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REVIEW ARTICLE

Carbon Sequestration by Wetlands: A Critical Review of Enhancement Measures for Climate Change Mitigation

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Abstract
Wetlands are among the most important ecosystems in the response strategy to climate change, through carbon sequestration (CS). Nevertheless, their current CS potential is declining due to human disturbance, with further decrease expected under global population growth and climate change scenarios. Literature has documented various measures that seek to enhance CS by wetlands and therefore enable these ecosystems remain vital in global carbon (C) balance and climate change mitigation. The objective of this review is to critically analyse these measures with respect to their feasibility and impact on wetland functioning, both in ecological and socio-economic perspectives. In doing this, we strive to address the concerns of wetland scientists, managers and other stakeholders pertaining CS by wetlands. Findings indicate that CS can be enhanced through both non-manipulative and manipulative measures. Non-manipulative measures aim at enhancing CS by increasing wetlands' spatial extent, while manipulative ones aim at altering characteristics of certain wetland components that influence CS. Their overall target is to increase organic matter input, apportion C to longer-lived pools, and increase residence times of C pools. Based on the identified research gaps, we recommend that CS actions for wetlands should prioritize conservation of existing natural wetlands. Additional measures should consider associated risks such as those on wetland flora and fauna, soil and hydrological regimes, and competing services. We further believe that successful implementation of non-manipulative measures for CS will require attachment of economic incentives that are not only foreseeable, but also adequate to match returns from competing land uses.

Keywords Wetlands · Carbon sequestration · Climate change · Carbon dioxide · Methane · Greenhouse gases

1 Introduction

Worldwide, one of the current frequently discussed topics is global climate change, which many climate scientists agree is a result of increased emission of greenhouse gases (GHGs) into the atmosphere, arising from anthropogenic activities. Carbon dioxide (CO_2), the major GHG, has been implicated in global warming and subsequent climate change. For example, the Intergovernmental Panel on Climate Change (IPCC) in its Fifth Assessment Report shows an increase in atmospheric concentration of CO_2 from 280 ppm in 1950 to about 400 ppm in 2000 (Pachauri et al. 2014). Further, computer models developed by the US Climate Change Science Program (CCSP) (Sundquist et al. 2008) project that global annual CO_2 emissions during the next century need to be reduced by more than 75% if its concentration in the atmosphere is to be stabilized at 550 ppm, just about twice the preindustrial level. If not countered, global climate change is predicated to have devastating impacts on natural and human systems, and is therefore, likely to become the most critical and complex environmental concern facing humanity over time (de Hipt et al. 2018; Erwin 2009; IPCC 2007; Olsson et al. 2015; Pachauri et al. 2014).

In a bid to ease global climate change and mitigate its impacts, efforts are being made to reduce emissions of CO_2 and other GHGs into the atmosphere. The 21st Conference of Parties (COP) to the United Nations Framework Convention on Climate Change (UNFCCC) at its meeting from 30th November to 12th December 2015 in Paris, France,

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negotiated the Paris Climate Agreement where countries increased their commitment to reduce GHG emissions. Except for the US that plans to withdraw from the Agreement despite its earlier pledge to reduce GHGs emission by 26–28% of 2005 levels by 2025, all major GHGs emitters promised to meet their pledges. China, India, Canada, Australia and Brazil set reduction pledges of 60–65%, 33–35%, 30%, 26–28% and 43%, respectively, by 2030 compared to 2005, European Union and Russia of $\geq 40\%$ and 25–30%, respectively, by 2030 compared to 1990, while Japan set its pledge at 26% by 2030 compared to 2013 (Bodansky 2016). Unlike the Kyoto Protocol that placed responsibilities of reducing emissions to only industrialized nations, this agreement seeks to achieve a faster and significant response to global climate change by obliging all emitters to take nationally determined action to reduce emissions of CO_2 into the atmosphere. However, growing evidence shows that without sequestering current atmospheric CO_2, global climate change and its associated impacts will not be addressed (Batjes 1999; IPCC 2007; Pachauri et al. 2014; Sundquist et al. 2008; Zhang et al. 2018).

Natural ecosystems such as wetlands have been shown to be among the most important, cost-effective and efficient options for sequestering atmospheric CO_2 (Adhikari et al. 2009; Lal 2008; Lane et al. 2016; Mitra et al. 2005; Nahlik and Fennessy 2016; Villa and Bernal 2017; Yu et al. 2012). Wetlands are the most productive ecosystems and have the highest soil carbon (C) density compared to other ecosystems such as forests and grass/shrub-lands (Kayranli et al. 2010; Villa and Bernal 2017; Zeng et al. 2014). According to Batjes (1999), wetlands have been net sinks of atmospheric CO_2 since the Last Glacial Maximum. The Millennium Ecosystem Assessment (MEA) Report (Alcamo 2003) supplements that wetlands are not only vital ecosystems in the response strategy to global climate change through carbon sequestration (CS), but also provide services that help in adapting to climate change impacts. Further still, compared to alternative options, wetlands sequester C at no or limited additional costs (Nahlik and Fennessy 2016; Sundquist et al. 2008; Villa and Bernal 2017).

The IPCC argues that natural ecosystems sequestering C must be enhanced for effective CS. The arguments are that either human disturbance (Davidson 2014; Howe et al. 2009; Lal 2005, 2008; Nahlik and Fennessy 2016; Zhang et al. 2015) or global climate change (DeLaune and White 2012; Erwin 2009; Kayranli et al. 2010; Moomaw et al. 2018; Post et al. 2004) have decreased the capacity of these ecosystems to sequester C. Further decline is even expected under global population growth and climate change scenarios (Villa and Bernal 2017). As a result, a number of studies have investigated the various measures to enhance C storage by natural ecosystems such as forests (De Deyn et al. 2008; Houghton and Nassikas 2018; Lal 2005; Yan et al. 2018) and agroecosystems (Lal 2002, 2018; Post et al. 2004; Zhu et al. 2010). However, examination of literature pertaining enhancement measures for CS by wetlands shows limited and scattered results, and where measures have been proposed they are constrained by a poor understanding of how they may affect wetland functioning, both in an ecological and socio-economic perspective. Adhikari et al. (2009) and Villa and Bernal (2017) have explained that unlike other ecosystems, wetlands being transitional zones between land and water are highly complex. Limitations of proper understanding of their processes must be the primary target for development of workable action plans to sustain their longevity.

