This chapter is a draft. It will be peer reviewed and subject to change.

5 Mitigation measures in freshwater ecosystems

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Highlights

- Aquatic environments can function as greenhouse gas (GHG) sources and sinks based on their environmental state and management. Land use, surrounding vegetation, pollution, human activities, hydrologic regime, and climate can influence the emissions profile of freshwater peatlands, marshes, swamps, lakes, streams, rivers, and tidal wetlands. While restoration of wetlands and floodplains are effective measures, stronger priority should be given to protecting existing natural wetlands and floodplains.
- In addition to Blue Carbon Ecosystems (including freshwater-dependent coastal and marine systems), emission reduction potential from freshwater systems needs to be more commonly included as measures to reduce atmospheric GHG emissions alongside sectors outside of land use, such as energy and transport.
- The potential (or use) of catchment and coastal zone scale policies, programmes and investments to support effective and sustainable emission reduction strategies needs to be recognised and adopted. GHG production in aquatic systems is driven by nutrient and organic carbon inputs from watersheds. Effective emission reduction strategies may entail integrated approaches for land management and regenerative agriculture, restricting nutrient loading (including improved wastewater treatment capacities), maintaining and improving ecohydrologic connections.
- Natural solution schemes (both Nature-based solutions and Green-grey infrastructure) need to be designed with the full range of ecosystem services objectives included alongside carbon sequestration to reduce risk of maladaptation. Carbon sequestration is only one of many valuable services provided by aquatic ecosystems. The multiple direct and indirect co-benefits, such as flood risk management, biodiversity recovery, sustainable community livelihoods, water quality improvement, that come with watershed scale aquatic ecosystem management need to be realised while integrating the emission reduction targets in the Nationally Determined Contributions (NDCs).
- Emission reduction goals and opportunities need to be given greater emphasis within broad water resources management strategies. There also needs to be financing mechanisms and tools in place to monitor and reduce emissions from freshwater ecosystems and Blue Carbon Ecosystems (BCE) management at the local, regional, and national level. Regulatory reform, capacity building and better data on aquatic environments, are needed to further opportunities and materialise implementation.

5.1. Introduction

Freshwater ecosystems such as wetlands, rivers and lakes are intimately linked to climate mitigation since aquatic environments can act as both greenhouse gas (GHG) sources and sinks based on their environmental conditions and management practices. However, the role of freshwater ecosystems in achieving climate mitigation targets has not yet been acknowledged to the extent reflecting their potential. Reviewing the mitigation potential of different freshwater ecosystems, this chapter makes a clear case that land and watershed-scale policies across different aquatic environments should be adopted for effective and sustainable strategies that support and enhance the role of freshwater ecosystems in mitigating climate change.

Furthermore, freshwater ecosystems need to be more commonly included within the GHG inventories. In order to achieve this, global datasets and reporting methods for freshwater ecosystems' health and coverage should be strengthened through both policies and financing mechanisms. In particular, countries need to be incentivized to develop robust inventories of aquatic ecosystems that can be used to safeguard conservation and mitigate GHG emissions. It is also critical to facilitate development of measurement technologies, especially in contexts where conventional measurement techniques cannot be used, in order to acquire standardised global data sets, targeting long-term, continuous, large-scale data that can be measured simply and at low cost.

Moreover, at the broadest level, governance across all levels needs to be strengthened. Possibilities to align policies such as National Adaptation Plans (NAPs), National Biodiversity Strategies and Action Plans (NBSAPs), and Integrated Coastal Zone Management, with NDCs should be explored to alleviate emissions and enhance sinks from freshwater. Blue Carbon Ecosystems (BCE), particularly mangrove swamps, are more commonly acknowledged for their mitigation potential and have received much greater attention compared to inland freshwater systems in this regard (IPCC 2014) Hence, in this chapter, we focus on freshwater ecosystems (wetlands, lakes, reservoirs and rivers) and freshwater-dependent coastal and marine systems.

This chapter examines the mitigation potential and water-related risks of inland freshwater systems and freshwater-dependent coastal and marine systems. In section 5.2 the mitigation measures addressed are divided into wetlands; rivers and streams; and lakes and reservoirs. Section 5.3 examines trade-offs related to freshwater-based mitigation as well as co-benefits, more specifically the enhancement of ecosystem services through mitigation measures; climate change adaptation and resilience benefits from mitigation measures; and nature-based solutions associated with the mitigation measures. In 5.4 potential implications for governance are mapped, including inclusion in national policies, system-level approaches, and implications of future climate change and socio-economic change. The chapter concludes with a future outlook in section 5.5.

5.2. Mitigation potential of inland freshwater systems and freshwater-dependent coastal and marine systems

Depending on the management, wetlands can act as GHG sources or sinks (Hamdan & Wickland, 2016). While emission, sink and sequestration patterns are widely studied and understood for some wetlands there is considerably less research on rivers and streams. This section elucidates the mitigation potential and measures based on existing knowledge (Table 5.1).

Table 5.1. List of mitigation measures in freshwater ecosystems addressed in this chapter, including data on climate mitigation potential when available in recent IPCC reports (IPCC 2019; IPCC 2022).

Mitigation measure	**Mitigation potential GtCO2-eq yr^{-1}**
Reduce conversion, draining, burning of peatlands	0.45–1.22
Reduce conversion of coastal wetlands (mangroves, seagrass and marshes)	0.11–2.25
Peatland restoration	0.15–0.81
Mangrove restoration through rewetting	0.07
Coastal wetland restoration	0.20–0.84
Reduced degradation or conversion of river corridors	-
River corridor restoration	-
Improved management of lakes and reservoirs	-

5.2.1. Mitigation measures in wetlands

Conserving and restoring wetlands, including peatlands and coastal wetlands, is a critical climate mitigation strategy. Wetlands have one of the highest stores of soil carbon in the biosphere, storing more than 30% of the estimated global carbon emissions (Nahlik & Fennessy, 2016). Despite covering about 7% of the world's surface, wetlands are considered the largest terrestrial carbon sinks due to their carbon sequestration capacity for a longer time scale in the past as well as future potential (Ramsar Convention on Wetlands, 2018) (Mitsch & Gosselink, 2015). The vegetation in marshes (minerotrophic wetlands dominated by herbaceous plants) and swamps (wetlands dominated by arboreal vegetation), through the process of photosynthesis, captures carbon dioxide and fixes it as organic matter in leaves, stems, and roots. Much of this organic matter eventually becomes incorporated into the soil. The saturated soils of wetlands have slower decomposition than dry soils. When plant productivity exceeds decomposition there is a net accumulation of carbon-rich soil. As a result, wetland soils sequester more carbon per unit volume than terrestrial soils (Bridgham, et al., 2006) (Mazurczyk & Brooks, 2018) (Kolka, et al., 2018) (Moomaw, et al., 2018). While natural wetlands are generally carbon sinks, drainage and other anthropogenic activities can make wetlands net sources of GHG instead. Moreover, although wetlands are considered important sinks for CO_2, almost all freshwater wetlands emit methane which has significantly higher global warming potential than CO_2. Since methane is split relatively quickly by oxidation in the atmosphere, while atmospheric CO_2 continues to be absorbed, the long-term carbon balance of intact peatlands is positive.

Reduce the conversion of wetlands for agriculture, urbanisation, aquaculture or coastal development. As noted, wetlands have some of the highest stores of soil carbon in the biosphere, storing more than 30% of the estimated global carbon emissions (Nahlik & Fennessy, 2016). Hence, maintaining these existing carbon pools in wetlands is important as their loss could significantly increase the concentration of atmospheric CO2 further contributing to the climate crisis (Anisha, et al., 2020). Between 1970 and 2015, the area of the world's natural inland and coastal wetlands declined by ~35% (Ramsar Convention on Wetlands, 2018). About 15% of the world's peatlands have been drained for agriculture, forestry and grazing, leading to release of the carbon stored in their soils and resulting in at least 5% of the total global anthropogenic emissions (Tanneberger, et al., 2017) (Joosten, et al., 2012). Mangrove forests have also experienced a loss of ~4.3% globally in the 20 years preceding 2016, predominantly due to direct human impacts (urbanisation, aquaculture, agriculture) (Global Mangrove Alliance, 2021). Preventing human induced degradation of wetlands that leads to GHG emissions also is important. A meta-analysis on GHG emission from global wetlands due to conversion estimates that at least 0.96 ± 0.22 Gt CO_2-eq of GHGs are released to the atmosphere each year from natural wetlands, accounting for 8.0%–9.6% of the annual global GHG emissions estimated by the IPCC in 2014. Drainage of all wetlands will result in increased emissions of CO2 as the soil organic matter is allowed to decompose. To formulate a mitigation strategy, it is important to understand the context-specific wetland management for emission reduction (Anisha, et al., 2020). The management of the landscape surrounding a wetland, too, plays an important role in reducing emissions, particularly regarding nutrient control. Vegetation structure and level of degradation, tree density and livestock grazing intensity, etc. can impact soil water content, groundwater tables, soil nutrients, soil salinity and several other factors, and thus have significant impact on annual GHG fluxes (Tan, et al., 2020) (Han, et al., 2014) (Herbst, et al., 2013).

Restoration of wetlands to increase the carbon sequestration capacity. However, different types of wetlands sequester carbon and emit GHGs in various ways. As a result, it is essential to understand the sequestration mechanisms and carbon dynamics specific to each wetland type and region to increase the capacity of wetlands to actively sequester carbon long term (Mazurczyk & Brooks, 2018). Hydrological regime, climate, wetland soil type, sediment deposition, decomposition rate and vegetation usually play important roles in a wetland's carbon storage mechanism (Moomaw, et al., 2018) (Zhao, et al., 2019) (Mitsch, et al., 2010) (Zhang, et al., 2002) (Mitsch, et al., 2013) (Mazurczyk & Brooks, 2018). The sequestration rate in temperate and tropical wetlands is 4 to 5 times greater than that found in boreal wetlands (Mitsch, et al., 2013). Examining more specific examples, Zhao et al (2019) studied the effects of water level and inundation duration on CO_2 uptake in Everglades National Park in the US and suggested that there was lower net CO_2 uptake during extended periods of high water, while a study on the impacts of drought conditions on wet soils suggests that decomposition rates and the subsequent carbon storage in peatlands and mineral soil wetlands differ during drought periods (Stirling, Fitzpatrick, & Mosley, 2020). The effects of the hydrological regime vary widely for different types of wetlands based on their region and also is one of the many drivers of carbon sequestration in those wetlands.

In addition to quantity and surrounding land-use, water quality plays a vital role in the emission pattern from freshwater ecosystems. To initiate greater carbon storage, one method would be to slow the rate of decomposition, which is directly related to the biochemical and physicochemical processes (e.g. lack of available oxygen, pH, nutrients, conductivity, etc.) in the wetland (Moomaw, et al., 2018) (Weil & Brady, 2016) (Pinsonneault, et al., 2016) (Mazurczyk & Brooks, 2018). For example, low pH reduces microbial activity which lowers the decomposition rates. Temperature changes affect the microbial and plant activity and influence the carbon storage capacity as well. Decomposition rates increase exponentially with temperature resulting in more carbon release (Moomaw, et al., 2018) (Batson, et al., 2015) (Mazurczyk & Brooks, 2018). Plant productivity and species composition are important in this regard and another proposed strategy to increase carbon storage in a wetland is to increase native species and fungi-based processes by planting perennial species.

Finally, whilst restoring wetlands will serve to mitigate emissions, the opposite is also true as mitigating climate change can also have a positive impact on wetlands (Yuan, at al., 2022). Altered hydrological regimes and more frequent or intense extreme weather events due to climate change will contribute to wetland degradation. Wetland loss and degradation increase GHG emission to the atmosphere, leading to positive feedback on climate change. In fact, global GHG emissions from wetlands are projected to increase by up to 78% under certain climatic conditions (with doubling of atmospheric CO_2) (Gedney, et al., 2019) (Salimi, et al., 2021). It is essential to address the climate change induced changes in wetland management in order to limit GHG emission. When there is a higher rate of decomposition than photosynthesis wetlands emit CO2 and decomposition depends mostly on thermal and hydrologic regimes. For example, drought resulting from higher temperatures might shift the role of peatland from a CO2 sink to a source, although higher temperatures with more water availability (through precipitation or rewetting) can promote more production than respiration and maintain the carbon sink (Vanselow-Algan, et al., 2015) (Salimi, et al., 2021). Shoreline erosion due to sea level rise or frequent and extreme weather events (triggered by climate change) cause losses of salt marshes and mangrove forests.

Wetlands include many different ecosystems, such as peatlands, mangroves, marshes, swamps and bogs. In the following sections, ecosystems with especially high impact on climate mitigation are highlighted.

Peatlands. Restoration and reduced conversion of peatlands have strong long-term mitigation potential. IPCC estimates a yearly emission reduction potential of 0.45–1.22 and 0.15–0.81 GtCO2e yr−1 respectively (IPCC 2022). Peatlands occur in all climate zones, from boreal to tropics. Globally, peatlands cover about 3% of landmass (Gorham, 1991), or approximately 4.2 million km^2 (Xu, et al., 2018). The area of peatlands in temperate and boreal regions are ~3,700,000 km2 storing a total C stock of 415 Pg C (Hugelius, et al., 2020) (Yu, 2012). The extent of tropical peats is about 450,000 km^2, occurring in regions of Asia, Africa, and America, storing about 105 Pg C, about 20% of the C stock in high latitudes (Rieley & Page, 2016) (Dargie, et al., 2017). Peatlands extensive C sink capacity play an important role in the global climate system and have exerted a cooling effect due to their sustained C sequestration over millennia despite their substantial CH_4 emissions (Frolking, et al., 2006) (Kirpotin, et al., 2021). It is estimated that investing in healthy and well-managed peatlands may achieve reductions of at least 5% of global anthropogenic CO_2 emissions (Joosten, 2016). The soils of peatlands at high latitudes generally contain >65% organic matter (Kolka , et al., 2016), while tropical peatland soils contain as much as 99% (Anshari, et al., 2010) (Page, et al., 2011b). The primary constituent of organic matter is elemental carbon (C). Recently, The C stock in tropical peats might be larger as areas of these wetlands may be underestimated (Gumbricht, et al., 2017) (Murdiyarso, et al., 2019).

However, similar to other wetlands, peatlands are being degraded worldwide, causing many peatlands to turn from carbon sinks to carbon sources. Anthropogenic disturbances such as peat harvesting, drainage, peat fires and land use changes, are major drivers that cause peats to become a source of atmospheric carbon dioxide (Andersen, et al., 2013) (Conchedda & Tubiello, 2020) (Hooijer, et al., 2015) (Kolka , et al., 2016) (Loisel & Bunsen, 2020) (Moore et al., 2013). The amount of GHG emissions originating from drained peats globally is about 6% of the global carbon dioxide emissions (Joosten et al., 2012). Under present land use management regimes, Urák et al. (2017) predicted about 25% of peatland areas would degrade by 2050 and contribute as a source to 8% of global carbon dioxide emissions. Using model-based projections of future peatland dynamics, Humpenöder et al. (2020) demonstrate that peatland conservation and restoration of about 60% of currently degraded peatlands is required to return the land system into a net CO2 sink within the 21st century. Peatland conservation and restoration therefore has a large climate mitigation potential and need to be at the heart of climate policies (Menichetti & Leifeld, 2018).

Northern peatlands, Prompt post-disturbance rewetting and revegetating has been shown to substantially reduce adverse climate impacts from degraded peatlands (Nugent, et al., 2019) (Günther, et al., 2020) and to return the C sequestration function of peatlands within a decadal timeframe (Nugent et al., 2018). Restoring natural hydrology and water table depth in peatlands is an important factor for the successful restoration of peatland ecosystem services (e.g. Gaffney, et al., 2020) and has been shown to substantially reduce GHG emissions from drained peatlands (Evans, et al., 2021). However, climate warming is expected to increase northern peatland water losses to the atmosphere through enhanced evapotranspiration putting peatland restoration success (and water security for human and economic purposes) at risk (Helbig, et al., 2020b). Long-term monitoring of GHG emissions from restored peatlands thus provides an important tool to quantify sustained climate benefits and to improve carbon credit schemes for peatland restoration projects (Günther, et al., 2018).

For tropical peatlands, critical measures include restoration of degraded peats and development of sustainable peat management to mitigate and adapt to climate change (Humpenöder, et al., 2020) (Menichetti & Leifeld, 2018), including rewetting. Tropical peat forests showed resilience to natural disturbances of past climate change in the mid Holocene and late Pleistocene (Cole, et al., 2019) (Hapsari, et al., 2018) (Ruwaimana, et al., 2020). Sorensen (1993) estimated that rates of C sequestration in tropical peat swamp forests in Indonesia ranged from 0.01 to 0.03 Gt y^{-1}. Intact tropical peat forests are rich in biodiversity in both terrestrial and associated aquatic habitats, but these are not properly valued for their wider benefits (Thornton et al., 2020). When many peat forests in Indonesia were logged in 1970 – 1990s, selected commercial timber species were removed and sold to earn foreign currency. This deforestation was then followed by conversion to agricultural land rather than allowing for peatland recovery. These anthropogenic disturbances caused ever-lasting cultural and environmental damages that lowered local community livelihoods, carbon stocks, and decreased biodiversity and ecosystem services (Anshari et al., 2022; Gandois et al., 2020; Hoyt et al., 2020).

The Ramsar Convention's latest Global Wetland Outlook (2021) stated that "Rewetting does not reduce emissions to zero: emissions depend on the extent to which the peatland water-table can be raised and kept high", emphasising the need for monitoring, long-term planning and sustainable management. The report also notes that despite high methane emissions at the initial stage of rewetting, the amount decreases over time when peat accumulation restarts and restored peatlands' contribution in global warming is lesser than drained state (Convention on Wetlands, 2021).

