peatlands (Gorham, 1991; Jobbágy and Jackson, 2000; Mitra et al., 2005) and coastal wetlands (Mcleod et al., 2011; Pendleton et al., 2012; Hopkinson et al., 2012) are large C sinks (Fig. 1b). The rest of wetland ecosystems (all temperate and tropical freshwater wetlands, as denoted in Fig. 1b) comprise a large C pool, but there are no accurate global estimates per wetland class within this category that can be compared to global C pool estimates of mangroves, saltmarshes, and boreal peatlands.

In this manuscript we present an overview of the biogeochemical process of C sequestration in wetlands (Section 2), an analysis the methods available to measure C sequestration and sediment accretion rates (Section 3), a review of published studies on carbon sequestration and sediment accretion in wetlands of the world (Section 4), a description of the policy framework that sets the ground for the inclusion of wetlands into management practices (Section 5), and an overview of remaining knowledge gaps and future directions (Section 6). The intention of this review is to connect science and practice, to help in overcoming the abovementioned challenges of accounting wetland carbon sequestration as an ecosystem service. Our approach to do so consists on synthetizing the scientific basis of the sequestration process and measurement methods valuable for policy as well as land managers and stakeholders' decision making, and describing current policies and guidelines that frame C sequestration within ecosystem management practices of wetland protection, restoration, and creation.

2. Carbon sequestration in wetlands: process overview

Carbon sequestration in wetlands refers to the carbon dioxide (CO_2) taken from the atmosphere that is transferred and accumulated into the wetlands' soil C pool as soil organic matter (SOM). As SOM accumulates its C is sequestered over time, as a result of the balance between C inputs and outputs. Inputs are constituted mainly by C contained in organic matter (OM) from senesced vegetation in and around the wetland (autochthonous), as well as C dissolved and suspended in inflowing waters and runoff (allochthonous). C outputs can consist of dissolved and suspended organic C in outflowing waters, but the main source are the inorganic forms emitted as CO_2 and CH_4 from the mineralization of OM during decomposition. While high C inputs can lead to high C accumulation, they can also yield high gas emissions. For example, Nahlik and Mitsch (2010) reported that CH_4 emissions were correlated with cumulative aboveground biomass productivity (ANPP) in the same wetlands that Bernal and Mitsch (2013a) found a strong correlation between ANPP and C sequestration in the soil. This indicates that the more C inputs to the soil, the more C to be potentially sequestered as SOM, but also the more abundant is the substrate to be degraded and exported to the atmosphere and to downstream waters (Freeman et al., 2004a). Therefore, C sequestration in wetlands cannot be understood without comprehending the factors that affect C decomposition and mineralization. These factors are described below.

- Substrate availability

The mechanisms that drive OM decomposition include a variety of biological, physical, and chemical processes that interact (Ewing et al., 2006; Baisden and Parfitt, 2007; Rasmussen et al., 2007; Schmidt et al., 2011), favoring the decomposition of easy-to-degrade organic materials by soil microbes (Laanbroek, 2010; Basiliko et al., 2012; de Graaff et al., 2014) and leaving behind to accumulate in the soil the recalcitrant components that microbes cannot further degrade efficiently (Davidson and Janssens, 2006; Bernal et al., 2016). Factors that are known to slow down OM decomposition and favor its accumulation in the soil are: scarcity of nutrients that limit microbial growth (Kuzyakov, 2002; Kuzyakov, 2010), physical protection or organic particles through aggregates (Six et al., 2002), and high content of organic compounds with low degradability (e.g. lignin; Hernes et al., 2007), among others. These physicochemical factors define the bioavailability of organic substrates and thus their recalcitrance.

The process of SOM decomposition is carried out by microbes through microbial growth and metabolism. These microbes obtain energy from the oxidation of organic compounds using electron acceptors in metabolic pathways that follow a predictable sequence based on their availability (Craft, 2001). To oxidize these organic compounds, microbes produce enzymes that aid in the degradation of OM (Kang et al., 2013); the more recalcitrant the compounds, the more necessary the enzymes to degrade them, and the higher the energetic cost for the microbial community to break down recalcitrant compounds into easier-to-degrade units. Microbially-available substrates become thereby scarce when SOM is recalcitrant. In addition to energy and electron acceptors, microbial growth requires nutrients that may not be supplied in enough