Recognizing that wetlands provide a variety of ecosystem services to different interest groups, efforts to mitigate climate change using wetlands must be well evaluated to ensure unbiased access to wetland services. In this review, our focus is on assessment of enhancement measures for CS by wetlands. We start by highlighting the concept of CS, and then make a critical analysis of the various enhancement measures by discussing challenges that may hinder their adoption. Further, we make a discussion concerning data overlap on CS that may limit a proper understanding of CS potential of wetlands. In doing this, we strive to address the concerns of wetland scientists, managers and other stakeholders pertaining enhancement of CS by wetlands and its contribution to global C balance and climate change mitigation.

A comprehensive literature search, with emphasis on recent studies was conducted according to PRISMAS guidelines (Moher et al. 2009) with help of literature data from Google Scholar® (https://scholar.google.com/) and ScienceDirect® (https://www.sciencedirect.com). Grey literature was also assessed, but much effort was made to get similar information in easily accessible published studies. Literature data analysis centred on documents that have explored enhancement options for increasing carbon sequestration by wetland ecosystems.

2 Concept of Carbon Sequestration by Wetlands

Carbon sequestration (CS), generally, denotes the process involved in the capture and long-term storage of atmospheric CO_2 in natural or human engineered sinks. In the present text, CS refers to the capture of atmospheric CO_2 and its long-term storage in wetlands, with minimal chances of being released back into the atmosphere. In quantitative terms, CS is an expression representing a change in C stocks, either in or between wetland ecosystems. The definite amount of C a wetland can sequester on a given temporal and spatial scale, termed as *carbon sequestration potential* denotes both the maximum rate of C storage (for example the rate of growth of plants) and the maximum amount of C that can be stored (for example in plants or soil) (Zhu et al. 2010). How long the stored C stays in the wetlands is still a debatable question, though a number of studies show that undisturbed wetlands can store C for several hundreds of years to millennia (Ezcurra et al. 2016; Kurnianto et al. 2015; Mitra et al. 2005).

The process of CS is majorly mediated by plants (wetland macrophytes), via photosynthesis. During photosynthesis, plants assimilate atmospheric CO_2 into their tissues as simple sugars, which are converted to complex materials such as lignin and cellulose and deposited into leaves, stems and roots, and finally in soil (as soil organic carbon; SOC) when plants die. Most wetland plants are known to use atmospheric CO_2 as their main C source, and because of their high rates of gross primary production compared to terrestrial plants, they have a high assimilation capacity for CO_2. An understanding of CS potential of wetlands at local and global scales is necessary to account for the contribution of wetlands in offsetting C emissions into the atmosphere.

Although biogeochemical processes in wetlands may be similar, factors controlling CS across wetland types are contested. Water table height (Adhikari et al. 2009; Olsson et al. 2015; Villa and Bernal 2017) and temperature (Mitsch et al. 2013; Olsson et al. 2015) have mostly been recognized to exert a primary control on CS because of their strong influence on organic matter decomposition. However, a number of studies have not found any significant impact of these factors on CS by wetlands. For instance, while assessing the impact of flooding depth and duration on organic matter decomposition, Mueller et al. (2016) concluded that organic matter decomposition rates are not directly driven by water table depth. Actually, Christensen et al. (1998) indicated earlier that even though water table height can have a big impact on C turnover, its influence might be weaker particularly at wetter sites, where other factors may exert a stronger control. Similarly, Villa and Bernal (2017) understand that in wetlands where soils are waterlogged and anoxic, temperature may exert a significantly less impact on organic matter decomposition, and subsequent CS. This is in agreement with Sjögersten et al. (2014) who appreciated that unlike cold regions, temperature is unlikely to be a major factor influencing CS in tropical regions. With climate change, alteration of temperature and precipitation regimes may further complicate understanding CS processes and their controls in wetlands.

Organic matter decomposition generates CO_2, which is then emitted back into the atmosphere. Because of the waterlogging that creates anoxic conditions in wetlands, anaerobic decomposition predominates. It is shown that anaerobic decomposition is far less efficient as compared to aerobic decomposition (Olsson et al. 2015; Villa and Bernal, 2017). As a result, most of the organic matter in wetlands remain undecomposed and later buried, resulting into CS. However, under anaerobic conditions, wetlands through the process of methanogenesis, emit C (as methane, CH_4) into the atmosphere, which is 28 times more effective than CO_2 in terms of global warming potential on a 100-year time scale (Moomaw et al. 2018). Recently, Angle et al. (2017) have nonetheless, differed from the widely known paradigm that microbial methanogenesis can only occur in anoxic habitats. They compared methanogenesis in oxic and anoxic soils at three wetland sites under differing land cover types: emergent vegetation, periodically exposed mud flats, and continuously submerged ones (open water). The authors learnt that CH_4 production and methanogenesis activity were up to ten and nine times, respectively, greater in oxygenated soils than anoxic ones. They further argued that up to 80% of methane fluxes in wetlands could be attributed to methanogenesis in oxygenated soils. This revelation calls for more studies both on temporal and spatial scales to comprehend the methane paradox in wetlands.

Apart from CO_2 and CH_4, particulate organic carbon (POC), particulate inorganic carbon (PIC), dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC) are the other C forms that can enter and exit wetlands (see Fig. 1 for C cycling in wetlands). Therefore, the net C sequestered by a wetland (known as net ecosystem C balance; NECB) is obtained as the difference between C input and output (Zhu et al. 2010). Post et al. (2004) emphasized that calculation of NECB should take care of all allochthonous C inputs into the wetland.