Tidal wetlands, often called coastal wetlands, include seagrass meadows, tidal swamps (freshwater and the saline mangrove swamps) and marshes (tidal wetlands without trees). Coastal wetlands may extend to the landward extent of tidal inundation and seaward to the maximum depth of vascular plants (Mitsch & Gosselink, 2015) (Wolanski, et al., 2009). Rates of carbon accumulation are estimated to be 31.2-34.4 $TgCyr^{-1}$ for mangrove swamps, 4.8-87.2 $TgCyr^{-1}$ for salt marshes and 41.4-112 $TgCyr^{-1}$ for seagrass meadows (Howard, et al., 2017) (Kennedy, et al., 2013). Coastal wetlands most affected by freshwater inputs are those in deltas and estuaries where waters of rivers and streams mix with seawater. Today, all three types of tidal wetland habitat face threats that can affect them in different ways from activities in watersheds such as agricultural intensification or urbanisation and nutrient pollution. For example, a lack of sediment supply threatens marshes and mangrove swamps, while reduced water clarity can threaten seagrass meadows. Sustainability of tidal forests and marshes is dependent upon continued vertical accretion of soil to maintain their surface elevation with respect to sea level (Kirwan & Megonigal, 2013), which is expected to rise at increasing rates with global warming (IPCC 2021). Increased sediment supply enhances this process while increased nitrogen from watersheds can cause a decline in production of roots that are key to soil accumulation and the storage of carbon belowground (Deegan, et al., 2012) (Darby & Turner, 2008). Delivery of excess nitrogen affects the ability of the tidal wetland to mitigate climate warming – as microbial activity can transform some of it to nitrous oxide, a greenhouse gas with 265 times the Global Warming Potential of CO_2 (on a 100-year timescale) (Roughan, et al., 2018) (Myhre et al. 2013). Nutrients are supplied to coastal waters from watersheds where sources are agriculture, sewage and run-off from urban land.

Upstream dams reduce the level of suspended sediment released from watersheds, even "mini dams" for hydropower which are purported to have less of an environmental impact can reduce sediment loads. With respect to coastal wetlands, mini dams have little advantage as they still retain sediments and in multiple numbers would have a considerable cumulative impact – akin to the situation of the small dams built to power mills in the US northeast. Even as old dams are being removed in places such as the US, new ones are being constructed and continue to be planned in other regions such as in Mexico. Many environmental and social factors are addressed when assessing the impact of dams, but generally these assessments have not included impacts on tidal wetlands. Promoting awareness of the links between sediment retention and loss of tidal wetland carbon sinks along with the potential for obtaining carbon credits as an alternative income source may encourage more balanced judgements when selecting sites for new dams.

Inland mineral-soil (IMS) wetlands or freshwater mineral-soil wetlands (FMSW) wetlands, include freshwater marshes and freshwater swamps. IMS wetlands account for approximately 39% of the total wetland area globally (Badiou, 2017). These freshwater wetlands have significant carbon stocks too due to their high productivity and waterlogged condition (Bernal & Mitsch, 2012) (Mitra, et al., 2005). Carbon sequestration in IMS wetlands occurs when in situ biomass production exceeds the decomposition rates (Mazurczyk & Brooks, 2018) (Bridgham, et al., 2006) (Moomaw, et al., 2018). Like peatlands, IMS wetlands also play an important role in climate change mitigation. Rates of C sequestration in peatlands is low compared to IMS wetlands (Bernal & Mitsch, 2012) (Zhang, et al., 2016) (IPCC, 2014). A study on the IMS wetlands in the United States Great Plains suggests that most of these organic soil carbon stocks were held in herbaceous freshwater mineral soil wetlands and the rest was found in woody freshwater mineral soil wetlands (Byrd, et al., 2015). Carbon stocks in IMS wetlands vary from 12 to 557 t C per hectare, depending on the type of wetland and climate (Bernal & Mitsch, 2008) (Page & Dalal, 2011) (Ausseil, et al., 2015). CO_2 and CH_4 fluxes from IMS wetlands vary depending on the hydrology, soil wetness, land use type (e.g., disturbed or restored), sediment texture, and vegetation (Batson, et al., 2015a) (Hondula, et al., 2021) (Pfeifer-Meister, et al., 2018).

Research on seasonally inundated forested IMS wetlands reveals that inundated soils switch from methane sources to sinks depending on water level, soil moisture, and the direction of water level change (rising and falling). In fact, it is reported that methane emissions are associated with inundation extent and duration, but not frequency or depth and emissions are higher when water levels are falling (Hondula, et al., 2021). An increase in CO_2 emission is also observed with soil drainage and emissions are reduced by 49% under long term waterlogged conditions (Tete, et al., 2015). Significant N_2O emissions are also associated with frequent drying of wetlands (Pennock, et al., 2010) (Badiou, 2017). Frequent wetting and drying events in IMS wetlands result in increased CH_4 emissions compared to static water level conditions (Hondula, et al., 2021) (Tete, et al., 2015) (Badiou, 2017) (Malone, et al., 2013).

It is a common practice to drain IMS wetlands as part of the preparation of land for agriculture, grazing, and forestry. A lower water level due to drainage leads to higher rate of decomposition, resulting in reduced carbon stocks. (Page & Dalal, 2011). Land conversion results in loss of stored carbon in soil through mineralization, which was otherwise protected due to the anaerobic conditions (Mitra, et al., 2003). Many other anthropogenic activities such as levee, dam and dike construction, irrigation, flow manipulation for water supply and wildlife management can significantly alter the hydrology of the IMS wetlands within the landscape (Mitsch & Gosselink, 2015) (IPCC, 2014). Several studies demonstrate an increase in CH_4 and N_2O emissions due to increased nutrient loading from anthropogenic activities and land use (Silva, et al., 2016) (Gonzalez-Valencia, et al., 2014).

Restoring and rewetting IMS wetlands is beneficial not just for the protection of the carbon stocks but also because of multiple ecosystem services (such as water quality control, flood risk reduction, etc.) received from wetlands (Global Water Partnership, 2018) (UNESCO & UN-Water, 2020) (Gleason, et al., 2009) (Richards & Craft, 2015). The soil carbon accumulation and sequestration rates are much higher in natural unaltered IMS wetlands compared to restored wetlands, but restored IMS wetlands have, over the long term, the potential to have regain a carbon stock similar to natural wetlands depending on factors such as hydrology, vegetation, soils, and land use (Tangen & Bansal, 2020) (Bruland & Richardson, 2005). Many studies suggest that CO_2 contributes the most to the total GHG emission profile from restored IMS wetlands, while CH_4 and N_2O contribute much less. Soil saturation has been identified as a key limiting factor in CH_4 and N_2O production in the restored wetlands (Nahlik & Mitsch, 2010) (Gleason, et al., 2009) (Phillips & Beeri, 2008) (Richards & Craft, 2015). Studies suggest that restored and created IMS wetlands have higher carbon sequestration rates and shorter time period in transitioning from a net source to net sink than many other restored/created ecosystems (Badiou, 2017) (Euliss Jr, et al., 2006).

Knowledge and data gaps in the mitigation potential of wetlands

- Conservation and restoration of wetlands in a country would require a national inventory of wetlands that includes soil type, climate zone, wetland type, size, vegetation composition, hydrological regime and management practices. Many countries in the world either do not have a national wetland inventory or are still in an initial stage of developing a national wetland inventory.
- GHG emissions from disturbed wetlands persist long after a wetland is restored or replaced by a mitigation wetland. The long-term carbon sequestration potential of restored wetlands eventually increases with time.
- Conservation and restoration of wetlands can have socio-economic trade-offs (see section 5.5). There needs to be a framework that can be used to assess potential trade-off scenarios.
- There is substantial uncertainty regarding the spatial extent of tropical peatlands (e.g. Brazilian peatlands) and associated carbon stocks. More field data is needed to reduce these uncertainties in order to reduce peatland loss and enhance peatland protection.
- The Ramsar Convention on Wetlands provides guidelines regarding peatland restoration. However, there is limited knowledge about how to restore degraded peat based on the hydrological system, drainage condition, types of peat soil, climate, and land uses.
- There are very limited examples of long-term sustainable management of restored peatlands while utilising their ecosystem services, which limits the capacity of informed decision-making. Long-term climate change impacts on peatland-based mitigation efforts are not well understood, rendering any assessment of their sustainability difficult.

- Build on emerging research on the impact of thawing permafrost regions, and develop guidance on mitigating large-scale carbon and methane release

[Placeholder - Box addressing the issues of permafrost]

5.2.2 Mitigation measures in rivers and streams

River systems can store a significant portion of terrestrial carbon, but due to lack of research and data the estimated mitigation potential is not known. Still, inland waters are increasingly recognized as a significant source of GHG emissions (Zhang, et al., 2021), while riverine floodplains have been acknowledged for their carbon storage (Sutfin, Wohl, & Dwire, 2016). Rivers and streams do not just connect the carbon stocks of land and sea (Ran, et al., 2015), but are also biogeochemical integrators in landscapes, both receiving and processing carbon, nitrogen and phosphorus and other biologically active elements (Crawford, et al., 2017).

Enhanced carbon storage in river systems. River systems are often referred to as river corridors which mainly includes the active channel and the riparian zone (floodplain, hyporheic zone) (Harvey & Gooseff, 2015). In a river corridor, organic carbon (OC) is stored in six forms, among which three primary OC reservoirs are i) fallen dead large wood (LW) in the channel and floodplain, ii) standing biomass of riparian vegetation and iii) soil organic carbon (SOC) which is technically the OC on and beneath the floodplain surface (Figure 5.1). Large fallen wood, with their long residence times in the river streams and the floodplain, stores OC and delivers particulate organic matter (POM) to the channel and the floodplain. Vegetation is also a significant reservoir of OC. However, floodplains are the most critical since they do not just support the biomass growth that is a source of LW, they facilitate the transport, accumulation, retention, and breakdown of organic matter (OM) received from the channel and the riparian vegetation (Sutfin, et al., 2016) (Wohl, et al., 2017) (Robertson, et al., 1999). A recent evaluation of carbon sinks within Amazonian floodplain lakes, estimates that the accumulation rates of carbon may exceed rates of emission from the river system (Sanders, et al., 2017).

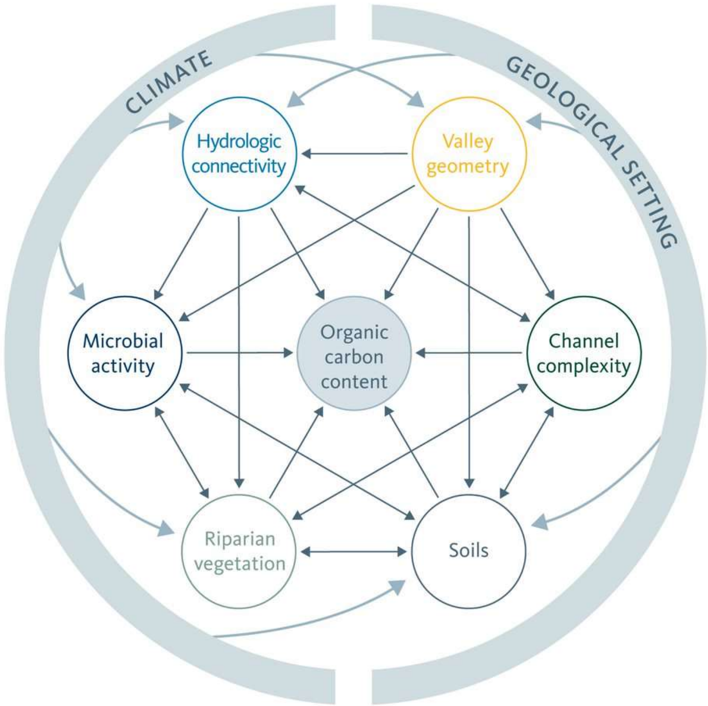

Figure 5.1. A conceptual diagram of regional and local controls on organic carbon (OC) reservoirs in a river system (adopted from Sutfin et al., 2016)

Several factors determine the carbon storage potential of a river corridor, such as geology, climate, channel complexity, valley geometry, hydraulic connectivity, microbial activity and riparian vegetation. As a river moves through different landscapes, above mentioned factors influence the travel time and retention of water, sediment, and OC. For example, a high degree of channel complexity increases the residence time of water, sediment, and POM, facilitates breakdown of OM and filters excess nutrients and DOC from surface and shallow subsurface waters (Sutfin & Wohl, 2017) (Sutfin, et al., 2016) (Wohl, et al., 2017).

The surface and shallow subsurface of the floodplain host a large reservoir for OC as soil organic carbon (SOC). For both small and large rivers, carbon storage is predominantly in the floodplain soil. During overbank flooding floodplains also act as sinks for inorganic, organic, dissolved, and particulate fractions of both nitrogen (N) and phosphorus (P) (Noe & Hupp, 2005) (Wohl, et al., 2017). Long-term carbon storage in the floodplain is determined by the source and form of OC as well as the residence time of the floodplain sediment. Longer residence time enables the retention of sediments and OC, eventually sequestering carbon. Once stored in the floodplain soil, even dissolved organic carbon (DOC) and POM take many years to travel through the river network (Sutfin, et al., 2016) (Cierjacks, et al., 2010). This function of floodplains has been observed in different ecoregions, such as mountainous floodplains and tropical semi-arid lowland floodplains (Omengo, et al., 2016) (Sutfin & Wohl, 2017) (Scott & Wohl, 2018).

In addition, hydrologic connectivity also impacts the carbon sequestration potential of floodplains. Hydrologic connectivity exists longitudinally within channels, laterally between floodplains and channels, and vertically between surface water, hyporheic flow, and groundwater. While lateral and longitudinal hydrologic connectivity facilitate the transport, accumulation, retention, and breakdown of OM, lateral and vertical connectivity, on the other hand, facilitate saturated conditions in floodplains which limit decomposition of OM, microbial metabolism, and mineralization of SOC. Transport of OM from catchments happens longitudinally and then is transported laterally between floodplains and river channels. Increased carbon accumulation and storage are facilitated by increased lateral and vertical hydrologic connectivity. When the lateral connectivity between stream and floodplain is interrupted, there is decreased retention of water and sediment which results in reduced carbon sequestration (Sutfin, et al., 2016).

Several anthropogenic activities affect the carbon storing capacity of floodplains, with one key one being disconnecting floodplains from the active channel through various activities. Some common examples would be constructing levees, embankments, and bank stabilisation, conversion of floodplain into agricultural land, urban expansion, etc. (Shen, et al., 2021) (Robertson, et al., 1999) (Noe & Hupp, 2005). Construction of levees and embankments confine the active channel and alienate the floodplain, limiting the overbank flows which lowers the rate of carbon deposition and sequestration (Wohl, et al., 2017). Flow regime changes through daming, dredging, straightening and/or bank stabilisation can alter the quality of in-channel organic matter and increase downstream fluxes (Sutfin, et al., 2016) (Robertson, et al., 1999).

Greenhouse Gas Emission from Rivers. Recent studies on GHG emissions from inland waters are revealing emission rates higher than previously estimated (Raymond, et al., 2013) (Saunois, et al., 2020) (Tian, et al., 2020). A study by Bastviken et al. (2011) had suggested that methane emissions from inland waters can offset 25% continental carbon sink and hence play important roles in the global carbon budget. The methane emission from inland water systems is estimated to range from 117–212 Tg $CH_4 yr^{-1}$, nearly an order of magnitude greater than the initial estimate of 1.5 Tg CH_4 yr^{-1} (DelSontro, et al., 2018) (Saunois, et al., 2020). In the case of N_2O, rivers and streams can be considered a significant source, depending on the organic matter and nutrient availability and other water quality parameters such as temperature, dissolved oxygen (DO), and pH (Quick, et al., 2019) (Stanley, et al., 2016) (Zhang, et al., 2021). Although the N_2O emission rate from rivers started to decline from 2010–2016 due to decreased use of nitrogen fertilisers, a fourfold increase was seen in global riverine N_2O emission in 2016 (291.3 ± 58.6 Gg N_2O-N yr^{-1}) compared to 1900 (70.4 ± 15.4 Gg N_2O-N yr^{-1}) (Yao, et al., 2020) (Maavara, et al., 2019) (Seitzinger, et al., 2000).

Although there is evidence that rivers are emitting GHGs, there is no comprehensive knowledge about the drivers of emission, the patterns and the variability. The understanding of emission from rivers is still constrained by a relatively small number of observations scattered around the world. These observations vary in measurement and upscaling methods, significant spatial and temporal in fluxes and uncertainties in the global area of the rivers (Allen & Pavelsky, 2018) (Zhang, et al., 2021) (Natchimuthu, et al., 2017) (Crawford, et al., 2017) (Maavara, et al., 2019). The table 5.1 below synthesises the findings.