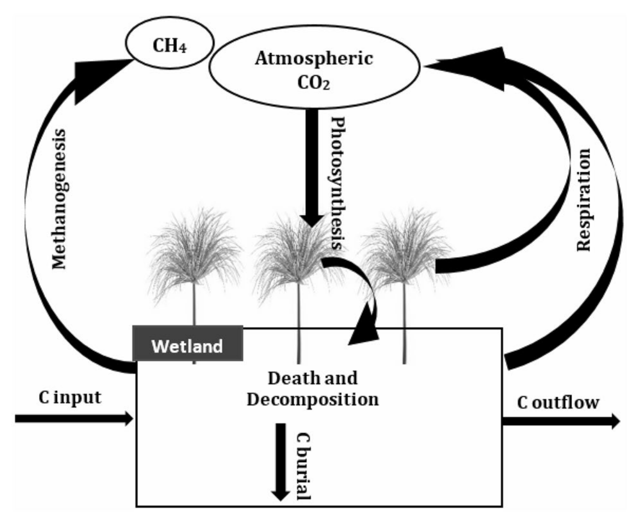

Fig. 1 Simplified schematic overview of C cycling in wetlands. C input represents lateral inflow of POC, PIC, DOC and DIC from upstream ecosystems while C outflow represents their exit from the wetland to downstream ecosystems. Methanogenesis represents the anaerobic part of respiration. Adapted from Kayranli et al. (2010)

3 Enhancement Measures for Carbon Sequestration by Wetlands

As earlier explained (see Fig. 1), photosynthesis converts atmospheric CO_2 to organic C while respiration (inclusive of methanogenesis), occurring mainly due to organic matter decomposition in the wetland, release C back into the atmosphere. Carbon sequestration (CS) by a wetland at a particular time can therefore, be calculated employing Eq. (1) for C mass balance:

$$C_{SW} = C_I - C_O. \tag{1}$$

From Eq. (1), enhancing CS implies finding measures that maximize C input (C_I) while at the same time minimizing C output (C_O), so that net CS (C_{SW}) is increased beyond current potential. Such measures aim at increasing organic matter input, apportioning C to longer-lived pools, and increasing residence times of C pools. These, however, under similar climatic conditions depend on the management state of wetlands (Post et al. 2004; Villa and Bernal 2017). Hence, any attempt to enhance CS by wetlands should target two fundamental approaches: (1) putting in place measures that increase wetland spatial extent (which we have referred to as non-manipulative approach), and (2) altering characteristics of certain wetland components involved in CS (referred to here as manipulative approach). We make a review of such measures under both approaches and discuss the challenges that may affect their implementation.

3.1 Non-manipulative Approach to Enhance Carbon Sequestration by Wetlands

Global wetland inventories indicate a decreasing trend of natural wetland area across all regions. The most recent study by Davidson (2014) shows average wetland area loss of 56.3% in Europe, 56.0% in North America, 45.1% in Asia, 44. 3% in Oceania and 43.0% in Africa, with global loss averaging between 54 and 57% since 1900 AD. To enhance CS, increment of wetland spatial extent has been proposed through the following measures:

3.1.1 Wetland Protection

Many studies have recommended protection of natural wetlands as a vital measure to enhance CS (Villa and Bernal 2017; Yu et al. 2012; Zhang et al. 2015). These studies demonstrate that functioning of natural wetlands, especially CS is highly sensitive to land use change. In this regard, wetland protection refers to safeguarding natural wetlands against human disturbances that can alter vegetation, hydrological and soil regimes. An earlier study by Gorham (1991) showed that only northern high altitude and tropical wetland peat contain over 600 Pg C, more than two-thirds that stored in the atmosphere (Moomaw et al. 2018) and twice the storage of world's forest biomass (Bonn et al. 2016). Howe et al. (2009) compared C stocks in natural and disturbed Australian wetlands and showed that natural wetlands stored between 15 and 50% more C than their disturbed counterparts. Nahlik and Fennessy (2016) also carried out a comprehensive assessment of CS potential of US wetlands for the top 120 cm soil horizon. The authors observed significantly high C stocks in natural wetlands (2.25 ± 0.28 Pg C) in only 5.5×10^6 ha compared to that in disturbed wetlands (1.63 ± 0.33 Pg C), despite covering a larger area (7.0×10^6 ha). Villa and Bernal (2017) showed that natural wetlands globally store about 400 Pg C in the top 1 m of soil, higher than any biome. The implication and justification for protection of natural wetlands, therefore, is that their conversion significantly contributes to atmospheric C pool compared to other ecosystems. This has already been confirmed in China by Zhang et al. (2015) who showed that compared to other natural ecosystems, wetland conversion caused the highest C loss (113 Tg) from 1995 and 2010.

However, we have noted that protecting wetlands by maintaining their natural components might not necessarily enhance CS per se. Minkkinen et al. (2002) assessed the impact of disturbance of Finnish wetlands on CS for the period 1900–2100. They showed that CS had increased from 2.2 Tg year^{-1} in 1900 when all natural wetlands were undisturbed, to 3.6 Tg year^{-1} in 2002 when about 60% of them had been disturbed by drainage. They further observed that the radiative forcing of the wetlands had decreased by about 3 mW m^{-2} over the same period. Wang et al. (2018) recently investigated the impact of disturbance (due introduction of invasive exotic plant species) on CS by Chinese wetlands. The authors note that wetlands (or wetland sections) disturbed by exotic invasive species *Spartina alterniflora* had significantly higher concentration and density of SOC compared to natural wetlands covered by the native species *Cyperus malaccensis*.

Natural wetlands, even under undisturbed conditions may also act as C sources, mainly as CH_4. Although the actual contribution of wetlands to global annual emission of CH_4 is still arguable, reported figures are of considerable significance, ranging from 20% to as high as 40% (Bloom et al. 2016; Laanbroek 2009; Parker et al. 2018; Sharifi et al. 2013). An effort by Wang et al. (1996) to summarize the contribution of wetlands by region to the total global wetland CH_4 emission reported that northern, temperate and tropical wetlands emit 34%, 5% and 60% in that order. A comparison of CH_4 emission rates between a created and natural freshwater wetland in Ohio, US, by Sha et al. (2011) showed that the natural wetland had significantly higher emission rates (21.5 mg CH_4 m^{-2} h^{-1}) than the created one

(13.5 mg CH_4 m^{-2} h^{-1}). The authors attributed these results to the history of higher net primary productivity in the natural wetland. In the Mississippi River Delta, Louisiana, US, an examination of the ecosystem level CH_4 fluxes from a natural tidal freshwater marsh and brackish marsh wetland by Holm et al. (2016) revealed annual emissions rates of 62.3 g CH_4 m^{-2} in the former and 13.8 CH_4 m^{-2} in the latter. Additionally, Pereyra and Mitsch (2017) compared CH_4 emissions from natural and disturbed freshwater cypress (*Taxodium distichum*) wetlands in Florida, USA. Findings indicated that fluxes from natural wetlands (15.6–49.5 mg CH_4 m^{-2} d^{-1}) were significantly higher than from disturbed wetlands (− 1.4 to 4.0 mg CH_4 m^{-2} d^{-1}). Sun et al. (2018) have studied the CH_4 flux of the two predominant natural wetland types (freshwater marshes and permafrost peatlands) in Northeast China. Results showed respective emission rates of 9.5 µg CH_4 m^{-2} s^{-1} and 1.34 µg CH_4 m^{-2} s^{-1}. According to Laanbroek (2009), natural wetlands show high CH_4 emission rates because the presence of vegetation plays a vital role by providing the C necessary for CH_4 production and at the same time aiding the release of the produced CH_4 through their internal gas lacunas.

Apart from CH_4, natural wetlands can further be C sources through fluvial discharge of DOC. Sobek et al. (2005) noted that 87% of surface water bodies such as lakes are supersaturated with and are net emitters of CO_2 whose source can be traced to DOC input from ecosystems such as wetlands. According to Sharifi et al. (2013), about 15% of the terrestrial organic matter flux to surface water bodies originates from wetlands. In New Zealand, Moore and Clarkson (2007) reported a close relationship between the proportion of wetlands within a catchment and DOC concentrations in stream and peat pore waters, while Lambert et al. (2014) established that wetlands are near-infinite reservoirs and important sources of DOC in a small headwater catchment of western France. Related results have been observed in Canada (Eckhardt and Moore 1990), the UK (Freeman et al. 2001), China (Wang et al. 2013) and Spain (González-Ortegón et al. 2018). Additionally, Richey et al. (2002) studied outgassing from Amazonian rivers as a source of atmospheric CO_2. Findings demonstrated that the role of these rivers in C loss through outgassing of CO_2 is vital, contributing 1.26 ± 0.3 Mg C ha^{-1} $year^{-1}$. The authors explained that the possible source of this CO_2 was DOC transported from wetlands, which was then respired and outgassed downstream. At basin level, they further showed a flux rate of 0.5 Gt C $year^{-1}$, which was in order of magnitude greater than fluvial export of organic C to the ocean. Indeed, Abril et al. (2014) concurred with Richey et al. (2002) by estimating that half of Amazonian wetlands' gross primary production is exported to river waters as DOC, as compared to only a few per cent of gross primary production exported by other upland ecosystems. Moreover, the authors were convinced that wetland C export was potentially large enough to account for at least the 0.21 Pg C emitted annually as CO_2 from the central Amazon River and its floodplains. Although these observations do not suggest that natural wetlands are net C emitters, they give an incite of potential for C emission, and raise a debate on the strength of protection of natural wetlands as an option to enhance CS and mitigate climate change. Arriving at any conclusion, however, requires a good understanding of the C budget of natural wetlands not only at local scales, but also at a global level. This is because C emission at a given wetland site may be compensated by net CS at a global scale.

Let that aside, even if natural wetlands are taken as net C sequesters, protecting them to enhance this ecosystem service appears difficult in the face of competing services. Whereas people may obtain some services from wetlands without causing physical alteration on the vegetation, hydrology, and soil, obtaining some types of services (such as provisioning) come at the expense of altering wetland structure to some degree. Common examples can be drawn from Africa and Asia. Many wetlands in these regions are increasingly being used for crop cultivation (predominantly rice, with paddy rice wetlands estimated to cover 18% of total global wetland area; Yoon 2009). In this case, two fundamental questions remain unanswered: (1) how effective are these wetlands in CS? and, (2) is there a possible scenario to optimize both CS and rice cultivation simultaneously in a wetland?

Although most studies refer to such agriculture converted wetlands as C sources, others argue that they can be of double benefits, providing food as well as sequestering C. Studies by Cai (1996) and Ti et al. (2012) on Chinese paddy rice wetlands reported high SOC in paddy soils and attributed it to the continuous inundation during most of the growing cycle. Similar results have been obtained in Japan (Nishimura et al. 2008) and India (Nath et al. 2016). What is important to note is that in all these studies, comparisons of CS were made to upland soils or other agriculture systems. It is therefore not clear how CS potential by paddy rice wetlands compares with that of the natural wetlands prior to their conversion. According to Batjes (1999), by returning the crop residues to soil, lost C in such agricultural wetland ecosystems can be recovered. However, unlike developed countries, such a measure may not have significant results on CS in developing countries where crop residues are used for various purposes such as animal feed, fuel, and construction material. Additionally, the impact of alternate periods of inundation and drying common in paddy rice wetlands on the residence time of C from rice residues has not been studied. Because of the increasing global population, demand for rice will inevitably follow the same trend. This will subsequently result into conversion of more area under natural wetlands into rice wetlands. Thus, CS and rice cultivation are competing services that must be harmonized to create a balance between climate change mitigation and food production.

Further still, whereas preventing human disturbances can protect natural wetlands and enhance CS, it may be challenging to control natural events. Take for example in Canada (Turetsky et al. 2002) and the US (Post et al. 2004) where wetland disturbance due to wildfires has been reported to emit approximately 6460 ± 930 Gg C year^{-1} and 0.2 Pg C year^{-1}, respectively. Unless studies on CS take into consideration such natural occurrences when evaluating CS potential of wetlands, the results are likely to be undermined.