Table 5.1: Synthesis of studies on riverine emission

Study site	GHG considered	Key findings	Source	Study approach

Chaohu Lake basin, China	N2O, CH4, CO2	-Urban rivers are emission hotspots (compared to forested and agricultural rivers) -Large nutrient supply and low oxygen levels drive the relatively high emission from urban rivers	(Zhang, et al., 2021)	This study investigates spatial variability of N_2O, CH_4, CO_2 emissions from river reaches that drain from different types of landscapes (i.e., urban, agricultural, mixed, and forest landscapes).
Sweden	N_2O	-Agricultural and forest streams have comparable N_2O fluxes despite higher total Nitrogen (TN) concentrations in agricultural streams -The percent saturation of N_2O in the streams is positively correlated with stream concentration of TN and negatively correlated with pH. The differentTN concentrations but similar N_2O concentrations in both the types of streams have been attributed to the low pH (<6) in forest soils and streams.	(Audet, et al., 2020)	This study analysed a data set comprising approximately 1,000 stream N_2O concentration measurements from agricultural and forest streams in Sweden covering temperate to the boreal zone, especially low-order streams.
United States	CO_2	- Streams and rivers in the US are supersaturated with carbon dioxide when compared with the atmosphere, emitting 97±32 Tg carbon each year. -The correlation between precipitation and CO_2 evasion is stronger than discharge and evasion due to the expansion of river surface area with greater delivery of water through precipitation and higher flushing and delivery of soil and riparian/wetland CO_2.	(Butman & Raymond, 2011)	The study is done on total conterminous US stream/river surface area of 40,600 km2.
Meuse River, Belgium	N_2O, CH_4, CO_2	- Surface waters are over-saturated in CO_2, CH_4, N_2O, acting as source of GHG to the atmosphere - Highest GHG fluxes observed during low water - Highest GHG fluxes observed in agriculture dominated catchments	(Borges, et al., 2018)	The study includes four seasonal surveys covering 50 stations, from yearly cycles in four rivers of variable size and catchment land cover, and from 111 groundwater samples.
Tibetan Plateau	N_2O, CH_4, CO_2	-The correlation between the precipitation and CO_2 emissions is stronger than that with Dissolved Organic Carbon (DOC) concentrations and water temperature (due to greater flushing and delivery of soil and riparian/wetland CO_2 to streams and rivers). -Positive trend in CH_4 concentrations with the increased DOC concentrations was observed, indicating that water temperature placed a certain influence on driving pressure of CH_4	(Qu, et al., 2017)	The study did one-time sampling from 32 sites in rivers of the Tibetan Plateau during2014 and 2015.

		increased in anaerobic decomposition. -Partial pressures of N_2O were correlated with dissolved nitrogen and were higher in mainstreams of the Tibetan rivers than those in tributaries due to anthropogenic activities around the mainstream.		
Sub-Saharan Africa	N_2O, CH_4, CO_2	- Riverine carbon dioxide and methane emissions increase with wetland extent and upland biomass - a positive relationship was found between CO_2 and CH_4 flux and precipitation across the region, with the exception of two Malagasy rivers	(Borges, et al., 2015)	The study is based on 12 rivers in sub-Saharan Africa, including seasonally resolved sampling at 39 sites, acquired between 2006 and 2014
Amazonian Basin	CH_4	-Biological oxidation in large Amazonian rivers is a significant sink of CH_4, representing up to 7% of the global soil sink. -The capacity for methane oxidation (MOX) can vary widely across various river types and hydrologic regimes. -The future river MOX process might be sensitive to environmental change, adding to the list of important climate feedback on natural greenhouse gas emissions.	(Sawakuchi, et al., 2016)	The study examines the cycling and flux of CH_4 in six large rivers in the Amazon basin, including the Amazon River in the year 2012, during high water and low water seasons. Methane oxidation (MOX) rate has been studied. MOX reduces the diffusive flux of CH_4 in the rivers.
Amazon and Congo	CH_4 and CO_2	-The pressure of CO_2 and CH_4 concentrations significantly increased from the mainstem to the small tributaries in both the rivers. - The analysis indicated a stronger contribution of CO_2 production from anaerobic organic matter degradation compared to aerobic respiration, which is speculated to be related to Carbon processing within the wetlands in vicinity.	(Borges, et al., 2015)	This study compares the CO_2 and CH_4 distributions in lowland river channels of the two largest rivers in the world and in the tropics, the Amazon (n = 136) and the Congo (n = 280), using a dataset of concurrent CO_2 and CH_4 concentration measurements in river channels

Despite a lack of coherent and generalised knowledge available to explain rivers emissions, and how to minimise emission from the rivers, there are some observations common across several studies. Nutrient loading and organic matter delivery, either due to anthropogenic activities (urbanisation or agriculture) or due to natural causes (vegetation or wetlands), are observed to increase river saturation with CO_2, CH_4 and N_2O. But it is a combined impact of multiple factors such as geomorphologic and hydrologic conditions, temperature, alternative electron acceptors, pH, etc. that can influence the emission of the GHGs. For example, Stanley et al. (2016) illustrates how the concerted impact of several factors influence methane emission in rivers (Figure 5.2) (Stanley, et al., 2016). Some studies found strong correlation between precipitation and CO_2 emission due to greater flushing and delivery of soil and riparian/wetland carbon to streams and rivers (Borges,et al., 2015) (Qu, et al., 2017) (Butman & Raymond, 2011). Borges et al (2015) attempted to draw parallels between two major rivers in the tropics and concluded that "*dynamics of dissolved* CH_4 *in river channels are less straightforward to predict and are related to the way hydrology modulates the connectivity between wetlands and river channels*." In fact, the main stems and tributaries of the same river tend to emit differently, and the emission rates tend to change based on the stream orders (Borges, et al., 2015) (Qu, et al., 2017) (Zhang, et al., 2021) (Audet, et al., 2020) (Raymond, et al., 2013).

[Placeholder: Figure 5.2]

Figure 5.2 Conceptual framework of controls on methane production and persistence in rivers. Controls mentioned here are geomorphology, hydrology, organic matter (OM), temperature (Temp), terminal electron acceptors (TEAs), and nutrients (Stanley, et al., 2016).

The following river system mitigation measures can be relevant to consider in future climate mitigation planning and implementation:

- **Connecting rivers to floodplains**: Construction of levees and embankments confine the active channel and alienate the floodplain as well as limiting the overbank flows, which lowers the rate of carbon deposition and sequestration in floodplains. Maintaining the lateral connectivity and ecosystem integrity in riparian areas can play an important role in increasing the carbon pool in the floodplains. Floodplains work as buffers and reduce nutrient loading to channels, which can be helpful with emission reduction.
- **Limiting channel alterations**: It is important to protect and restore the physical complexity of river corridors that enables carbon storage. Channel alterations through dredging, straightening and/or bank stabilisation can alter the quality of in-channel organic matter and increase downstream fluxes. Maintaining rivers' lateral and vertical hydrologic connectivity enhances carbon sequestration potential compared to longitudinal connectivity.
- **Limiting nutrient and organic matter loading in rivers**: Nutrient loading and organic matter delivery, either due to anthropogenic activities (urbanisation or agriculture) or due to natural causes (vegetation or wetlands), are observed to be making the rivers saturated with CO_2, CH_4 and N_2O. Monitoring, managing and limiting nutrient and organic matter loading in rivers can reduce GHG emission from rivers. Connecting rivers to flood plains also reduces nutrient loading in channels, which can be helpful with emission reduction.
- **Context-specific monitoring**: River corridors' carbon sequestration potential is dependent on regional and local controls, such as geology, climate, hydrology, geomorphic characteristics, etc. Also, for emissions, the mainstream and tributaries of the same river tend to emit differently, and the emission rates tend to change based on the stream orders. Studies also suggest the variation in emission in different river reaches is related to their proximity to urban, agricultural or forested landscape. It is unlikely that there will be a generalised solution that fits all rivers, and management plans should be context specific.
- **Watershed-scale management approach**: Whether for enhanced carbon sequestration or emission reduction, management approaches and decisions should be taken from the watershed scale. Carbon fluxes in rivers are affected by grazing, cropping on floodplains (nutrient source) or soil erosion due to removal of native species (POC loading). Rather than treating river systems as isolated segments, a watershed-scale management that addresses the complex dynamics of the catchment, can yield better outcomes.

Knowledge gaps in the mitigation potential of rivers and streams

Significant knowledge gaps remain, however, which are critical to address in order to realise the full mitigation potential of rivers and streams. Most significantly:

- The understanding of the spatial extent and magnitude of changes in riparian soil organic carbon content and biomass is currently only based on a handful of studies focused on some regions. There is no global-scale comprehensive understanding of how the historical and ongoing riparian modification impacts carbon dynamics in the rivers systems.
- The interactions among water, sediment yield, flow regime, biomass and primary productivity, soil moisture and soil organic carbon are complex and nonlinear. Future climate change would likely impact river corridors around the world in different ways. We need a better understanding of the mechanics of the carbon fluxes and their transformations in the river corridors.
- There is evidence that rivers are emitting GHGs, but there is no concrete organised knowledge about the drivers of emission, the patterns, and the variability. The understanding of emission from rivers is constrained by a relatively small number of observations scattered around the world, varying measurement and upscaling methods, significant spatial and temporal variation in fluxes and uncertainties in the global area of the rivers.
- There are few studies that explored the emissions of all three major GHGs. Estimating the emission potential of rivers without incorporating the emission of all three gases would be incomplete.
- In several river basins, the source of pollution (for example, industries) and the point of sequestration (for example, the river corridor) may not be under the same jurisdiction. Policies need to consider such gaps and find a way to minimize them.

5.2.3 Mitigation measures in lakes and reservoirs

Lakes, either natural or created with dams (reservoirs), play a key role in global carbon cycling despite taking up less than 4% of earth’s non-glaciated land area (Verpoorter, et al, 2014) (Raymond, et al., 2013) (Beaulieu, et al., 2020) (Bastviken, et al, 2011) (DelSontro, et al., 2018) (Stanley, et al., 2016). Lakes and reservoirs can trap land-derived carbon (through carbon burial) in their sediments (Mendonça, et al., 2017). Mendonça et al. (2017) recommended considering lakes and reservoirs as a “new sink” for land-derived organic carbon, particularly because organic carbon is more efficiently preserved in inland water sediments than in other depositional environments (such as soils) and sediment delivery to the sea has decreased. Cole et al (2007), too, acknowledges the high carbon burial potential for reservoirs due to high sedimentation, but also warns about the unknown fate of reservoir sediment post dam removal.

However, lakes and reservoirs also have high methane (CH_4) production (compared to CO_2) in nutrient-rich (eutrophic) sediments (Beaulieu, et al., 2020) (Berberich, et al., 2020) (DelSontro , et al., 2018). Despite considerable rates of carbon burial, eutrophic freshwater with carbon carrying sediments can become greater net GHG sources at a centennial time scale. This is a key concern, considering that methane’s global warming potential is 28 times greater than CO_2 over a 100-year time horizon (Myhre et al. 2013). In fact, a study by DelSontro et al (2018) shows GHG emissions from lakes and reservoirs are equivalent to 20% of CO_2 emission from global fossil fuel every year (DelSontro , et al., 2018).

Lake size, depth, sedimentation rates, dissolved organic carbon concentration and productivity rate (lake's ability to support plant and animal life defines its level of productivity, or trophic state) alongside environmental factors such as temperature and precipitation have been identified as the drivers of GHG emission in reservoirs and lakes (DelSontro, et al, 2018) (Sanches, et al, 2019) (Beaulieu, et al, 2019) (Waldo, et al, 2021) (Berberich, et al, 2020). Shallow and tropical lakes and reservoirs have high emission rates of the GHGs, but methane is of most importance due to its link with lake and reservoir productivity and its high global warming potential (Gunkel, 2009) (DelSontro, et al, 2018) (Sanches, et al, 2019). Higher watershed area-to-surface area ratio of reservoirs usually result in high sediment and nutrient loading from the surrounding catchment compared to natural lakes, triggering greater production rates and carbon burial, increasing methane generation in the system (Berberich, et al., 2020). Per surface area mean N_2O emission rates are substantially lower for natural lakes than for reservoirs as well (Lauerwald, et al., 2019).

An important concern with lakes and reservoirs is high aquatic productivity in response to nutrient increase, known as eutrophication. There is a significant relationship between freshwater eutrophication and GHG emissions (Li, et al., 2021) (DelSontro, et al., 2018) (Mendonça, et al., 2017) (Sanches, et al., 2019) (Berberich, et al., 2020). In fact, there is a positive feedback loop between eutrophication in lakes and reservoirs and GHG emissions, meaning that freshwater eutrophication and GHG emissions are strengthened by each other. To put in simple words, when nutrient loading crosses a critical threshold, submerged plants are gradually replaced by other aquatic macrophytes or algae. Firstly, shift in the dominant primary producer from submerged plants to algae affects the GHG emission since submerged plants more effectively reduce CH_4 emissions. Secondly, algae become the primary dominant producer in the lake or reservoir system, which plays an important role in the freshwater system's emission dynamics. Algae has higher CO_2 uptake rate (compared to other aquatic macrophytes) and can effectively reduce CO_2 emissions. On the other hand, harmful algal blooms (HABs) cause high production of CH_4 and N_2O. Warmer temperatures increase algal production, resulting in more CH_4 and N_2O emission (Plouviez, et al., 2019) (Su, et al., 2019) (Burlacot, et al., 2020). This increased emission further contributes to climate change and increasing temperature (Li, et al., 2021). Li et al. (2021) illustrated the positive feedback loops between freshwater eutrophication and GHG emissions (Figure 5.3). Productivity of inland waters is projected to increase in the coming decades, due to both increased mean temperature and increased nutrient loading, which makes the climatic impact of HABs an important concern (Beaulieu, et al., 2019). Watershed-scale soil erosion control and nutrient reductions may help reduce GHG emission from lakes and reservoirs (Berberich, et al., 2020).

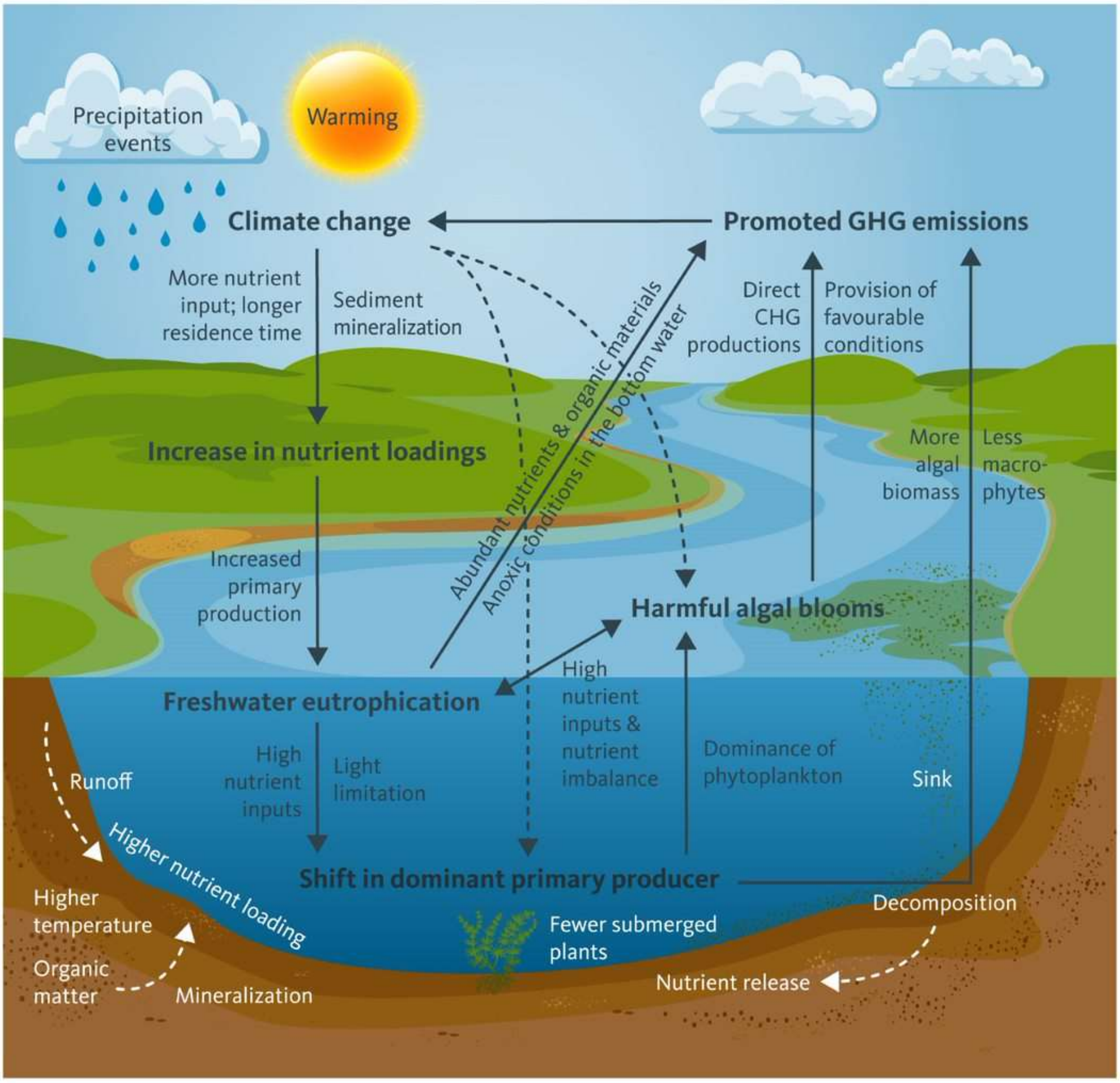

Figure 5.3 The positive feedback loops between freshwater eutrophication and GHG emissions (Source: SIWI)

High nutrient and organic matter loading are common factors that influence the emissions from lakes and reservoirs, resulting in increased high aquatic productivity. Lake and reservoir characteristics (depth, temperature, sediments, rooted aquatic macrophytes) and catchment attributes (land use, terrestrial net primary production, and human activities) are also driving factors. One key measure that is physically easy to implement for lakes and reservoirs is nutrient and organic matter control, but there may be other factors that increase difficulty. Although reservoirs are increasingly recognised to emit significant amounts of GHG, there are thousands of small and large-scale dams, and more are being constructed. It is important to discuss and implement the measures that can reduce emission in already existing reservoirs and how to reduce emissions in new reservoirs.