3.1.2 Wetland Restoration

In situations where wetlands have been lost or degraded, they together with their lost functions including CS can be brought back through restoration. For instance, a special presentation to COP15 (in Copenhagen, Denmark) by the Ramsar Convention's Scientific and Technical Review Panel (STRP) highlighted that degraded wetlands are already a significant source of atmospheric CO_2 and that returns from their restoration are 100 times that of alternative C mitigation options. Lal (2002) established that restoration of degraded Chinese ecosystems including wetlands can sequester lost organic C in soil at a rate of 100–200 kg ha^{-1} year^{-1}, resulting into a total C pool of 0.014–0.028 Pg year^{-1}, while Xiaonan et al. (2008) showed that between 2006 and 2010, restoration of Chinese wetlands sequestered 6.57 Gg C year^{-1}. Lamers et al. (2015) estimated that restoration of drained European wetlands could sequester 400 g C m^{-2} y^{-1} on average, higher than the 0.1–1.0 t C ha^{-1} year^{-1} observed earlier by Freibauer et al. (2004). In central Estonia, Järveoja et al. (2016) found out that C emission of 3-year-old restored peatlands was less than half that of unrestored peatlands. A more recent study by Chen et al. (2017) compared C stock of a restored wetland in Illinois, USA, with that of two unrestored wetlands (sedge meadow and marsh). Results showed that C stock of the restored wetland was 25% and 46% greater than that of sedge meadow and marsh wetland, respectively, just 3 years after restoration. On a global scale, the IPCC (2000) estimated a net CS rate of 4 Mt C year^{-1} in 2010, with only 230 million hectares of restored wetland area. These findings give hope that restored wetlands can enhance CS in future, and as such, restoration efforts are being fast-tracked especially in the developed world such as North America (DeLaune and White 2012; Yu et al. 2017), Europe (Gumiero et al. 2013; Moreno-Mateos and Comin 2010; Verhoeven 2014), Australia (Bachmann 2016; Page and Dalal 2001) and parts of Asia (Furukawa 2013; Xiaonan et al. 2008).

However, although restoration aims to bring a lost or degraded ecosystem to its original state prior to the loss or degradation, some ecologists doubt the ability of restoration to recover fully a lost or damaged ecosystem to its original state. Moreno-Mateos et al. (2012) carried out an analysis of 621 restored wetland sites distributed across the world and observed that biological structure (determined mostly by plant assemblages), and biogeochemical functioning (determined primarily by CS in soils) remained on average 26% and 23%, respectively, lower than in reference sites, a century after restoration. A meta-analysis of literature by Yu et al. (2017) on the influence of wetland restoration on CS in the US showed that after 11–20 years of restoration, soil C was significantly lower by 51.7% than in natural wetlands. Above all, it is interesting to note that some degraded wetlands may be un-restorable. An example is drawn from The Netherlands (Zedler 2000), where drained fens resisted restoration despite provision of the necessary requirements. With these observations, there is a possibility that even if lost or degraded wetland ecosystems are restored, some of the lost ecosystem functions, including CS might not be recovered fully.

Additionally, wetland restoration may be less feasible in the developing world. Let alone the high costs involved and the rapidly growing population that will require land for food production and other related activities, reference conditions required for restoration are not readily available. Global wetland studies (Davidson 2014; Mitra et al. 2005) show limited availability of wetland data in the developing world, and for that reason, getting reference conditions for their restoration becomes intricate. In absence of reference conditions, two concerns arise: (1) to what extent will the degraded or lost wetlands and their CS service be recovered? and, (2) in case wetlands are 'restored' with non-reference conditions, what will be their impact on CS and other ecosystem services? In the same sense, our general observation is that whereas most studies on newly restored wetlands conclude that they sequester C, their net C balance is still uncertain. We are convinced that making a conclusion on whether wetland restoration enhances CS or not is dependent upon the age of the wetland since wetland processes establish overtime. Zedler (2000) supports our opinion by explaining that wetland restoration is a complicated process whose definite results may be realized after long periods ranging from decades to centuries.

3.1.3 Wetland Creation

Creation of wetlands is being advocated for to help compliment functions of existing or lost wetlands. Mitsch et al. (2013) recommended that to increase the current net CS (that is either negative or not significantly different from zero) by North American wetlands, new wetlands should be created. The authors show that created wetlands have CH_4 emission rates lower or comparable to natural wetlands after 13–15 years. In a related study, Mitsch et al. (2014) showed that the CS rate (219–267 g C m^{-2} $year^{-1}$) of two created riverine wetlands in Ohio, USA, was significantly higher than that measured in a reference natural wetland (140 ± 16 g C m^{-2} $year^{-1}$), 15 years after creation.

Created wetlands (commonly known as constructed wetlands) have been traditionally applied for wastewater treatment. For example, in Europe and the US alone, about 5000 and over 4000 wetlands, respectively, have been constructed to treat wastewater (Yoon 2009). According to Chen et al. (2017), constructed wetlands used for wastewater treatment may be net C sinks. They explain that because constructed wetlands have a smaller water column compared to natural wetlands with stagnant water and larger sediments, they can sequester more C (per unit area) than natural wetlands. Mander et al. (2008) investigated the CS potential of constructed wetlands treating domestic wastewater in Estonia and recorded a sequestration potential ranging from 649 to 484 kg C $year^{-1}$. In these studies, nevertheless, an information gap remains in examining the dynamics of CO_2 and CH_4 fluxes whose understanding is required to draw the C balance of constructed wetlands. Similarly, design considerations of constructed treatment wetlands give priority to improving wastewater purification efficiency rather than CS. No information is available on the design that can enhance both wastewater treatment and CS. Further still, because plant uptake constitutes a significant pollutant removal mechanism, removal of mature vegetation is inevitable in wastewater treatment constructed wetlands. This, on the other hand, implies negative impacts on CS since plants that would provide organic matter are removed. Thus, if constructed wetlands are to act as effective C-stores, such systems should be set up for this purpose, though they may also offer some other ecosystem services that do not cause physical disturbance to wetland structure. However, as already observed in the case of wetland restoration, we believe that wetland creation is not feasible in regions experiencing high population growth rates since the available land is being converted to agriculture and other uses. Indeed, Villa and Bernal (2017) have observed a positive correlation between wetland loss and population growth.

Though literature on economic incentivization of wetland creation and management for CS was not found, it has been successfully applied on forests ecosystems, and we think it can work for wetlands as well. Jackson and Baker (2010) established that by attaching prices of USD 10, 20 and 30 per ton of CO_2 equivalent (tCO_2e), CS could be enhanced by eliminating 1.8, 2.5 and 2.9 billion tCO_2e per year, respectively, of global emissions arising from deforestation. The authors clarify that the price incentive motivates individuals to create new and properly manage existing forests. Equally, the latest protocol for prioritization of wetland restoration and creation (Comín et al. 2014; Fig. 2) underscores that successively approach to wetland restoration and creation does not involve consideration of only scientific and technical issues (such as biogeochemistry, hydro-geography and morphology) but also evaluates socio-economic aspects.

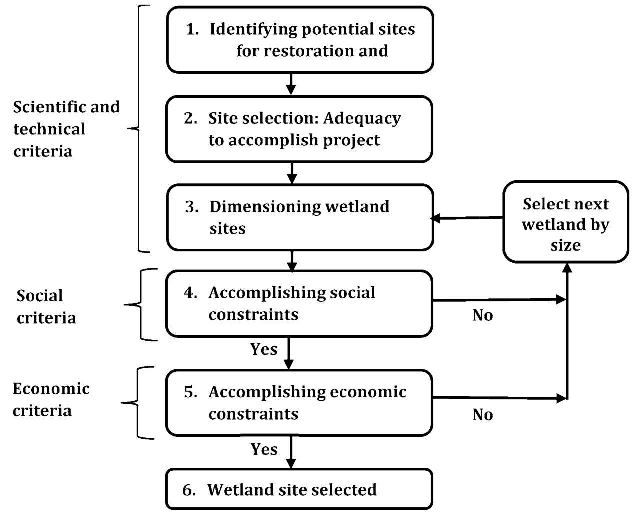

Fig. 2 Protocol for prioritization of wetland restoration and creation. Adopted from Comín et al. (2014)

3.2 Manipulative Approach to Enhance Carbon Sequestration by Wetlands

It is already acceptable that organic matter decomposition is the most important processes that facilitates C release from wetlands. As a result, to enhance CS, measures that can help to suppress organic matter decomposition have been proposed to enhance CS. Such measures that work by altering the characteristics of wetland components involved in organic matter decomposition and subsequent CS are discussed below:

3.2.1 Biotechnology

Enhancement of C storage by biotechnology through manipulation of soil microbes and vegetation has been suggested as a potential measure (Flores et al. 2005; King 2011; Lal 2008; Post et al. 2004; Trivedi et al. 2013). Manipulation of soil microbes and plants involves undertaking actions intended to modify the genetic makeup and community composition of the microbes and plants. Proponents of this CS enhancement measure point out that because microbial communities in wetland soils are involved in organic matter decomposition, while vegetation contributes the organic matter, they can be manipulated to reduce organic matter decomposition rates and hence increase CS. King (2011) observed that CS in soil is affected by specific members of microbial communities, whose identification would be a great step towards enhancement of CS. This was supported by Monson et al. (2006) who from their study on a subalpine forest ecosystem in Colorado, USA, noted that changes in microbial community composition at the genus level could be responsible for seasonal changes in specific aspects of organic matter decomposition. A study by Oni et al. (2015) on microbial communities and organic matter composition in surface and subsurface wetland sediments of the Helgoland mud area, North Sea, also showed a strong positive correction between total organic carbon (TOC) content and specific microbial populations. Further, Laanbroek (2009) established that sulphate-reducing bacteria play a significant role in limiting CH_4 emission in coastal marine wetland ecosystems. However, the authors highlight that even though these bacteria are abundant in freshwater wetlands, they do not always respond immediately to the supply of fresh sulphate, and as a result, their manipulation could be an important step to suppress CH_4 emission from freshwater wetlands.

Nevertheless, manipulation of microbial populations might not be an important measure for enhancing CS since the activity of microbial communities may depend on other environmental parameters. Kim et al. (2007) reported no strong positive correlation between soil organic matter content and microbial communities, at least at a coarse level resolution in a pristine Brazilian forest. Likewise, Tang et al. (2018) recently investigated the impacts of microbial community on C mineralization across three climatically contrasting sites. Their findings indicated that the role of microbial community in C mineralization is weaker than that of other factors such as temperature, pH and soil substrate type. In the same perception, because control of soil and climatic conditions is unrealistic, especially at field level, we consider that enhancing CS through manipulation of soil microorganisms might not be a feasible option.

Whereas we did not find a specific study where manipulation of wetland vegetation has been done to enhance CS, it has been investigated on terrestrial plants. In a study "Transgenic *Bt* plants decompose less in soil than non-*Bt* plants", Flores et al. (2005) observed that soil microcosms amended with biomass of plants genetically modified by a bacterium *Bacillus thuringiensis* (*Bt*) emitted significantly less C (as CO_2) compared to biomass from non-*Bt* modified plants. The authors attribute this finding to high accumulation of lignin in *Bt*-modified plants, which makes the resultant biomass resistant to decomposition. For both soil microbes and wetland plants, uncertainties still exist in understanding the specific genes involved in mobilization and stabilization of C, and how their manipulation may affect wetland functioning, including flora and fauna assemblages, and nutrient cycling.

3.2.2 Use of Biochar

Biochar, a porous carbonaceous solid, is produced by thermochemical conversion of biomass in an oxygen depleted atmosphere. The use of biochar C has been traditionally practised in terrestrial agricultural systems where its application on soil improves soil chemical characteristics such as causing increment in soil pH, plant nutrient availability and SOC, and decreasing aluminium toxicity (Filiberto and Gaunt 2013). Because of its long residence time in soil which can range from 100 to 1000 years (Yadav et al. 2017), application of biochar C to soil is considered a CS enhancement measure as well.