The following mitigation measures in lakes and reservoirs can be relevant to consider in future climate mitigation planning and implementation:

- **Nutrient and organic matter control for eutrophication management**: As noted above, reducing nutrients, primarily nitrogen and phosphorus, and organic matter loading can lower the rate of eutrophication. This can be done by reducing use of fertiliser, minimising nutrient loads in the catchment, phosphate stripping at sewage treatment works, and installing vegetated buffer strips adjacent to water bodies that trap eroding soil particles, etc. (Paerl, et al., 2020) (McCrackin, et al., 2017) (Berberich, et al., 2020). These bring the added benefit of improved water quality (Li, et al., 2021) (Yan, et al., 2021). Nutrient loading control can be a long withstanding measure for ecological restoration and emission reduction.
- **Managing drawdown, operating level and downstream emission in reservoirs**: Fluctuating water tables and shallow littoral areas[1] produce considerably more methane in reservoirs than natural lakes or other surface waters (Harrison, et al., 2017). The water level management should be optimised to minimise methane emission from the sediments and the littoral zone. Water-level management, such as avoiding drawdown in the dry season, can be a useful approach to minimise the source of GHGs. Downstream methane emissions from reservoirs can be reduced by selectively withdrawing water from near the reservoirs' surface (where methane concentration is lesser than greater depths) (Yan, et al., 2021) (Keller, et al., 2021) (Harrison, et al., 2017) (Harrison, et al., 2021).
- **Technology for methane management:** Methane emission can be reduced using a methane capture technology (which converts the captured methane into energy) and a technology to increase the dissolved oxygen (such as installing aerating device) in the water (Fearnside, 2007).
- **Management of older dams and dam removal processes**: Dam removal mobilises the sediments, nutrients and organic carbon from the reservoir resulting in a high potential for emission. Dam removal can also affect the downstream river channel by eroding the stream bed and the nutrient-rich sediments. On the other hand, deposited nutrients do not necessarily remain trapped in the reservoir when an old out-of-operation dam is left in place. Due to decreased sediment and nutrient elimination efficiencies, the reservoir can turn into a nutrient source to the surrounding landscape (Maavara, et al., 2019). Hence, management of old dams and dam removal needs to take the remobilization, mineralization and the subsequent emissions of deposited sediment, nutrients and organic carbon into consideration.
- **Conception and planning of new hydropower dams**: The role of hydropower as a clean energy source is increasingly being revisited. Moreover, dams are found to affect river ecosystems, biodiversity, and society, with a potential impact on the emissions from river systems. As mentioned above, emissions can occur not just during the operating years, but also when dams are old or removed, which should be taken into consideration. During the decision-making process for new or rehabilitated dam development, there should be thorough accounting of short- and long-term impacts and the benefits of proposed projects at the conception and planning stage so that emissions can be minimised if the development proceeds (Fearnside, 2007). It is necessary to consider the GHG exchanges before and after the impoundment. The difference between pre- and post-reservoir emissions from the whole river basin indicates the GHG status of the reservoir (UNESCO/IHP, 2008).

[1] The shallow transition zone between dry land and the open water area of the reservoir where aquatic plants grow

Box 5.1: Concerns about Hydropower Reservoirs

Hydropower dams have been under much scrutiny over the last decade, over the discussion of whether or not they can be considered a clean energy source since the reservoirs created by these dams emit globally significant amounts of GHG (Prairie, et al., 2021) (Deemer, et al., 2016) (Tremblay, et al., 2005) (Fearnside, 2006) (Fearnside, 2007). The total annual GHG emissions from global reservoirs amount to 2.3% of total emissions from inland freshwaters (Yan, et al., 2021). Until very recently, global estimates of GHG emissions from reservoirs have been based on the assumption that all reservoirs in a similar climate and region would emit in a similar manner (Harrison, et al., 2021) (Prairie, et al., 2021). Estimation of GHG fluxes in reservoirs has also been focused solely on diffusive gas fluxes until very recently when ebullition fluxes have been considered in the estimation (Deemer, et al., 2016) (DelSontro , et al., 2018) (Harrison, et al., 2021).

Reservoirs emit all three major GHGs, but the estimation of N_2O on a global scale has been very limited due to scarcity of data (Deemer, et al., 2016) (Yan, et al., 2021). Emission of CO_2 and CH_4 happens in four ways: CO_2 diffusion, CH_4 diffusion, ebullition, and degassing, among which CH_4 emission through degassing has been incorporated in the global GHG budget of reservoirs very recently. Recent findings suggest that more methane (CH_4) that leaves the reservoir through ebullition is transported downstream from reservoirs (Harrison, et al., 2021) (Keller, et al., 2021). Organic content and nutrient loading, reservoir sediments, primary productivity, and water temperature are the primary contributing controllers of GHG emission from reservoirs, but emissions can also be impacted by the characteristics of reservoirs (temperature, depth, thermal stratification, and trophic status, etc.) and their catchments (land use, terrestrial net primary production, and human activities) (Yan, et al., 2021) (Prairie, et al., 2021). Reservoir drawdown areas[2] are hotspots for carbon dioxide (CO_2) emissions (Keller, et al., 2021).

Although Deemer et al. (2016) showed that some reservoirs can be CO_2 and N_2O sinks, several other recent studies suggest that reservoirs are a net source of carbon. In their first 2-5 years of construction, newly formed hydroelectric reservoirs emit almost 3-10 times more greenhouse gases compared to the natural lakes of the same size and continue to release CO_2 and CH_4 during the plant operating lifetime (Fearnside, 2006) (Tremblay, et al., 2005). Considering the additional GHG emissions in the drawdown areas, Keller et al (2021) suggest that hydroelectric reservoirs emit more carbon than they bury (Keller, et al., 2021) (Harrison, et al., 2017).

Knowledge gaps in the mitigation potential of lakes and reservoirs

There is a high level of uncertainty and knowledge gaps regarding different aspects of GHG fluxes from lakes and reservoirs. Some of the key knowledge gaps and opportunities include:

- Although reservoirs emit all three major GHGs, few reservoirs have measurement records for all three gases, with the least number of data points being for N_2O (Deemer, et al., 2016).
- There is noticeable variation in the estimation of GHG emission from lakes and reservoirs. The global aerial coverage of reservoirs, including small reservoirs, is not well-documented, which is why different studies used different areas and calculation periods which introduces variation in the

[2] Areas where sediment is exposed to the atmosphere due to water-level fluctuations

estimation of GHG fluxes. In addition, GHG emissions from lakes and reservoirs show high spatial and temporal variability (Ion & Ene, 2021) (Yan, et al., 2021).

- There is no standardised or widely accepted method for GHG emission estimation in reservoirs. Until recently, emission through ebullition and degassing pathways was not incorporated into the total GHG budget estimation. Downstream GHG emissions remain poorly studied although it might be an important pathway of GHG release to the atmosphere too (Keller, et al., 2021) (Yan, et al., 2021).
- There is substantial uncertainty about how the impacts from climate change might affect GHG emission from lakes and reservoirs in the future. GHG fluxes will likely be impacted by potential changes in the reservoirs (e.g., direct inputs, water management) and their watershed (e.g., land use, microclimate) due to climate change (Yan, et al., 2021).
- Only a handful of studies have examined the combined effects of land management change and climate change on nutrient loading, and these have only been focused on individual watersheds. Socioeconomic changes have an important bearing on how landscape management would be altered in the future. This uncertainty makes estimation of future GHG fluxes difficult (Sinha, et al., 2019).

5.3. Co-benefits and trade-offs regarding freshwater-based mitigation

Freshwater systems provide several important benefits for nature and human society, including provision of food and water, water quality improvement, disaster risk reduction, habitat protection, sediment retention and nutrient cycling, economic, cultural and recreational benefits, etc. (Dybala, et al., 2019) (de Groot, et al., 2008) (Doswald & Osti, 2011) (Anisha, et al., 2020). Mitigation measures based upon freshwater systems, for example conservation of wetlands or nutrient loading control, can offer some specific direct and indirect co-benefits. However, it is recognised that some socio-economic, socio-political and development trade-offs would occur if freshwater systems are increasingly managed for GHG reduction and increased carbon sequestration. This section highlights possible co-benefits and trade-offs regarding freshwater-based mitigation measures.

5.3.1 Enhancement of ecosystem services through mitigation measures

Burkhard and Maes (2017) define ecosystem services as the contributions of ecosystem structure and function to human well-being. In simple words, ecosystem services are the benefits humans obtain from the ecosystem. The Millennium Ecosystem Assessment (MEA) identifies these services in four broad categories: a) Provisioning services[3], b) Regulating services[4], c) Cultural services[5] and d) Supporting services[6] (Millennium Ecosystem Assessment, MEA, 2005). Mitigation measures within freshwater systems, such as pollution control, wetland conservation and restoration, hydrology and vegetation monitoring, etc., outlined in section 5.2 of this chapter can enhance the delivery of the ecosystem services across all categories. Enhancement of ecosystem services refers to change in the service that leads to greater benefits for people compared to existing scenarios (Millennium ecosystem assessment, MEA, 2005). Table

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[3] provisioning services entail directly providing goods or services to humans, such as food, water, timber, etc.

[4] regulating services are the benefits provided by ecosystem processes that moderate natural phenomena, such as climate regulation, biological control, soil erosion prevention.

[5] cultural services are the non-material benefits people obtain from ecosystems such as recreational, aesthetic and spiritual benefits

[6] supporting services are the functions or processes that are necessary for the production of other ecosystem services such as water and nutrient cycling, pollination, soil formation, etc.

5.2 outlines some examples of how mitigation measures in freshwater system enhance the ecosystem services in different service categories:

Table 5.2 Enhancement of ecosystem services through freshwater-based mitigation measures

Ecosystem Service Category	Function	Example
Provisioning	Water supply	Pollutant control in rivers and lakes improves water quality which can be used by humans for drinking, swimming, fishing or other activities (Mitsch, 1992) (Dosskey, 2001). Flooded wetlands play a role in groundwater recharge (Gupta, et al., 2020).
	Food	Protected and restored wetlands and well-managed floodplains foster edible plants, shrubs, herbs and animals (Buckton, 2018) (Leaman, 2018).
	Habitat	Protected and restored wetlands, lakes and rivers with less pollutant are usually habitat, breeding ground and refuge for different species of birds, mammals, amphibians, fishes and reptiles (Grizzetti, et al., 2019) (Flores-Rios, et al., 2020).
Regulating	Pollutant control	Protected, restored and/or constructed wetlands play a role in pathogen removal, and nutrient retention, removal and breakdown (Vymazal, 2018) (Mackenzie, 2018).
	Disaster risk reduction	Wetland and floodplain expansion can reduce flood risk through enhanced hydraulic connectivity (Tomscha, et al., 2021) (McInnes, 2018a). Coastal wetlands provide protection from storms and coastal erosion (Millennium ecosystem assessment,MEA, 2005).
	Water quality regulations	Protected and restored wetlands, with their vegetation cover, can trap sediments, remove pollutants and protect rivers and lakes from nutrient overload (Mitsch, 1992) (Mitsch,et al., 2005a).
	Erosion regulation	Vegetated wetlands (swamps and marshes.) trap sediments and regulate erosion (Ford, et al., 2016) (Fagorite, et al., 2019).
	Micro-climate regulation	Wetlands (protected, restored and constructed), alongside rivers and lakes, have a positive effect on the surrounding microclimate with relative cooling impact (Sun, et al., 2012) (McInnes, 2018b).

Supporting	Biogeochemical cycling	Restored wetlands can store elements such as nitrogen, phosphorus, and carbon for long periods in the soil, and also supply these elements to surrounding ecosystems, which is unlikely to happen in a drained condition (Everard, 2018c) (Tomscha, et al., 2021).
	Water storage	Water moving through a protected or restored wetland is often slowed by vegetation on the wetland and this can further promote water retention, infiltration and storage (Carter, 1996) (Feng, et al., 2021) (Millennium ecosystem assessment,MEA, 2005).
	Hydric soil development	Wetland restoration prompts hydric soil[7] development, which enables the growth and regeneration of vegetation that are adapted to grow in saturated/inundated and low-oxygen conditions (Amon, et al., 2005) (Mitsch, et al. 2005a) (Millennium ecosystem assessment,MEA, 2005).
	Biomass production	The nutrients and water retained my floodplains and wetlands aid vegetation in growth and biomass production. Wetland restoration supports native plant species richness (Tomscha, et al., 2021) (Millennium ecosystem assessment,MEA, 2005).
Cultural	Recreation	Nutrient and sediment loading control in rivers and lakes can enhance water clarity which directly and indirectly contributes to recreational benefits, including swimming, boating, fishing, etc. (Angradi, et al., 2018).
	Aesthetics	Enhanced water clarity in rivers and lakes can increase their visual appeal and improved water quality also contributes to enhancement in biodiversity which adds value to the aesthetics (Angradi, et al., 2018) (Papayannis & Pritchard, 2018).

5.3.2 Climate change adaptation and resilience benefits from mitigation measures

Ecosystem-based adaptation[8] (EbA) responses to climate change have increasingly received acknowledgement as a cost-effective, proven and sustainable solution for climate change adaptation. Freshwater systems (rivers, streams, lakes, wetlands) are commonly deemed as key components in EbA (World Bank, 2009) (Colls, et al., 2009) (UNEP & IUCN, 2021). Adaptation benefits received from

[7] A hydric soil is a soil that formed under conditions of saturation, flooding, or ponding long enough during the growing season to develop anaerobic conditions in the upper part.

[8] EbA integrates the use of biodiversity and ecosystem services into an overall strategy to help people adapt to the adverse impacts of climate change (Colls, et al., 2009)

ecosystems are also termed as adaptation services[9]. Freshwater-based climate change mitigation measures are mostly based around protecting and restoring the water bodies to healthy states. Freshwater systems' role in climate change adaptation has been emphasised due to their ability to persist through climate change effects and continue providing ecosystem services (Morelli, et al., 2016) (Lavorel, et al., 2015) (Colls, et al., 2009) (Colloff, et al., 2016). Climate change is predicted to affect freshwater systems, but floodplain ecosystems and well-managed wetlands, even if in a low-diversity state, are likely to persist under climate change and provide adaptation services (Lavorel, et al., 2015). In fact, many areas with large water bodies have persisted through the climatic changes of the Holocene, proving their resilience (Morelli, T. L., et al., 2016). But there are concerns over whether this can be maintained under changing environmental conditions through the intersection of land-uses and climate change.

Climate change is predicted to increase the intensity of extreme precipitation events and the risk of flooding in some parts of the world and intensify drought events in some other parts (Cook, et al., 2018) (Tabari, 2020). Freshwater-based climate change mitigation measures, such as efforts to connect rivers to floodplains and protect and restore wetlands, are recognised adaptation measures for increased flood and drought risk (Lavorel, et al., 2015) (Endter-Wada, et al., 2020) (Opperman, et al., 2009) (Vigerstol, et al., 2021). Protection or restoration of floodplains has the highest potential in mitigating riverine flood risks since it provides for natural storage and diversion in regularly flooded areas (Vigerstol, et al., 2021) (Opperman, et al., 2009).

Seifollahi-Aghmiuni et al (2019) highlighted well-managed wetlands' capability of retaining runoff water and refilling aquifers which helps minimise drought-induced stress on water reservoirs or stresses that occur due to increased temperatures. Endter-Wada, et al. (2020) discussed how riparian wetlands associated with beaver dams can alleviate impacts of wildfire by creating broad and diffused floodplain habitats that are more resistant to burning. As mean earth temperature is on the rise, the cooling effects created by rivers, lakes and wetlands provide adaptive services to both humans (particularly in urban areas) and animals (through creating climate change refugia[10]) (Costanza, et al., 1997) (Chang, et al., 2007) (Morelli, et al., 2016) (Sun, et al., 2012).

In an urban setting, wetlands (including reservoirs, lakes, and rivers) create "urban cooling islands" that maintain lower temperatures in an area compared with its surroundings. In fact, water bodies are relatively more efficient than other types of green spaces due to the higher rate of evapotranspiration (Gober, et al., 2010) (Hathway & Sharples, 2012). Hence, protecting and restoring the urban wetlands can bring both mitigation and adaptation benefits. The cooling effect of water bodies enables the creation of climate change refugia for local populations, wildlife and fisheries. In large water bodies and their surrounding areas (deep lakes and wetlands for instance), more solar energy is used in evaporation than in surface heating, which buffers regional warming (Morelli, et al., 2016). Protection and restoration of riparian wetlands and forested wetlands can enhance the adaptive capacity of different terrestrial species in a warming climate. The hydrologic connectivity between river and floodplain is considered a key predictor of species richness of floodplain invertebrates (Tomscha, et al., 2017). This hydrologic connectivity also enhances climate change

[9] Adaptation services are the ecosystem processes and services that benefit people by increasing their ability to adapt to change. These benefits may come from existing but newly used services from the ecosystems that persist under changing climate, or from novel services supplied by ecosystems that transformed due to change in climate (Colloff, et al., 2016)

[10] climate change refugia are areas relatively buffered from contemporary climate change over time that enable persistence of valued physical, ecological, and socio-cultural resources (Morelli, et al., 2016)

resilience in many species through allowing movement to new areas when current habitats become unsuitable due to climate change (Morelli, et al., 2016) (Cassin & Matthews, 2021).

5.3.3 Nature-based Solutions associated with freshwater ecosystems' mitigation measures

Nature-based Solution (NbS)[11] is an umbrella concept that addresses the utilisation of natural capital in seeking solutions (Cassin & Matthews, 2021). These nature-centric solutions are applicable in different sectors, such as water resources management, disaster risk reduction, water quality control, agricultural technology, and climate change adaptation. NbS are deemed sustainable due to their ability to cope with different conditions without altering much of their structure or functionality (robustness) and also when an environmental condition exceeds a threshold, they can adapt by altering their structure and operating conditions (Folke, 2006) (Mauroner & Anisha, et al., 2021).

NbS are advanced and deliberate applications of using ecosystem services to meet objectives. Floodplain restoration and management, potentially a freshwater-based mitigation measure, is an effective NbS for flood mitigation, biodiversity protection and surface water quality control (Keesstra, et al., 2018) (Perosa, et al., 2021) (Acreman, et al., 2021) (Jakubínský, et al., 2021). Lo et al. (2021) evaluated the flood mitigation potential of floodplain expansion (titled 'Room for the River') compared to three other grey/hard infrastructure solutions (levee extension in variable lengths) on the Nangang River area in Taiwan. The authors considered 'Room for the River' as the best suited flood mitigation measure due to its effectiveness that comes with multiple co-benefits compared to grey solutions that are a single-purpose infrastructure optimised to solve narrowly defined problems (Lo, et al., 2021). Perosa et al. (2021) discussed floodplain restoration as NbS for flood protection in three locations of the Danube catchment in Europe and estimated the benefits in terms of monetized ecosystem services. The study estimated a total gain of ecosystem services worth approximately 5 million USD per year in all three locations combined (Perosa, et al., 2021). Based on a comprehensive review of over 400 case studies on different NbS across the African continent, Acreman et al. (2021) concluded that floodplain wetlands are effective NbS options for flood protection and sediment generation in Africa.

Restored and protected wetlands, even constructed wetlands, are commonly acknowledged as effective NbS for disaster risk reduction, flood management, water quality improvement and climate change adaptation (Keesstra, et al., 2018) (Cabral, et al., 2017) (UNEP, 2014) (Liquete, et al., 2016). In their discussion on the effect of NbS in land management for enhancing ecosystem services, Keesstra et al. (2018) included an example of vegetative sediment trapping measures in Ethiopia where wetlands, along with grassed waterway, were utilised to trap the sediment in its transport path. This provided solutions for widespread soil loss and sediment overload in the lakes and reservoirs downstream and was deemed superior to other options (Keesstra, et al., 2018). Another study in the eastern Free State province of South Africa examined the role of wetlands in disaster risk reduction (such as drought, veld fires and floods) and concluded that well-managed and protected wetlands are effective buffers and can effectively reduce the risk of veld fires, floods and drought, whereas degraded wetlands substantially lack the risk mitigating capacity. The authors emphasised restoring the degraded wetlands and monitoring the ecological state of the protected wetlands in order to utilise wetlands as efficient, cost-effective, community-driven NbS for disaster risk reduction (Belle, et al., 2018).

[11] NbS can be defined as "actions to protect, sustainably manage, and restore natural and modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham, et al. (eds.)., 2016)

NbS are usually multipurpose, able to address different issues and aid other solution approaches while contributing to the safety, health, well-being, livelihoods etc. of local populations (Cassin & Matthews, 2021). UNEP (2014) outlined some NbS for water resource management and compared them against traditional grey solutions (Table 3). In this table, freshwater-based mitigation measures, such as reconnecting rivers to floodplains, wetland conservation/restoration, constructing wetlands and riparian buffers, are observed to be the most multipurpose NbS that can address issues regarding water quality regulation, water supply regulation and extreme weather moderation (UNEP, 2014).

Table 5.3 Nature-based Solutions for water resource management (UNEP, 2014)

Water management issue (Primary service to be provided)		Green Infrastructure solution	Location: Watershed	Location: Floodplain	Location: Urban	Location: Coastal	Corresponding Grey Infrastructure solution (at the primar service level)
Water supply regulation (including drought mitigation)		Re/afforestation and forest conservation	■				Dams and groundwater pumping Water distribution systems
		Reconnecting rivers to floodplains		■			
		Wetlands restoration/conservation	■	■	■		
		Constructing wetlands	■	■	■		
		Water harvesting*	■	■	■		
		Green spaces (bioretention and infiltration)			■		
		Permeable pavements*			■		
Water quality regulation	Water purification	Re/afforestation and forest conservation	■				Water treatment plant
		Riparian buffers		■			
		Reconnecting rivers to floodplains		■			
		Wetlands restoration/conservation	■	■	■		
		Constructing wetlands	■	■	■		
		Green spaces (bioretention and infiltration)			■		
		Permeable pavements*			■		
	Erosion control	Re/afforestation and forest conservation	■				Reinforcement of slopes
		Riparian buffers		■			
		Reconnecting rivers to floodplains		■			
	Biological control	Re/afforestation and forest conservation	■				Water treatment plant
		Riparian buffers		■			
		Reconnecting rivers to floodplains		■			
		Wetlands restoration/conservation	■	■	■		
		Constructing wetlands	■	■	■		
	Water temperature control	Re/afforestation and forest conservation	■				Dams
		Riparian buffers		■			
		Reconnecting rivers to floodplains		■			
		Wetlands restoration/conservation	■	■	■		
		Constructing wetlands	■	■	■		
		Green spaces (shading of water ways)			■		
Moderation of extreme events (floods)	Riverine flood control	Re/afforestation and forest conservation	■				Dams and levees
		Riparian buffers		■			
		Reconnecting rivers to floodplains		■			
		Wetlands restoration/conservation	■	■	■		
		Constructing wetlands	■	■	■		
		Establishing flood bypasses		■			
	Urban stormwater runoff	Green roofs			■		Urban stormwater infrastructure
		Green spaces (bioretention and infiltration)			■		
		Water harvesting*	■	■	■		
		Permeable pavements*			■		
	Coastal flood (storm) control	Protecting/restoring mangroves, coastal marshes and dunes				■	Sea walls
		Protecting/restoring reefs (coral/oyster)				■	

5.3.4 Trade-offs in utilising freshwater-based mitigation

Some of the major drivers of wetland degradation and loss have been urban expansion and infrastructure development, conversion for agriculture and grazing, land-use change, water withdrawal, pollutant overload, etc. (Galatowitsch, 2018) (Mitsch, 2005b). The conversion and restoration measures and pollutant control measures that are tied to climate change mitigation seem to require trade-offs with many of these aspects that had replaced and degraded the wetlands in the first place. In many countries, development is often centred on economic growth along with infrastructure development intended to facilitate growth, and other values are not given a similar priority especially if they are seen as being conflicting. Without economic values of ecosystem services provided by mitigation measures being considered more commonly, implementing freshwater-based mitigation measures might be perceived as requiring major trade-offs with economic and infrastructural growth (Mauroner & Anisha, et al., 2021) (Rozenberg & Fay, 2019) (World Bank, 2012). For example, increasing environmental flow to a degraded wetland or floodplain for restoration purposes might have competition with irrigation water for agriculture (de Groot, et al., 2008). In the section below we discuss some of the trade-offs and competing interests in implementing freshwater-based mitigation measures:

- **Tradeoffs among the ecosystem services themselves**: As discussed in section 5.2 above, freshwater-based mitigation measures deliver a wide range of ecosystem services. But many wetlands in the world are valued and utilised mainly for their provisioning services, directly delivering food, water, timber and other products to the communities as opposed to the wider spectrum of benefits. The importance of the supporting and regulating services can be overlooked by decision-makers, although these services are essential in strengthening the provisioning services received, not just from the wetlands but from the other elements in the ecosystem (such as forest and biodiversity). Mitigation measures, emphasising protection and restoration of healthy ecological state of wetlands, should help support calls of minimising the overexploitation of wetlands, which might seem like a trade-off with how the wetland has been traditionally utilised (Mandishona & Knight, 2022).
- **Trade-offs between floodplain protection and agriculture:** Encroachment of agricultural land into riverine floodplains is common around the world (Verhoeven & Setter, 2010) (Pullanikkatil, et al., 2020). Protection, restoration, and expansion of floodplain wetlands for climate change mitigation, even with all of their benefits in sediment retention, water quality improvement and pollutant control, stand as a trade-off with agricultural expansion, which is critical for present and future food security. Nonetheless, when floodplain wetlands are drained and degraded, their potential to deliver regulating and supporting ecosystem services become limited, which might affect agricultural provisioning services. A study conducted on the Hula Wetland in Israel illustrates as an example of how degrading wetland conditions resulted in declining agricultural production services over time (Cohen-Shacham, et al., 2011).
- **Trade-offs in urban floodplain restoration:** Floodplains in urban areas are often converted into human settlements, industrial settlements, and recreational spots, especially where most floodplains have been disconnected from the rivers. Hence, mitigation measures that entail connecting rivers to floodplains and restoration of floodplains can be seen as having trade-offs with the interests of an urban population. The conflict of interest among the stakeholders can be minimised if the NBS offered by the mitigation measures can be factored into the cost-benefit analysis and a multifunctional floodplain management approach can be adopted (Jakubínský, et al., 2021) (Sanon, et al., 2012).
- **Trade-offs with community practices and local land-use:** Implementing mitigation measures might have conflicting interests with cultural and social practices. If not communicated and collaborated with local communities and stakeholders, based upon principles of Free, Prior and Informed Consent, implementation of the mitigation measure will very likely face resistance. Conservation can also limit access to the fresh-water ecosystem and its services for indigenous

peoples and local communities. This conflict of interests can be minimised with communication, education, inclusion and multisectoral collaboration (Dahlberg & Burlando, 2009) (Boughton, et al., 2019).

- **Trade-offs between wetland restoration and biodiversity:** The factors that influence freshwater-based mitigation measures are nutrient cycling and nutrient control, soil organic matter, biomass, decomposition rates, and potential denitrification (section 5.2). But restoring wetlands for carbon and nutrient storage and removal might not be favourable for biodiversity in all cases. In fact, it should not be expected that all ecosystem services would be maximised at a restoration site (Peralta, et al., 2017) (Jessop, et al., 2015). A study conducted on a restored wetland in the USA suggested sites with less biodiversity had greater soil organic matter, biomass, decomposition rates, and denitrification potential (Jessop, et al., 2015).

5.4. Policy status

Many countries in the world have existing policies that address the conservation, restoration or management of wetlands, but less so when considering other aquatic ecosystems. There are also international agreements (e.g., treaties, conventions, protocols) in place to ensure shared understanding of sustainable management of wetlands and to shape actions that can protect the wetlands and the ecosystems surrounding them. The Convention on Wetlands of International Importance especially as Waterfowl Habitat, commonly known as the Ramsar Convention on Wetlands, is the longest established of the intergovernmental environmental agreements and the most relevant to wetlands internationally with 172 parties (nations or states) as signatories as of 2021 (Davidson 2018; Gardner 2018). According to the Ramsar convention's definition of wetlands, all freshwater systems (including rivers, streams, lakes, reservoirs, etc.) discussed in this chapter, are wetlands. This section discusses how freshwater-focused climate mitigation measures have been included in the Ramsar Convention and some countries' national policies.

5.4.1. Ramsar Convention on Wetlands of International Importance

As a multilateral environmental agreement, the Ramsar Convention provides a framework for national action and international cooperation on conservation and wise use of wetlands and their resources (Finlayson 2012). Initially, the Ramsar Convention had its emphasis on the conservation and wise use[12] of wetlands as habitat for waterbirds. The Convention has broadened its scope of implementation over the years, now addressing wise water use for enhanced ecosystem services, sustainable development, and biodiversity conservation, in addition to wetland conservation. (RamsarConvention 2016). While the ecosystem services provided by wetlands have been repeatedly addressed in the convention, the role of wetlands particularly in climate regulation was highlighted much later in the process. Until 2008, the Ramsar Convention's Strategic Plans did not address the importance of wetlands as carbon sinks (RamsarConvention 1996). The Briefing Note 4 provided by Ramsar Convention in 2012 acknowledged carbon sequestration as one of the key benefits of wetland restoration and The Ramsar Strategic Plan 2009-2015 significantly emphasised wetlands' role in climate change mitigation (RamsarConvention 2012; RamsarConvention 2008). Whether or not wetland-based climate mitigation was highlighted, the Ramsar Convention's emphasis on wetland conservation and restoration throughout the years can be considered an indirect but effective measure in supporting the role of wetlands in climate change mitigation.

[12] Wise use of wetlands is the maintenance of their ecological character, achieved through the implementation of ecosystem approaches, within the context of sustainable development (RamsarConvention 2005)

In the latest Strategic Plan (Resolution XII.2: The Ramsar Strategic Plan 2016-2024), the Ramsar Convention mentioned restoration of wetlands for their role in climate change mitigation and adaptation as one of the targets to achieve the strategic goal of wise use of all wetlands (Ramsar Convention Secretariat, 2015). In Briefing Note 10, published in 2018, the wise use and restoration of wetlands is identified as "essential to protect stored carbon and reduce avoidable carbon emissions" (Ramsar Convention Secretariat, 2018). In the latest two Global Wetland Outlook reports (published in 2018 and 2021), the importance of wetland conservation and restoration for climate change mitigation, mostly in peatlands and coastal blue carbon ecosystems (BCE), was significantly highlighted. The Ramsar Convention provides detailed guidelines on the management and restoration of both peatland and BCE to enhance their climate mitigation potential (Ramsar Convention on Wetlands, 2018) (Convention on Wetlands, 2021).

Wetland conservation and restoration are essential to utilise their potential in climate change mitigation. For example, drained peatlands stop sequestering carbon and lose previously stored carbon through decomposition processes for a long period of time - resulting in GHG emissions. Rewetting or restoring wetlands, particularly peatlands, can significantly reduce CO2 emissions (also other GHGs) and may reinitiate carbon sequestration, but rewetted peatlands might not return to the undisturbed natural conditions that allow high climate mitigation potential even within decades. Hence, conservation of these wetlands is the best measure to maximise the mitigation benefits and restoration is still better than having drained wetlands (Kreyling , et al., 2021) (Günther, et al., 2020) (Joosten, 2015). For years, the Ramsar Convention's effort in global wetland conservation and restoration played a big role in protecting the carbon pools in wetlands. Although Ramsar identifies rivers, streams, lakes, reservoirs as wetlands, they have not included noticeable guidelines to minimise emissions from these systems and more effort is required.

5.4.2 National policies

A national-level policy related to wetlands has the capacity to outline goals related to wetland management, timelines for achievement of those goals, roles and responsibilities of various actors, and budget commitments (Gardner, 2018b). The Ramsar Convention recommends that parties develop National Wetland Policies to implement the Convention at national and regional levels (Ramsar Convention Secretariat, 2015) (Ramsar Convention Secretariat, 2010) (Bonells, 2018). While many countries have specific wetland-specific national policies, others include wetland related policies under other broader environmental policies or land-use and water-use policies. Peimer et al. (2017) examined wetlands policies in 193 countries and found that only 9 % have an existing wetland-specific policy, 38 % have a broad environmental policy or law that includes wetlands, 18% have wetland policy in development and 23% have no national-level environmental policy or strategy to protect wetlands.

Wetland-specific national policies can be very important in protecting and managing wetlands and ensuring they maintain their role in climate change mitigation (Peimer, et al., 2017). For example, adoption of a National Wetlands Policy in Uganda in the early 1990s paved the way to inclusion of wetlands in many of their national policies and eventually reflected in Uganda's updated NDC for the period of 2021-2030, where wetlands have been included under one the key sectors of Agriculture, Forestry and Other Land-Use (AFOLU) for mitigation (Mafabi, 2018) (Ministry of Water and Environment, Uganda, 2021) (see Chapter 3), and is one of the few examples on this within the first round of NDCs. Chile also developed a National Wetlands Strategy (NWS) in 2005 which enables coordinated and efficient protection of wetlands and aligns with the country's National Biodiversity Strategy. To achieve one of the objectives of the strategy, the country has created a National Wetlands Inventory (Suárez-Delucchi, 2018). As per Chile's latest NDC updates, the country considers wetlands in their mitigation strategy now (See Box 5.2).

In the United States, wetlands are included in several different broader environmental policies, management plans, acts, regulations and even executive orders. The US adopted the "No net loss" policy (a policy also adopted by the European Union) for wetland preservation in 1989 with the goal to balance wetland loss with replacement wetlands, mainly through reclamation, mitigation, and restoration to maintain the total areal coverage of wetlands in the country (Everard, 2018b). The policy showed promising results in the initial years, but 62,300 acres of wetland was reported lost from 2004 to 2009 (Smaczniak, 2018). One of the key measures to maintain "No net loss" is wetland offsets also called compensatory wetlands that entails creation or restoration of wetlands of at least the same area as the lost wetland (Fennessy & Dresser, 2018). As these compensatory/replacement wetlands may be significantly different from the natural wetlands in character and function, their role in climate change mitigation also may vary greatly (Neubauer & Verhoeven, 2019) (BenDor & Riggsbee , 2011) (Fennessy & Dresser, 2018) (Everard, 2018b). Neubauer & Verhoeven (2019) maintain that GHG emissions from disturbed wetlands persist long after a wetland is restored or replaced by a mitigation wetland until they become net sinks. Hence, from a climate change mitigation perspective, stronger priority should be given to protecting existing natural wetlands (Neubauer & Verhoeven, 2019).

It is important that wetland-specific national policies emphasise wetland conservation, restoration, and wise use. But if nations are considering wetlands for climate change mitigation, it needs to be eventually reflected in their NDCs as well as national and local policies with quantitative emission targets. Wetland related measures should be considered an integral part of NDCs (Anisha, et al., 2020). Box 5.2 illustrates some examples of wetland-centric mitigation measures in NDCs. Inclusion of freshwater-related policies in national-level documents, such as NAPs, NBSAPs, and Integrated Coastal Zone Management can lay the groundwork for NDCs and vice versa in the future.

Box 5.2: Integration in NDC

Freshwater and tidal wetlands were included in most enhanced NDC's prepared in the last two years prior to January 2022. Within Annex 1 countries, references to wetlands are mainly noted through recognition within LULUCF category targets, although parties such as Canada and Iceland included actions to restore wetlands as part of their measures. Freshwater ecosystem measures, including protection, rehabilitation, and enhancement activities, are more commonly found within updated NDC's from Non-Annex 1 parties, including both adaptation and mitigation. In the first round of NDC's, only seven non-Annex 1 parties included measures relating to wetlands, most notably Uganda, and most of these related to adaptation, although Uganda did include measures within its mitigation section. Similarly, in the first round, only a few countries, most notably the Bahamas, noted the role of mangrove swamps as a carbon sink and their ecological functions.

In comparison, a total of 65 Non-Annex 1 countries (57%), out of 114 Non-Annex 1 NDC released between 2019-2022 have included wetland measures in their enhanced NDC's, with a further four including wetlands within their inventories. Most of these wetland measures are still mainly adaptation priorities, but recognition of the role of wetlands in mitigation or in integrated mitigation/adaptation also increased. significantly. Approximately 18 Non-Annex 1 parties included specific wetlands mitigation measures (16% of total), and 25 countries specifically included mangrove forests in their mitigation priorities, noted mainly as 'Blue Carbon' priorities. Of note are measures by the Democratic Republic of the Congo with respect to the important role of peatlands nationally and globally, especially regarding emissions reductions. Measures for wetlands found in mitigation sections were much less detailed when compared with measures found in adaptation sections.

Acknowledgement of the role of mangrove ecosystems in both mitigation and adaptation was much higher in enhanced NDC's, most notably Belize and Colombia. Forty-nine countries included mangrove measures within their respective enhanced NDC's including close to 62% of those countries hosting mangrove ecosystems, but as above, a smaller number including mangrove measures within their mitigation sections.

The potential role of other water-related ecosystems such as rivers or lakes in mitigation was not directly found in any enhanced NDCs, despite recent research suggesting that overly degraded systems may be a strong source of emissions. However, water pollution through inadequate wastewater management, and its impact on freshwater ecosystems and their capacity to provide ecosystem services, was noted in many adaptation sections, and was much more prominent compared with the first round of NDCs.