Although studies involving use of biochar C in wetlands are limited, it has been recommended as one of the most important manipulative CS enhancement measures. "Biochar addition to wetland peat soils may be a potential avenue for decreasing greenhouse gas emissions in order to maximize ecosystem services like carbon sequestration from these crucial habitats", Keenan (2016) suggested following a study investigating the effects of biochar C on wetland and agricultural soil C emissions in North Carolina, USA. A study in Japan by Pratiwi and Shinogi (2016) reported that application of biochar C in paddy rice wetlands reduced CH_4 emission into the atmosphere by 45.2–54.9%. In Guangdong Province, China, Qin et al. (2016) investigated the impact of biochar C application on CH_4 and gross GHG emissions in a paddy rice wetland. Results showed that at biochar C addition rates of 10 t ha^{-1} and 20 t ha^{-1}, CH_4 emissions (mg m^{-2} h^{-1}) were 1.07 ± 0.52 and 0.99 ± 0.21, respectively, both significantly lower than those recorded in the control wetland plot (1.47 ± 0.39), where biochar C was not added. Similarly, gross GHG emissions (kg CO_2^{-e} ha^{-1}) were 1111.68 ± 508.94 and 918.59 ± 227.42, respectively, both significantly lower than that measured in the control plot (1329.08 ± 411.77). The findings further indicate that both CH_4 and gross GHG emissions reduce with increase in biochar C application rate. The rationale is that biochar C is recalcitrant, exhibiting slower decomposition and turnover rates, unlike the labile C which undergoes faster decomposition rates. Nonetheless, in paddy rice wetlands where the main carbon substrate (rice straw) is added to improve soil fertility, our view is that replacing it with biochar C to enhance CS will be met with resistance from farmers since it's likely to compromise the quality and quantity of rice yields. Further, from these studies, an understanding of the influence of the introduced biochar C on native C, microbial communities, water infiltration properties of wetland soils, and future vegetation re-colonization is still lacking.

3.2.3 Fertilization

Fertilization of soil through addition of nutrients is shown to enhance CS, and is being commonly applied in forest ecosystems, which naturally are nutrient poor (Dou et al. 2015; Ehtesham and Bengtson 2017; Gilliam et al. 2018; Lal 2005) and in agricultural soils (Batjes 1999; Freibauer et al. 2004; Lal 2018) which constantly lose soil nutrients due to continuous cultivation. The authors argue that improved status of soil nutrients such as nitrogen (N), phosphorus (P) and sulphur (S) enhances net primary production (NPP) and hence CS. Batjes (1999) highlighted that because P and S contents are intrinsic soil properties, unless they are increased by fertilization, they may set a limit to NPP and hinder CS. The author further showed that sequestration of 10,000 kg C in soil requires about 833 kg N, 200 kg P and 143 kg S. LeBauer and Treseder (2008) analysed 126 studies investigating the impact of N fertilization on NPP of various ecosystems (including wetlands) spread across the entire globe (excluding Antarctica). They observed that N fertilization had a significant impact on NPP, with an average increase of 29%.

However, wetlands, given that they occupy the lowest points of any catchment, receive runoff from the catchment that comes along with nutrients making them more nutrient-rich compared to upland ecosystems. As a result, justification for enhancement of CS through fertilization could be possibly applicable to altered agricultural wetlands such as those used for paddy rice cultivation (Cai 1996; Nath et al. 2016; Nishimura et al. 2008; Ti et al. 2012). Nonetheless, adoption of this CS enhancement measure could result into nitrate contamination of ground and surface drinking water sources, while increased concentrations of N and P can affect ecology of wetlands and surface water ecosystems like streams and lakes, through eutrophication. Irrespective of these impacts, N fertilization to enhance CS can still be challenged. Whereas N fertilization is known to enhance CS by increasing NPP, Villa and Bernal (2017) show that increasing N levels in soil lowers the carbon-to-nitrogen (C:N) ratio, which leads to increased organic matter decomposition and hence reduction in CS. Batjes (1999) and Lal (2005, 2018) advise that if the intention of N fertilization is to enhance CS, soil C content should first be determined to inform the level of N addition. The C:N ratio that significantly lowers organic matter decomposition has been recommended as one that is greater than 30:1 (Dioha et al. 2013).

3.2.4 Use of Humic Acids

To increase residence time of C and thus CS, addition of humic acids to soil has been recommended (Piccolo 1996; Spaccini et al. 2000, 2002). According to Piccolo (1996), addition of humic acids to soil could increase the residence time of organic C by several hundred years. The explanation given is that humic acids prevent rapid decomposition of the labile organic matter entering soil with litter or plant residues. For example, a 3-month experimental study by Spaccini et al. (2002) examined the effect of humic acids on soil CS. Results showed that soil samples added with humic acids either from lignite or compost retained, respectively, 96.0% and 90.4% organic carbon, significantly higher than that (30.8%) retained in the soil sample where humic acids were not added. The study further revealed that highest CS was in soil samples with silt and clay-sized particles. Even though the study was conducted using upland soils, this result gives a suggestion that humic acids can have a significant impact on wetland CS given that such soil particle sizes are a common characteristic of wetland soils. The anxiety, however, relate to a limited understanding of how humic acids may impact wetland soil physico-chemical characteristics, including nutrient availability and uptake.

4 Data Overlap on Carbon Sequestration by Wetlands

To account for the contribution of implementation of various measures for enhancing CS by wetlands on global C balance and climate change mitigation, knowledge of current CS potential is paramount. However, there is no consensus on the actual C stock sequestered in wetlands. According to Mitra et al. (2005), wetlands contain between 350 and 535 Pg C. An estimate by Adhikari et al. (2009) showed that out of the total 1550 Pg C held in organic soils, 150 Pg is in wetland soils. Percentagewise, studies report wetland C content as 12% (Erwin 2009), 20–25% (Yu et al. 2012), 35% (Zeng et al. 2014), 30% (Bonn et al. 2016) and ~33% (Villa and Bernal 2017) of the world's C in organic soils. Consequently, it is hard to quantify precisely the carbon pool arising from undertaking enhancement measures if current C stocks are unknown.

A number of reasons have been observed for occurrence of these uncertainties in accounting for global C stocks in wetlands. For example, to date, the exact global wetland area remains unknown as it is reported with great variation such as 2% (Post et al. 1982), 3% (Maltby and Immirzi 1993), 5–6% (Mitra et al. 2005), 4–6% (Yu et al. 2012), and 5–8% (Nahlik and Fennessy 2016) of the earth's land surface. According to Mitra et al. (2005), the divergent views in reporting global wetland area originate from lack of a harmonized definition of wetlands. Whereas the Ramsar Convention on wetlands adopted a broad definition, considering wetlands as "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres", the definition varies from place to place depending on the objectives and background of the author. Other researchers (Liu et al. 2017; Rebelo et al. 2009; Serran and Creed 2016) however, blame the disagreement over global wetland area on differences in mapping techniques.

Still, even if the wetland definition was unanimous and mapping techniques accurate and similar, C stock in wetlands would continue to be reported differently due to variations in approaches (e.g. soil depth) used for its estimation. Though it is a general idea that the depth of C accumulated in a wetland increases over time (Fig. 3), differences occur among soil types and on a spatial extent, and can therefore affect estimation of CS. With these uncertainties, studies have reported C stocks in wetlands with varying considerations of soil depths such as at less than 1 m (Chen et al. 2017; Howe et al. 2009; Lane et al. 2016; Wang et al. 2018), up to 1 m (Adame et al. 2015; Friess et al. 2016; Köchy et al. 2015; Page and Dalal 2001), while others have exceeded 1 m (Chimner and Karberg 2008; Nahlik and Fennessy 2016). According to Chimner and Karberg (2008), the depth of organic soil in various wetlands around the world may be significantly deeper, exceeding 10 m in some cases. In the same way, because most studies report wetland C storage as a percentage of organic soil, difficulty arises in determining C storage per unit area especially with imprecise knowledge on organic soil depth, plant C allocations, C use efficiency, dynamics and residence times in different wetland types. We also observed that CS studies have given little attention to freshwater wetlands, especially in the tropical region. This is likely to affect the accuracy of reporting global C stocks in wetlands.

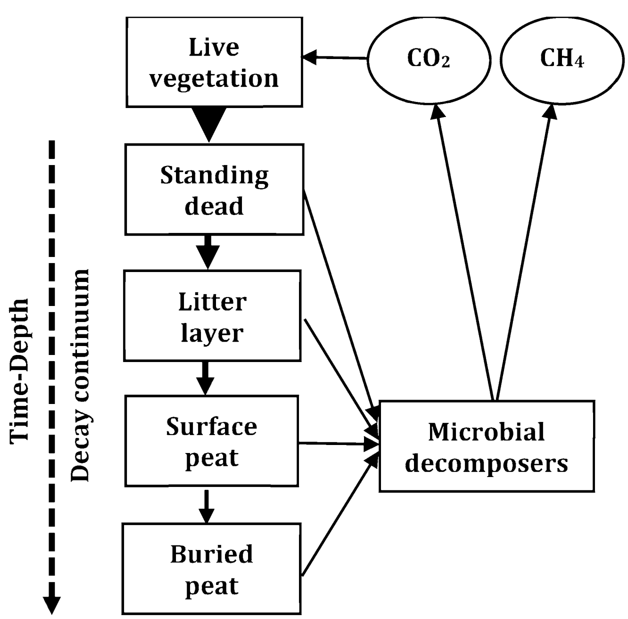

Fig. 3 Time–depth continuum of organic matter decomposition and accretion in wetlands. Adopted from DeBusk et al. (2001)

5 Conclusions and Recommendations

Wetlands are vital ecosystems in global climate change mitigation, through CS. However, anthropogenic activities especially conversion to other land uses are affecting their capacity to sequester C. This has necessitated finding options to enhance the ability of wetlands to sequester C and enable them remain relevant in global C balance and climate change mitigation. Increasing wetland spatial extent, and manipulating certain wetland components can enhance CS of wetlands beyond current potential. However, the feasibility of adopting these CS enhancement measures is limited in various aspects. Enhancing CS by increasing wetland spatial extent is challenged by existence of competing land uses, which may be more economically sound than wetland conservation. Additionally, increasing wetland spatial extent may not necessarily enhance CS potential of wetlands given existence of competing ecosystem services, such as harvesting of vegetation for various purposes. On the other hand, enhancing CS by manipulating various wetland components involved in CS suffers uncertainties of its impacts on wetland functioning. As a result, whereas it is important to enhance CS potential of wetlands to help mitigate climate change, enhancement options need to strike a balance between CS and other ecosystem services provided by wetlands. In the same sense, we argue that C management efforts should give priority to conservation of existing natural wetlands. Nonetheless, we make the following general recommendations where considerations for enhancement of CS by wetlands are to be made:

1. Economic incentives that are not only foreseeable but also adequate to match economic returns from competing land uses need to be attached to wetlands to stimulate conservation of existing natural wetlands, and restoration and creation of new ones. A similar approach as one for REDD+, a United Nation's programme aimed at reducing emissions from deforestation and forest degradation could be adopted.
2. Studies need to evaluate how prioritization of CS, and manipulation of wetland components may influence wetland functioning and other competing ecosystems services provided by wetlands. This calls for involvement of all stakeholders, including local wetland users whose livelihoods are dependent on wetlands and who are directly impacted by implementation of any wetland management decision.
3. Long-term investigation of C dynamics and cycling in restored and created wetlands will help understand controls of CS and residence times C in these wetland

ecosystems. The same is needed for tropical freshwater wetlands, given their high productivity, and the high and insignificantly variable temperatures throughout the year, which can have a big influence on organic matter decomposition. Additionally, research on the influence of natural events such as wild fires on wetland C fluxes will help improve accuracy of global wetland C accounting.

4. Lastly, harmonization of wetland definition, extent, and approach used to assess C stocks in wetlands will help reduce the uncertainties in reporting CS potential of wetlands.

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Compliance with Ethical Standards

Conflict of interest On behalf of all authors, the corresponding author states that there is no conflict of interest.

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