Examples of mitigation measures include:

Belize: *Enhance the capacity of the country's mangrove and seagrass ecosystems to act as a carbon sink by 2030, through increased protection of mangroves and by removing a cumulative total of 381 KtCO2e between 2021 and 2030 through mangrove restoration.*

Sierra Leone: *Organic manure to reduce fertiliser use that has the tendency of depleting the soil fertility and polluting wetlands.*

South Sudan: *Conservation and sustainable use of wetlands for improved carbon sequestration. South Sudan will collaborate with international research institutes and agencies to conduct research on the release of methane emissions from the Sudd wetland and develop measures to sustainably manage and mitigate high emissions coming from the country's wetlands.*

Uganda: *The measure aims to increase wetland coverage from 8.9% in 2020, to 9.57% in 2025, and approximately 12% by 2030 through the implementation of wetland management practices such as demarcation, gazettement, and restoration of degraded wetlands. The mitigation reduction potential for this measure is expected to account for 0.4 MtCO2e by 2030.*

Background to the NDCs are found in Chapter 3, section 3.2.1 and box 3.2.

(SIWI/GIZ NDC study (forthcoming))

5.5. Potential implications for governance

Inclusion in national policies. Section 5.3.2 illustrates the importance of having wetland focused national policies to bring traction in wetland-focused climate change mitigation measures. Uganda and Chile (cases mentioned previously in this chapter) demonstrate a clear example of that. It is important that wetland-specific national policies emphasise wetland conservation, restoration, and wise use. But whether nations are considering wetlands or other freshwater systems for climate change mitigation, reflects in their NDCs. Freshwater related mitigation measures should be considered as an integral part of NDCs (Anisha, et al., 2020). However, inclusion of freshwater related policies in national-level documents, such as NAPs, NBSAPs, and Integrated Coastal Zone Management can lay the groundwork for NDCs.

Systems-level approach. Many of the mitigation measures outlined in section 5.2 are applicable to most freshwater systems. For example, nutrient control benefits rivers, lakes, reservoirs and other wetlands alike for climate change mitigation, as GHG production in aquatic systems is mainly fueled by inputs from the watershed. Effective emission reduction strategies should entail coordinated approaches for land management, restricting nutrient loading, maintaining and improving ecohydrological connections. Inland water bodies constantly interact with other components of the ecosystem (vegetation, landform, biodiversity, humans) and among themselves through subsurface flow, groundwater flow, ecohydrological connectivity, sediment and organic matter exchange. Hence, mitigation benefits measures cannot be sustainably materialised if the activities are undertaken in isolation. System level approaches, be it on a local level, sub-regional level or regional level, can minimise the potential trade-offs among different interests. This requires inter-sectoral coordination and achieving policy synergies. Management and planning ought to consider the different scales at which socio-ecological systems might interact with freshwater systems and make sure the dynamics are sustainable.

Implications of future climate change. Climate change is predicted to affect freshwater systems. But floodplain ecosystems and well-managed wetlands, even if in a low-diversity state, are likely to persist under climate change and provide adaptation services (Lavorel, et al., 2015). In fact, many areas with large water bodies have persisted through the climatic changes of the Holocene, proving their resilience (Morelli, T. L., et al., 2016). It is uncertain though, if the freshwater systems would persist with the same characteristics that enable them to sequester carbon over long periods of time (Sutfin, et al., 2016) (Yan, et al., 2021). For example, more rainfall due to climate change will increase flushing and delivery of soil and riparian/wetland carbon to streams and rivers which will result in more GHG emissions. Peatlands will release more carbon if drought conditions prevail. Tidal wetlands will be affected by sea level rise. Hence, the planning should not be based on historic or present trends but should take future climate change scenarios into consideration. Developing an understanding of how ecosystems might transform under climate change can assist in adopting measures that can adapt as conditions change.

Implication of socio-economic change. As discussed in section 5.2, anthropogenic activities have disturbed the carbon pool in freshwater systems and increased GHG emission from the systems, and probably will continue to do so. For example, societal choices would determine the future total nitrogen (TN) loading in the freshwater system. Future global population and their socio-economic choices would determine global demand for food and agriculture, bioenergy, assumptions about trade, and assumptions about agricultural management practices which would eventually determine the TN loading in the freshwater systems, although the practices might vary regionally (Sinha, et al., 2019). The planning and management of freshwater-based mitigation measures should consider these socio-economic changes for successful implementation.

5.6. Conclusions and future outlook

This chapter takes a “Problem-Cause-Solution” approach to address freshwater-ecosystem-based climate mitigation: It discusses under what circumstances the long-term carbon sinks, the freshwater systems, become carbon sources and how to undo or minimise that shift to continue benefiting from their potential to sequester carbon. The chapter also outlines the incentives and planning considerations. These mitigation measures come with substantial co-benefits and align with Sustainable Development Goals (SDG), but their adoption might need to be tailored according to the local and regional context.

Historically, climate change mitigation potential of freshwater systems has been highly underrated. Although freshwater marshes, swamps and peatlands have been more regularly included in the discussion recently (but yet not sufficiently), the role of rivers, lakes and reservoir management and its impact on whether these are sources or sinks are still not reflected in important national policies (e.g. NDCs). Freshwater systems have been rather considered carbon neutral or as sinks. Most freshwater systems were not significant sources of GHGs before being exposed to anthropogenic disturbances. Freshwater systems in most parts of the world have been subjected to some kind of disturbance which imposes a risk of those systems becoming net sources of GHG emissions. Every signatory party under the Paris Agreement has some potential to include freshwater-based mitigation targets in their NDCs and it is essential that inclusion of freshwater systems is mainstreamed.

5.7. References

Acreman, et al. (2021). Evidence for the effectiveness of nature-based solutions to water issues in Africa. Environmental Research Letters, 16(6), 063007.

Allen , G. H., & Pavelsky, T. M. (2018). Global extent of rivers and streams. Science, 361(6402), 585-588.

Altor, A. E., & Mitsch, W. J. (224-234). Methane flux from created riparian marshes: Relationship to intermittent versus continuous inundation and emergent macrophytes. 2006: Ecological Engineering, 28(3).

Amon, J. P., Jacobson, C. S., & Shelley, M. L. (2005). Construction of fens with and without hydric soils. Ecological Engineering, 24(4), 341-357.

Andersen, R., Chapman, S., & Artz, R. (2013). Microbial communities in natural and disturbed peatlands: a review. Soil Biology and Biochemistry, 57, 979-994.

Angradi, T. R., Ringold, P. L., & Hall, K. (2018). Water clarity measures as indicators of recreational benefits provided by US lakes: Swimming and aesthetics. Ecological indicators, 93, 1005-1019.

Anisha, N. F., Mauroner, A., Lovett, G., Neher, A., et al. (2020). Locking Carbon in Wetlands: Enhancing Climate Action by Including Wetlands in NDCs. Corvallis, Oregon and Wageningen, The Netherlands: Alliance for Global Adaptation and Wetlands International.

Anshari, G. Z., Afifudin, M., Nuriman, M., Gusmayanti, E., Arianie, L., Susana, R., . . . Rafiastanto, A. (2010). Drainage and land use impacts on changes in selected peat properties and peat degradation in West Kalimantan Province, Indonesia. Biogeosciences, 7(11), 3403-3419.

Audet, J., Bastviken, D., Bundschuh, M., Buffam, I., Feckler, A., Klemedtsson, et al. (2020). Forest streams are important sources for nitrous oxide emissions. Global change biology, 26(2), 629-641.

Ausseil, A. G., Jamali, H., Clarkson, B. R., & Golubiewski, N. E. (2015). Soil carbon stocks in wetlands of New Zealand and impact of land conversion since European settlement. Wetlands ecology and management, 23(5), 947-961.

Badiou, P. (2017). The Importance of Freshwater Mineral Soil Wetlands in the Global Carbon Cycle. Institute for Wetland and Waterfowl Research, Duck Unlimited Canada.

Barbier, E. B., Hacker, S. D., Kennedy, C., Koch, E. W., Stier, A. C., & Silliman, B. R. (2011). The value of estuarine and coastal ecosystem services. Ecological monographs, 81(2), 169-193.

Bastviken, D., Tranvik, L. J., Downing, J. A., Crill, P. M., & Enrich-Prast, A. (2011). Freshwater methane emissions offset the continental carbon sink. Science, 331(6013), 50-50.

Batson, J., Noe, G. B., Hupp, C. R., Krauss, K. W., et al. (2015). Soil greenhouse gas emissions and carbon budgeting in a short-hydroperiod floodplain wetland. Journal of Geophysical Research: Biogeosciences, 120(1), 77-95.

Batson, J., Noe, G. B., Hupp, C. R., Krauss, K. W., Rybicki, N. B., & Schenk, E. R. (2015a). Soil greenhouse gas emissions and carbon budgeting in a short-hydroperiod floodplain wetland. Journal of Geophysical Research: Biogeosciences, 120(1), 77-95.

Beaulieu, J. J., DelSontro, T., & Downing, J. A. (2019). (2019). Eutrophication will increase methane emissions from lakes and impoundments during the 21st century. Nature communications, 10(1), 1-5.

Beaulieu, J. J., Waldo, S., Balz, D. A., Barnett, W., Hall, A., Platz, M. C., & White, K. M. (2020). Methane and carbon dioxide emissions from reservoirs: controls and upscaling. Journal of Geophysical Research: Biogeosciences, 125(12).

Beck, M. W., Sherwood, E. T., Henkel, J. R., Dorans, K., Ireland, K., & Varela, P. (2019). Assessment of the cumulative effects of restoration activities on water quality in Tampa Bay, Florida. Estuaries and Coasts, 42(7), 1774-1791.

Belle, J. A., Jordaan, A., & Collins, N. (2018). Managing wetlands for disaster risk reduction: A case study of the eastern Free State, South Africa. Jàmbá: Journal of Disaster Risk Studies, 10(1), 1-10.

BenDor, T. K., & Riggsbee , J. A. (2011). A survey of entrepreneurial risk in U.S. wetland and stream compensatory mitigation markets. Environmental Science Policy 14, 301–314.

Berberich, M. E., Beaulieu, J. J., Hamilton, T. L., Waldo, S., & Buffam, I. (2020). Spatial variability of sediment methane production and methanogen communities within a eutrophic reservoir: Importance of organic matter source and quantity. Limnology and oceanography, 65(6), 1336-1358.

Bernal, B., & Mitsch, W. J. (2008). A comparison of soil carbon pools and profiles in wetlands in Costa Rica and Ohio. Ecological Engineering 34, 311–323.

Bernal, B., & Mitsch, W. J. (2012). Comparing carbon sequestration in temperate freshwater wetland communities. Global Change Biology, 18(5), 1636-1647.

Bonells, M. (2018). National Wetland Policies: Overview. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Borges, A. V., Abril, G., Darchambeau, F., Teodoru, C. R., Deborde, J.,et al. (2015). Divergent biophysical controls of aquatic CO 2 and CH 4 in the World's two largest rivers. Scientific Reports, 5(1), 1-10.

Borges, A. V., Darchambeau, F., Lambert, T., Bouillon, S., Morana, et al. (2018). Effects of agricultural land use on fluvial carbon dioxide, methane and nitrous oxide concentrations in a large European river, the Meuse (Belgium). Science of the Total Environment, 610, 342-355.

Borges, A. V., Darchambeau, F., Teodoru, C. R., Marwick, T. R., et al. (2015). Globally significant greenhouse-gas emissions from African inland waters. Nature Geoscience, 8(8), 637-642.

Boughton, et al. (2019). Trade-offs and synergies in a payment-for-ecosystem services program on ranchlands in the Everglades headwaters. Ecosphere, 10(5), e02728.

Bridgham, S. D., Megonigal, J. P., Keller, J. K., Bliss, N. B., & Trettin, C. (2006). The carbon balance of North American wetlands. Wetlands 26, 889-916.

Brown, C., Shaker, R. R., & Das, R. (2018). A review of approaches for monitoring and evaluation of urban climate resilience initiatives. Environment, development and sustainability, 20(1), 23-40.

Bruland, G., & Richardson, C. (2005). Spatial variability of soil properties in created, restored, and paired natural wetlands. Soil Science Society of America Journal, 69, 273–284.

Buckton, S. (2018). Products from Wetlands: Overview. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Burkhard, B., & Maes, J. (2017). Mapping Ecosystem Services. Sofia: Pensoft Publishers.

Burlacot, A., Richaud, P., Gosset, A., Li-Beisson, Y., & Peltier, G. (2020). Algal photosynthesis converts nitric oxide into nitrous oxide. Proceedings of the National Academy of Sciences, 117(5), (pp. 2704-2709).

Butman, D., & Raymond, P. A. (2011). Significant efflux of carbon dioxide from streams and rivers in the United States. Nature Geoscience, 4(12), 839-842.

Byrd, K., Ratliff, J., Bliss, N., et al. (2015). Quantifying climate change mitigation potential in the United States Great Plains wetlands for three greenhouse gas emission scenarios. Mitigation and Adaptation Strategies for Global Change, 20(3), 439-465.

Cabral, I., Costa, S., Weiland, U., & Bonn, A. (. (2017). Nature-Based Solutions to Climate Change Adaptation in Urban Areas:Linkages between Science, Policy and Practice. Springer.

Cantonati, M; Poikane, S; Pringle, C M, et al. (2020). Characteristics, main impacts, and stewardship of natural and artificial freshwater environments: consequences for biodiversity conservation. Water, 12(1), 260.

Carter, V. (1996). Technical aspects of wetlands: wetland hydrology, water quality, and associated functions. USGS Water Supply Paper, 2425, 1-25.

Cassin, J. (2021). Nature-based solutions for source water protection in North America. In In Nature-based Solutions and Water Security (pp. 179-214.). Elsevier.

Cassin, J., & Matthews, J. H. (2021). Nature-based solutions, water security and climate change: Issues and opportunities. In In Nature-based Solutions and Water Security (pp. 63-79). Elsevier.

Chang, C. R., Li, M. H., & Chang, S. D. (2007). 2007. A preliminary study on the local cool-island intensity of Taipei city parks. Landscape and Urban Planning 80, 386–395.

Chmura, G. L., Anisfeld, S. C., Cahoon, D. R., & Lynch, J. C. (2003). Global carbon sequestration in tidal, saline wetland soils. Global biogeochemical cycles, 17(4).

Cierjacks, et al. (2010). Carbon stocks of soil and vegetation on Danubian floodplains. Journal of Plant Nutrition and Soil Science, 173(5), 644-653.

Cohen-Shacham, E., Dayan, T., Feitelson, E., & De Groot, R. S. (2011). Ecosystem service trade-offs in wetland management: drainage and rehabilitation of the Hula, Israel. Hydrological Sciences Journal, 56(8), 1582-1601.

Cohen-Shacham, E., Walters, G., Janzen, C., & Maginnis, S, (eds.). (2016). Nature-based solutions to address global societal challenges. Gland, Switzerland: IUCN.

Cole, J.J., Prairie, Y.T., Caraco, N.F. et al. (2007). Plumbing the Global Carbon Cycle: Integrating Inland Waters into the Terrestrial Carbon Budget. Ecosystems 10, 172–185.

Cole, L. E., Bhagwat, S. A., & Willis, K. J. (2019). Fire in the swamp forest: palaeoecological insights into natural and human-induced burning in intact tropical peatlands. Frontiers in Forests and Global Change, 2, 48.

Colloff, M. J., Lavorel, S., Wise, R. M., Dunlop, M., Overton, I. C., & Williams, K. J. (2016). Adaptation services of floodplains and wetlands under transformational climate change. Ecological Applications, 26(4), 1003-1017.

Colls, A., Ash, N., & Ikkala, N. (2009). (2009). Ecosystem-based Adaptation: a natural response to climate change. Gland, Switzerland: IUCN.

Conchedda, G., & Tubiello, F. N. (2020). Drainage of organic soils and GHG emissions: validation with country data. Earth System Science Data, 12(4), 3113-3137.

Convention on Wetlands. (2021). Global Wetland Outlook: Special Edition 2021. Gland, Switzerland: Secretariat of the Convention on Wetlands.

Convention on Wetlands. (2021). The contributions of blue carbon ecosystems to climate change mitigation. Briefing Note No. 12. Gland, Switzerland: Secretariat of the Convention on Wetlands.

Cook, B. I., Mankin, J. S., & Anchukaitis, K. J. (2018). Climate change and drought: From past to future. Current Climate Change Reports, 4(2), 164-179.

Costanza, R., et al. (1997). The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Crane, M., & Xian, M. (2006). Urban growth and seagrass distribution in Tampa Bay, Florida. . In: Avecedo, W., Taylor, J.L., Hester, D.J., Mladinich, C.S., Glavac, S. (Eds.), Rates, Trends, Causes, and Consequences of Urban Land-use Change in the United States.

Crawford, J. T., Loken, L. C., West, W., Crary, B., Spawn, S. A., Gubbins, N., . . . Stanley, E. H. (2017). Spatial heterogeneity of within-stream methane concentrations. J. Geophys. Res. Biogeosci., 122, 1036–1048.

Cullen-Unsworth, L. C., Nordlund, L. M., Paddock , J., Baker, S., McKenzie, L. J., & Unsworth, R. K. (2014). Seagrass meadows globally as a coupled social–ecological system: Implications for human wellbeing. Marine Pollution Bulletin, 83(2), 387-397.

Dahlberg, A. C., & Burlando, C. (2009). Addressing trade-offs: Experiences from conservation and development initiatives in the Mkuze Wetlands, South Africa. Ecology and Society, 14(2).

Darby, F. A., & Turner, R. E. (2008). (2008). Effects of eutrophication on salt marsh root and rhizome biomass accumulation. Marine Ecology Progress Series, 363, 63-70.

Dargie, G. C., Lewis, S. L., Lawson, I. T., Mitchard, E. T., Page, S. E., Bocko, Y. E., & Ifo, S. A. (2017). Age, extent and carbon storage of the central Congo Basin peatland complex. Nature, 542(7639), 86-90.

Davidson, N. C. (2018). Ramsar convention on wetlands: scope and implementation. In F. C. (eds), In The Wetland Book I: Structure and function, management, and methods (pp. 451-458). Springer.

de Groot, R., Finlayson, M., Verschuuren, B., Ypma , O., & Zylstra, M. (2008). Integrated assessment of wetland services and values as a tool to analyse policy trade-offs and management options: A case study in the Daly and Mary River catchments, Northern Australia.Supervising Scientist Report 198 . Darwin NT: Supervising Scientist.

Deegan, L. A., Johnson, D. S., Warren, R. S., Peterson, B. J., Fleeger, J. W., Fagherazzi, S., & Wollheim, W. M. (2012). Coastal eutrophication as a driver of salt marsh loss. Nature, 490(7420), 388-392.

Deemer, et al. (2016). Greenhouse gas emissions from reservoir water surfaces: a new global synthesis. BioScience, 66(11), 949-964.

DelSontro , T., Beaulieu,, J. J., & Downing, J. A. (2018). Greenhouse gas emissions from lakes and impoundments: Upscaling in the face of global change. Limnology and Oceanography Letters, 3(3), 64-75.

Dosskey, M. G. (2001). Toward quantifying water pollution abatement in response to installing buffers on crop land. Environmental Management, 28(5), 577-598.

Dosskey, M. G. (2001). Toward quantifying water pollution abatement in response to installing buffers on crop land. Environmental Management, 28(5), 577-598.

Doswald, N., & Osti, M. (2011). Ecosystem-based approaches to adaptation and mitigation – good practice examples and lessons learned in Europe. Bonn, Germany: Bundesamt für Naturschutz (BfN).

Duarte, C. M., & Krause-Jensen, D. (2017). Export from seagrass meadows contributes to marine carbon sequestration. Frontiers in Marine Science, 4, 13.

Duarte, C. M., Losada, I. J., Hendriks, I. E., Mazarrasa, I., & Marbà, N. (2013). The role of coastal plant communities for climate change mitigation and adaptation. Nature Climate Change 3, 961–968.

Duarte, C. M., Middelburg, J. J., & Caraco , N. (2005). Major role of marine vegetation on the oceanic carbon cycle. Biogeosciences, 2(1), 1-8.

Dybala, K. E., Steger, K., Walsh, R. G., Smart, D. R., Gardali, T., & Seavy, N. E. (2019). Optimizing carbon storage and biodiversity co-benefits in reforested riparian zones. Journal of Applied Ecology, 56(2), 343-353.

Endter-Wada, J., Kettenring, K. M., & Sutton-Grier, A. (2020). Protecting wetlands for people: Strategic policy action can help wetlands mitigate risks and enhance resilience. Environmental Science & Policy, 108, 37-44.

Euliss Jr, N. H., Gleason, R. A., Olness, A., McDougal, R. L. et al. (2006). North American prairie wetlands are important nonforested land-based carbon storage sites. Science of the Total Environment, 361 (1-3), 179-188.

Evans, C. D., Peacock, M., Baird, A. J., Artz, R., Burden, A., Callaghan, N., . . . Cumming, A. (2021). Overriding water table control on managed peatland greenhouse gas emissions. Nature, 593(7860), 548-552.

Everard, M. (2018a). National Wetland Policy: Ghana. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Everard, M. (2018b). No Net Loss: Overview. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Everard, M. (2018c). Nutrient Cycling in Wetlands: Supporting Services. In F. C. (eds), The Wetland Book. Dordrecht: Springer.

Fagorite, V. I., Odundun, O. A., Iwueke, L. E., Nwaigbo, U. N., & Okeke , O. C. (2019). Wetlands; A review of their classification, significance and management for sustainable development. Wetlands, 5(3).

Fearnside, P. M. (2006). Greenhouse gas emissions from hydroelectric dams: Reply to Rosa et al. Climatic Change, 75(1-2), 103-109.

Fearnside, P. M. (2007, Janurary 9). Why hydropower is not clean energy. França: Scitizen.

Feller, I. C., Friess, D. A., Krauss, K. W., & Lewis, R. R. (2017). The state of the world's mangroves in the 21st century under climate change. Hydrobiologia, 803(1), 1-12.

Feng, J., Liang, J., Li, Q., Zhang, X., Yue, Y., & Gao, J. (2021). Effect of hydrological connectivity on soil carbon storage in the Yellow River delta wetlands of China. Chinese Geographical Science, 31(2), 197-208.

Fennessy, M. S., & Dresser , A. R. (2018). No Net Loss Case Study: Structural and Functional Equivalence of Mitigation Wetlands. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Finlayson, C. M. (2012). Forty years of wetland conservation and wise use. Aquatic Conservation: Marine and Freshwater Ecosystems 22, 139–143.

Flores-Rios, A., Thomas, E., Peri, P. P., Amelung, W., Duarte-Guardia, S., Borchard, N., . . . Steffan-Dewenter, I. (2020). Co-benefits of soil carbon protection for invertebrate conservation. Biological Conservation, 252, 108859.

Fluet-Chouinard, E., Messager, M. L., Lehner, B., & Finlayson, C. M. (2018). Freshwater Lakes and reservoirs. In In The wetland book II: Distribution, description, and conservation (pp. 125-142). Springer.

Folke, C. (2006). Resilience: the emergence of a perspective for social–ecological systems analyses. Global Environmental Change 16, 253–267.

Ford, H., Garbutt, A., Ladd, C., Malarkey, J., & Skov. , M. W. (2016). Soil stabilization linked to plant diversity and environmental context in coastal wetlands. Journal of vegetation science, 27(2), 259-268.

Fourqurean, J. W., Duarte, C. M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M. A., . . . Serrano, O. (2012). Seagrass ecosystems as a globally significant carbon stock. Nature geoscience, 5(7), 505-509.

Friedlingstein, P., O'sullivan, M., Jones, M. W., Andrew, R. M., Hauck, J., Olsen, A., . . . Le Quéré, C. (2020). Global carbon budget 2020. Earth System Science Data, 12(4), 3269-3340.

Frolking, S., Roulet, N., & Fuglestvedt, J. (2006). How northern peatlands influence the Earth's radiative budget: Sustained methane emission versus sustained carbon sequestration. Journal of Geophysical Research: Biogeosciences, 111(G1).

Günther, A., Böther, S., Couwenberg, J., Hüttel, S., & Jurasinski, G. (2018). Profitability of direct greenhouse gas measurements in carbon credit schemes of peatland rewetting. Ecological Economics, 146, 766-771.

Günther, A., Barthelmes, A., Huth, V., Joosten, H., Jurasinski, G., Koebsch, F., & Couwenberg, J. (2020). Prompt rewetting of drained peatlands reduces climate warming despite methane emissions. Nature communications, 11(1), 1-5.

Gaffney, P. P., Hugron, S., Jutras, S., Marcoux, O., Raymond, S., & Rochefort, L. (2020). Ecohydrological change following rewetting of a deep-drained northern raised bog. Ecohydrology, 13(5), e2210.

Galatowitsch, S. M. (2018). Natural and anthropogenic drivers of wetland change. In In The wetland book II: Distribution, description, and conservation (pp. 359-367). Springer Nature.

Gao , X., Ouyangi, W., Lin, C., Wang, K., Hao, F., Hao, X., & Lian, Z. (2020). Considering atmospheric N2O dynamic in SWAT model avoids the overestimation of N2O emissions in river networks. Water Research 174.

Gardner, R. C. (2018a). Framework of International Conventions. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Gardner, R. C. (2018b). National Wetland Policies: The Basics. In F. C. (eds), The Wetland Book. Dordrecht: Springer.

Gedney, N., Huntingford, C., Comyn-Platt, E., & Wiltshire, A. (2019). Significant feedbacks of wetland methane release on climate change and the causes of their uncertainty. Environmental Research Letters, 14(8), 084027.

GIZ. (2020). Stop Floating, Start Swimming: Water and climate change – Interlinkages and prospects for future action. Bonn: Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ) GmbH.

Gleason, R. A., Tangen, B. A., Browne, B. A., & Euliss, N. H. (2009). Greenhouse gas flux from cropland and restored wetlands in the Prairie Pothole Region. Soil Biology and Biochemistry, 41, 2501–2507.

Global Water Partnership. (2018). Preparing to Adapt: The Untold Story of Water in Climate Change Adaptation Processes. Stockholm, Sweden: Global Water Partnership.

Gober, P., Brazel, A., Quay, R., Myint, S., Clarke, S. G., Miller, A., & Rossi, S. (2010). Using watered landsapes to manipulate urban heat island effects. Journal of the American Planning Association, vol. 76, 109-121.

Gonzalez-Valencia, R., Sepulveda-Jauregui, A., Martinez-Cruz, K., Hoyos-Santillan, J., Dendooven, L., & Thalasso, F. (2014). Methane emissions from Mexican freshwater bodies: correlations with water pollution. Hydrobiologia, 721(1), 9-22.

Gorham, E. (1991). Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecological applications, 1(2), 182-195.

Greening, H., Janicki, A., & Sherwood, E. T. (2018). (2018). Seagrass recovery in tampa Bay, Florida (USA)., . In The Wetland Book: II: Distribution, Description, and Conservation (pp. 495-506). Springer.

Greening, H., Janicki, A., Sherwood, E. T., Pribble, R., & Johansson, J. O. (2014). Ecosystem responses to long-term nutrient management in an urban estuary: Tampa Bay, Florida, USA. Estuarine, Coastal and Shelf Science, 151, A1-A16.

Greening, H; Janicki, A; Sherwood, E T;. (2018). Seagrass recovery in tampa Bay, Florida (USA). In C. M. Finlayson, G. R. Milton, R. C. Prentice, & N. C. Davidson, The Wetland Book: II: Distribution, Description, and Conservation (pp. 495-506). Dordrecht: Springer.

Greiner, J. T., McGlathery, K. J., Gunnell, J., & McKee, B. A. (2013). Seagrass restoration enhances “blue carbon” sequestration in coastal waters. PloS one, 8(8), e72469.

Grizzetti, B., Liquete, C., Pistocchi, A., Vigiak, O., Zulian, G., Bouraoui, F., . . . Cardoso, A. C. (2019). Relationship between ecological condition and ecosystem services in European rivers, lakes and coastal waters. Science of the Total Environment, 671, 452-465.

Grizzetti, B., Passy, P., Billen, G., Bouraoui, F., Garnier, J., & Lassaletta, L. (2015). The role of water nitrogen retention in integrated nutrient management: assessment in a large basin using different modelling approaches. Environmental Research Letters, 10(6), 065008.

Gumbricht, T., Roman-Cuesta, R. M., Verchot, L., Herold, M., Wittmann, F., Householder, E., . . . Murdiyarso, D. (2017). An expert system model for mapping tropical wetlands and peatlands reveals South America as the largest contributor. Global change biology, 23(9), 3581-3599.

Gunkel, G. (2009). Hydropower–A green energy? Tropical reservoirs and greenhouse gas emissions. CLEAN–Soil, Air, Water, 37(9), 726-734.

Gupta, G., Khan, J., Upadhyay, A. K., & Singh, N. K. (2020). Wetland as a Sustainable Reservoir of Ecosystem Services: Prospects of Threat and Conservation. In In Restoration of wetland ecosystem: A trajectory towards a sustainable environment (pp. 31-43). Singapore: Springer.

Hamdan, L. J., & Wickland, K. P. (2016). Methane emissions from oceans, coasts, and freshwater habitats: New perspectives and feedbacks on climate. Limnology and Oceanography, 61(S1), S3-S12.

Han, G., Xing, Q., Yu, J., Luo, Y., Li, D., Yang, L., et al. (2014). Agricultural reclamation effects on ecosystem CO2 exchange of a coastal wetland in the Yellow River Delta. Agriculture, Ecosystems & Environment, 196, 187-198.

Hapsari, K. A., Biagioni, S., Jennerjahn, T. C., Reimer, P., Saad, A., Sabiham, S., & Behling, H. (2018). Resilience of a peatland in Central Sumatra, Indonesia to past anthropogenic disturbance: Improving conservation and restoration designs using palaeoecology. Journal of Ecology, 106(6), 2473-2490.

Harrison, J. A., Deemer, B. R., Birchfield, M. K., & O'Malley, M. T. (2017). Reservoir water-level drawdowns accelerate and amplify methane emission. Environmental science & technology, 51(3), 1267-1277.

Harrison, J. A., Prairie, Y. T., Mercier-Blais, S., & Soued, C. (2021). Year-2020 Global Distribution and Pathways of Reservoir Methane and Carbon Dioxide Emissions According to the Greenhouse Gas From Reservoirs (G-res) Model. Global Biogeochemical Cycles, 35(6), e2020GB006888.

Harvey, J., & Gooseff, M. (2015). River corridor science: hydrologic exchange and ecological consequences from bedforms to basins. Water Resources Research 51, 6893–6922.

Hathway, E. A., & Sharples, S. (2012). The interaction of rivers and urban form in mitigating the Urban Heat Island effect: A UK case study. Building and Environment, vol. 58, 14-22.

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B. D., Aurela, M., Barr, A. G., . . . Chi, J. (2020b). Increasing contribution of peatlands to boreal evapotranspiration in a warming climate. Nature Climate Change, 10(6), 555-560.

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B., Aurela, M., Barr, A. G., . . . Desai, A. R. (2020a). The biophysical climate mitigation potential of boreal peatlands during the growing season. Environmental Research Letters, 15(10), 104004.

Herbst, M., Friborg, T., Schelde, K., Jensen, R., Ringgaard, R., et al. (2013). Climate and site management as driving factors for the atmospheric greenhouse gas exchange of a restored wetland. Biogeosciences, 10(1), 39-52.

Holgerson, M. A., & Raymond, P. A. (2016). Large contribution to inland water CO 2 and CH 4 emissions from very small ponds. Nature Geoscience, 9(3), 222-226.

Hondula, K. L., Jones, C. N., & Palmer, M. A. (2021). Effects of seasonal inundation on methane fluxes from forested freshwater wetlands. Environmental Research Letters, 16(8), 084016.

Hooijer, A., Vernimmen, R., Mawdsley , N., Page, S., Mulyadi, D., & Visser, M. (2015). Assessment of impacts of plantation drainage on the Kampar Peninsula peatland, Riau. Deltares report, 1207384.

Howard, J., Sutton-Grier, A., Herr, D., Kleypas, J., Landis, E., Mcleod, E., . . . Simpson, S. (2017). Clarifying the role of coastal and marine systems in climate mitigation. Frontiers in Ecology and the Environment, 15(1), 42-50.

Hugelius, G., Loisel, J., Chadburn, S., Jackson, R. B., Jones, M., MacDonald, G., . . . Treat, C. (2020). Large stocks of peatland carbon and nitrogen are vulnerable to permafrost thaw. Proceedings of the National Academy of Sciences, 117(34) (pp. 20438-20446). National Academy of Sciences.

Humpenöder, F., Karstens, K., Lotze-Campen, H., Leifeld, J., Menichetti, L., Barthelmes, A., & Popp, A. (2020). Peatland protection and restoration are key for climate change mitigation. Environmental Research Letters, 15(10), 104093.

Hutchinson, G. E. (1975). A treatise on limnology: vol III limnological botany. New York: Wiley.

Ion, I. V., & Ene, A. (2021). Evaluation of Greenhouse Gas Emissions from Reservoirs: A Review. Sustainability, 13(21), 11621.

IPCC. (2013). Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. In T. F.-K. Stocker. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

IPCC. (2014). 2013 Supplement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories: Wetlands, Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (eds). Switzerland: IPCC.

IPCC 2019a. Climate Change and Land: An IPCC Special Report on Climate Change, Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems. Shukla, P. R., Skea, J., Calvo Buendia, E. et al. (eds). IPCC

IPCC 2019b.

IPCC 2022. Climate Change 2022: Mitigation of Climate Change. Contribution of Working Group III to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change." Shukla, P. R., Skea, J., Slade, R., et al. (eds) Cambridge University Press: Cambridge University Press: Cambridge, UK and New York, NY, USA

Jakubínský, et al. (2021). Managing floodplains using nature-based solutions to support multiple ecosystem functions and services. Wiley Interdisciplinary Reviews: Water, 8(5), e1545.

Jastrow, J. D., Amonette, J. E., & Bailey, V. L. (2007). Mechanisms controlling soil carbon turnover and their potential application for enhancing carbon sequestration. Climatic Change, 80(1), 5-23.

Jessop, J., Spyreas, G., Pociask, G. E., Benson, T. J., Ward, M. P., Kent, A. D., & Matthews, J. W. (2015). Tradeoffs among ecosystem services in restored wetlands. Biological Conservation, 191, 341-348.

Joosten, H. (2015). Peatlands, climate change mitigation and biodiversity conservation: An issue brief on the importance of peatlands for carbon and biodiversity conservation and the role of drained peatlands as greenhouse gas emission hotspots (Vol. 2015727). Nordic Council of Ministers.

Joosten, H. (2016). Peatlands across the globe. In In Peatland restoration and ecosystem services: Science, policy and practice (Vol. 2016) (pp. 19-43). Cambridge, UK: Cambridge University Press.

Joosten, H., Tapio-Biström, M. L., & Tol, S. (2012). Peatlands: guidance for climate change mitigation through conservation, rehabilitation and sustainable use. Rome: Food and Agriculture Organization of the United Nations.

Keesstra, S., Nunes, J., Novara, A., Finger, D., Avelar, D., Kalantari, Z., & Cerdà, A. (2018). The superior effect of nature based solutions in land management for enhancing ecosystem services. Science of the Total Environment, 610, 997-1009.

Keller, P. S., Marcé, R., Obrador, B., & Koschorreck, M. (2021). Global carbon budget of reservoirs is overturned by the quantification of drawdown areas. Nature Geoscience, 14(6), 402-408.

Kennedy, H., Beggins, J., Duarte, C. M., Fourqurean, J. W., Holmer, M., Marbà, N., & Middelburg, J. J. (2010). Seagrass sediments as a global carbon sink: Isotopic constraints. Global biogeochemical cycles, 24(4).

Kirpotin, S. N., Antoshkina, O. A., Berezin, A. E., Elshehawi, S., Feurdean, A., Lapshina, E. D., . . . Volkov, I. V. (2021). Great Vasyugan Mire: How the world's largest peatland helps addressing the world's largest problems. Ambio, 50(11), 2038-2049.

Kirwan, M. L., & Megonigal, J. P. (2013). Tidal wetland stability in the face of human impacts and sea-level rise. Nature, 504(7478), 53-60.

Kolka , R., Bridgham, S. D., & Ping, C. L. (2016). Soils of peatlands: histosols and gelisols. In M. M.J. Vepraskas, & C. (. Craft, In Wetlands soils: genesis, hydrology, landscapes and classification. Boca Raton, FL.

Kolka, R., Trettin, C., Tang, W., Krauss, K., Bansal, S., et al. (2018). Terrestrial wetlands. In N. Cavallaro, G. Shrestha, R. Birdsey, A. Mayes, R. Najjar, S. Reed, . . . Z. e. Zhu, Second state of the carbon cycle report (SOCCR2): A sustained assessment report (pp. 507-567., 507-567). Washington, DC: US Global Change Research Program: 507-567., 507-567.

Kortelainen, P., Larmola, T., Rantakari, M., Juutinen, S., Alm, J., & Martikainen, P. J. (2020). Lakes as nitrous oxide sources in the boreal landscape. Global change biology, 26(3), 1432-1445.

Kreyling , J., Tanneberger, F., Jansen, F., van der Linden, S., Aggenbach, C., Blüml, V., . . . Kotowski , W. (2021). Rewetting does not return drained fen peatlands to their old selves. Nature communications, 12(1), 1-8.

Kroeger, K.D., Crooks, S., Moseman-Valtierra, S., et al. . (2017). Restoring tides to reduce methane emissions in impounded wetlands: A new and potent Blue Carbon climate change intervention . Scientific Report 7, 11914 .

LaRose , S., Price, J., & Rochefort, L. (1997). Rewetting of a cutover peatland: hydrologic assessment. Wetlands, 17(3), 416-423.

Lauerwald, R., Regnier, P., Figueiredo, V.,Enrich-Prast, A., Bastviken, D., et al. (2019). Natural lakes are a minor global source of N2O to the atmosphere. Global Biogeochemical Cycles, 33(12), 1564-1581.

Lavorel, S., Colloff, M. J., Mcintyre, S., Doherty, M. D., Murphy, H. T., Metcalfe, D. J., . . . Williams, K. J. (2015). Ecological mechanisms underpinning climate adaptation services. Global change biology, 21(1), 12-31.

Leaman, D. (2018). Medicinal Plants in Wetlands. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Li, Y., Shang, J., Zhang, C., Zhang, W., Niu, L., Wang, L., & Zhang, H. (2021). The role of freshwater eutrophication in greenhouse gas emissions: A review. Science of The Total Environment, 144582.

Liquete, C., Udias, A., Conte, G., Grizzetti, B., & Masi, F. (2016). Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits. Ecosystem Services, 22, 392-401.

Lo, W., Huang, C. T., Wu, M. H., Doong, D. J., Tseng, L. H., Chen, C. H., & Chen, Y. J. (2021). Evaluation of Flood Mitigation Effectiveness of Nature-Based Solutions Potential Cases with an Assessment Model for Flood Mitigation. Water, 13(23), 3451.

Loisel, J., & Bunsen, M. (2020). Abrupt fen-bog transition across southern patagonia: Timing, causes, and impacts on carbon sequestration. Frontiers in Ecology and Evolution, 273.

Mäkinen, K., & Khan, S. (2010). Policy Considerations for Greenhouse Gas Emissions from Freshwater Reservoirs. Water alternatives, 3(2).

Maavara, T., Chen, Q., Van Meter, K., Brown, L. E., Zhang , J., Ni, J., & Zarfl, C. (2020). River dam impacts on biogeochemical cycling. Nature Reviews Earth & Environment, 1(2), 103-116.

Maavara, T., Lauerwald, R., Laruelle, G. G., Akbarzadeh, Z., Bouskill, N. J., Van Cappellen, P., & Regnier, P. (2019). Nitrous oxide emissions from inland waters: Are IPCC estimates too high? Global change biology, 25(2), 473-488.

Mackenzie, S. (2018). Managing Urban Waste Water. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Mafabi, P. (2018). National Wetland Policy: Uganda. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Malone, S. L., Starr, G., Staudhammer, C. L., & Ryan, M. G. (2013). Effects of simulated drought on the carbon balance of Everglades short-hydroperiod marsh. Global Change Biology, 19(8), 2511-2523.

Mandishona, E., & Knight, J. (2022). Feedbacks and Trade-Offs in the Use of Wetland Ecosystem Services by Local Communities in Rural Zimbabwe. Sustainability, 14(3), 1789.

Manzello, D. P., Enochs, I. C., Melo, N., Gledhill, D. K., & Johns, E. M. (2012). Ocean acidification refugia of the Florida Reef Tract. PLoS ONE, 7, e41715.

Marescaux , A., Thieu, V., & Garnier, J. (2018). Carbon dioxide, methane and nitrous oxide emissions from the human-impacted Seine watershed in France. Science of the Total Environment, 643, 247-259.

Mauroner & Anisha, et al. (2021). Operationalizing NBS in low-and middle-income countries: Redefining and “greening” project development. In In Nature-based Solutions and Water Security (pp. 423-443). Elsevier.

Mazurczyk, T., & Brooks, R. P. (2018). Carbon storage dynamics of temperate freshwater wetlands in Pennsylvania. Wetlands Ecology and Management, 26(5), 893-914.

McCrackin, M. L., Jones, H. P., Jones, P. C., & Moreno-Mateos, D. (2017). Recovery of lakes and coastal marine ecosystems from eutrophication: A global meta-analysis. Limnology and Oceanography, 62(2), 507-518.

McInnes, R. (2018a). Flood Management and the Role of Wetlands. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

McInnes, R. J. (2018b). Local Climate Regulation by Urban Wetlands. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., . . . Silliman, B. R. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering. Frontiers in Ecology and the Environment, 9(10), 552-560.

Mendonça, R., Müller, R. A., Clow, D., Verpoorter, C., Raymond, P., Tranvik, L. J., & Sobek, S. (2017). Organic carbon burial in global lakes and reservoirs. Nature communications, 8(1), 1-7.

Menichetti, L., & Leifeld, J. (2018). The weight of peatland conservation and restoration in the global cycle of C and N. EGU General Assembly Conference Abstracts (p. 6672). European Geosciences Union.

Millennium ecosystem assessment,MEA. (2005). Ecosystems and human well-being (Vol. 5). Washington, DC: Island press.

Ministry of Water and Environment, Uganda. (2016). Uganda’s Intended Nationally Determined Contribution (INDC). Retrieved from United Nations Framework Convention on Climate Change: https://www4.unfccc.int/sites/ndcstaging/PublishedDocuments/Uganda%20First/INDC%20Uganda%20final%20%2014%20October%20%202015.pdf

Ministry of Water and Environment, Uganda. (2021). Uganda’s Interim Nationally Determined Contribution (NDC). Retrieved from United Nations Framework Convention on Climate Change: https://www4.unfccc.int/sites/ndcstaging/PublishedDocuments/Uganda%20First/Uganda%20interim%20NDC%20submission_.pdf

Mitra, S., Wassmann, R., & Vlek , P. L. (2005). An appraisal of global wetland area and its organic carbon stock. Current Science, 88, 25–35.

Mitra, S., Wassmann, R., & Vlek, P. L. (2003). Global inventory of wetlands and their role in the carbon cycle (No. 1546-2016-132267).

Mitsch , W. J., & Gosselink, J. G. (2015). Wetlands, 5th edn. Hoboken: JohnWiley & Sons, Inc.

Mitsch, W. J. (1992). Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecological Engineering, 1(1-2), 27-47.

Mitsch, W. J. (2005b). Wetland creation, restoration, and conservation: a wetland Invitational at the Olentangry river wetland research park. Journal Ecological Engineering, 241-251.

Mitsch, W. J., Zhang, L., Anderson, C. J., Altor, A. E., & Hernández, M. E. (2005a). Creating riverine wetlands: ecological succession, nutrient retention, and pulsing effects. Ecological Engineering, 25(5), 510-527.

Mitsch, W.J., Bernal, B., Nahlik, A.M. et al. . (2013). Wetlands, carbon, and climate change. Landscape Ecology 28, 583–597.

Mitsch, W.J., Nahlik, A., Wolski, P. et al. . (2010). Tropical wetlands: seasonal hydrologic pulsing, carbon sequestration, and methane emissions. Wetlands Ecological Management 18, 573–586.

Moomaw, W. R., Chmura, G. L., Davies, G. T., Finlayson, C. M., et al. (2018). Wetlands in a changing climate: science, policy and management. Wetlands, 38(2), 183-205.

Moore, P. A., Pypker, T. G., & Waddington, J. M. (2013). Effect of long-term water table manipulation on peatland evapotranspiration. Agricultural and Forest Meteorology, 178, 106-119.

Morelli, T. L., et al. (2016). Managing climate change refugia for climate adaptation. PLoS One, 11(8), e0159909.

Murdiyarso, D., Saragi-Sasmito, M. F., & Rustini, A. (2019). Greenhouse gas emissions in restored secondary tropical peat swamp forests. Mitigation and Adaptation Strategies for Global Change, 24(4), 507-520.

Nahlik, A. M., & Fennessy, M. S. (2016). Carbon storage in US wetlands. Nature Communications, 7(1), 1-9.

Nahlik, A. M., & Mitsch, W. J. (2010). Methane emissions from created riverine wetlands. Wetlands, 30, 783–793.

Natchimuthu, S., Wallin, M. B., Klemedtsson, L., & Bastviken, D. (2017). Spatio-temporal patterns of stream methane and carbon dioxide emissions in a hemiboreal catchment in Southwest Sweden. Scientific reports, 7(1), 1-12.

Neubauer, S. C. (2014). On the challenges of modeling the net radiative forcing of wetlands: reconsidering Mitsch et al. 2013. Landscape Ecology, 29(4), 571-577.

Neubauer, S. C., & Verhoeven, J. T. (2019). Wetland effects on global climate: mechanisms, impacts, and management recommendations. In A. &. (eds), In Wetlands: ecosystem services, restoration and wise use (pp. 39-62). Cham: Springer.

Noe, G. B., & Hupp, C. R. (2005). Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. Ecological Applications, 15(4), 1178-1190.

Nugent, K. A., Strachan, I. B., Roulet, N. T., Strack, M., Frolking, S., & Helbig, M. (2019). Prompt active restoration of peatlands substantially reduces climate impact. Environmental Research Letters, 14(12), 124030.

Nugent, K. A., Strachan, I. B., Strack, M., Roulet, N. T., & Rochefort, L. (2018). Multi-year net ecosystem carbon balance of a restored peatland reveals a return to carbon sink. Global Change Biology, 24(12), 5751-5768.

Omengo, F. O., Geeraert, N., Bouillon, S., & Govers, G. (2016). Deposition and fate of organic carbon in floodplains along a tropical semiarid lowland river (Tana River, Kenya). Journal of Geophysical Research. Biogeosciences, 121(4), 1131-1143.

Opperman, J. J., Galloway, G. E., Fargione, J., Mount, J. F., Richter , B. D., & Secchi, S. (2009). Sustainable floodplains through large-scale reconnection to rivers. Science, 326(5959), 1487-1488.

Ott, C. A., & Chimner, R. A. (2016). Long-term peat accumulation in temperate forested peatlands (Thuja occidentalis swamps) in the Great Lakes region of North America. Mires & Peat, 18.

Paerl, et al. (2020). Mitigating eutrophication and toxic cyanobacterial blooms in large lakes: The evolution of a dual nutrient (N and P) reduction paradigm. Hydrobiologia, 847(21), 4359-4375.

Page, K. L., & Dalal, R. C. (2011a). Contribution of natural and drained wetland systems to carbon stocks, CO2, N2O, and CH4 fluxes: an Australian perspective. Soil Research, 49(5), 377-388.

Page, S. E., Rieley, J. O., & Banks, C. J. (2011b). Global and regional importance of the tropical peatland carbon pool. Global Change Biology, 17, 798–818.

Papayannis, T., & Pritchard, D. (2018). Cultural, Aesthetic, and Associated Wetland Ecosystem Services. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Peimer, A. W., Krzywicka, A. E., Cohen, D. B., Van den Bosch, K., Buxton, V. L., Stevenson, N. A., & Matthews, J. W. (2017). National-Level Wetland Policy Specificity and Goals Vary According to Political and Economic Indicators. Environmental Management 59, 141–153.

Pennock, D., Yates, T., Bedard-Haughn, A., Phipps, K., et al. (2010). Landscape controls on N2O and CH4 emissions from freshwater mineral soil wetlands of the Canadian Prairie Pothole region. Geoderma, 155(3-4), 308-319.

Peralta, A. L., Muscarella, M. E., Matthews, J. W., Zak, D., & Ardon-Sayao, M. (2017). Wetland management strategies lead to tradeoffs in ecological structure and function. Elementa: Science of the Anthropocene, 5.

Perosa, F., Gelhaus, M., Zwirglmaier, V., Arias-Rodriguez, L. F., Zingraff-Hamed, A., Cyffka, B., & Disse, M. (2021). Integrated valuation of nature-based solutions using tessa: three floodplain restoration studies in the Danube catchment. Sustainability 13(3), 1482.

Pester, M., Knorr, K. H., Friedrich, M. W., Wagner, M., & Loy, A. (2012). Sulfate-reducing microorganisms in wetlands – famleless actors in carbon cycling and climate change. (Frontiers in microbiology, 3, 72.

Pfeifer-Meister, L., Gayton, L. G., Roy, et al. (2018). Greenhouse gas emissions limited by low nitrogen and carbon availability in natural, restored, and agricultural Oregon seasonal wetlands. PeerJ, 6, e5465.

Phillips, R., & Beeri, O. (2008). The role of hydropedologic vegetation zones in greenhouse gas emissions for agricultural wetland landscapes. Catena, 72, 386–394.

Pidgeon, E. (2009). Carbon sequestration by coastal marine habitats: Missing sinks. In D. Laffoley, & G. Grimsditch, The Management of Natural Coastal Carbon Sinks (pp. 47–51). Gland, Switzerland: IUCN.

Pinsonneault, A. J., Moore, T. R., & Roulet, N. T. (2016). Temperature the dominant control on the enzyme-latch across a range of temperate peatland types. Soil Biology and Biochemistry, 97, 121-130.

Plouviez, M., Shilton, A., Packer, M. A., & Guieysse, B. (2019). Nitrous oxide emissions from microalgae: potential pathways and significance. Journal of Applied Phycology, 31(1), 1-8.

Prairie, Y. T., Mercier-Blais, S., Harrison, J. A., Soued, C., del Giorgio, P., Harby, A., . . . Nahas, R. (2021). A new modelling framework to assess biogenic GHG emissions from reservoirs: The G-res tool. Environmental Modelling & Software, 143, 105117.

Pullanikkatil, D., Mograbi, P. J., Palamuleni, L., Ruhiiga, T., & Shackleton, C. (2020). Unsustainable trade-offs: Provisioning ecosystem services in rapidly changing Likangala River catchment in southern Malawi. Environment, Development and Sustainability, 22(2), 1145-1164.

Qu, B., Aho, K. S., Li, C., Kang, S., Sillanpää , M., Yan, F., & Raymond, P. A. (2017). Greenhouse gases emissions in rivers of the Tibetan Plateau. Scientific reports, 7(1), 1-8.

Quick, A. M., Reeder, W. J., Farrell, T. B., Tonina, D., Feris, K. P., & Benner,, S. G. (2019). Quick, A. M., Reeder, W. J., Farrell, T. B., Tonina, D., Feris, K. P., & Benner, S. G. (2019). Nitrous oxide from streams and rivers: A review of primary biogeochemical pathways and environmental variables. Earth-science reviews, 191, 224-262.

Ramesh, R., Banerjee, K., Paneerselvam, A., Raghuraman , R., Purvaja, R., & Lakshmi, A. (2019). Importance of seagrass management for effective mitigation of climate change. In R. R. Krishnamurthy, M. P. Jonathan, S. Srinivasalu, & B. Glaeser, Coastal Management: Global Challenges and Innovations (pp. 283-299). Academic Press.

Ramsar Convention. (2005). Resolution IX.1 Annex A: A Conceptual Framework for the wise use of wetlands and the maintenance of their ecological character. 9th Meeting of the Conference of the Contracting Parties to the Convention on Wetlands. Kampala, Uganda: Secretariat of the Convention on Wetlands, Gland, Switzerland.

Ramsar Convention Fact Sheet 6. (2015). The Ramsar Convention: What's it all about? Retrieved from Ramsar Convention on Wetlands: https://www.ramsar.org/sites/default/files/documents/library/fs_6_ramsar_convention_0.pdf

Ramsar Convention on Wetlands. (2016). An Introduction to the Ramsar Convention on Wetlands, 7th ed. (previously The Ramsar Convention Manual). Gland, Switzerland: Ramsar Convention Secretariat.

Ramsar Convention on Wetlands. (2018). Global Wetland Outlook: State of the World's Wetlands and their Services to People. Gland, Switzerland: Ramsar Convention Secretariat.

Ramsar Convention on Wetlands. (2018a). Resolution XIII.13 Restoration of degraded peatlands to mitigate and adapt to climate change and enhance biodiversity and disaster risk reduction. Dubai, United Arab Emirates: Ramsar Convention on Wetlands.

Ramsar Convention on Wetlands. . (2002). Resolution VIII.6: A Ramsar Framework for Wetland Inventory. Wetlands: water, life, and culture. 8th Meeting of the Conference of the Contracting Parties to the Convention on Wetlands.

Ramsar Convention Secretariat. (1996). STRATEGIC PLAN 1997-2002. Retrieved from 6th Meeting of the Conference of the Contracting Parties : https://www.ramsar.org/sites/default/files/documents/pdf/key_strat_plan_1997_e.pdf

Ramsar Convention Secretariat. (2008). Resolution X.1 The Ramsar Strategic Plan 2009-2015. Retrieved from 10th Meeting of the Conference of the Parties to the Convention on Wetlands : https://www.ramsar.org/sites/default/files/documents/library/key_res_x_01_e.pdf

Ramsar Convention Secretariat. (2010). National Wetland Policies: Developing and implementing National Wetland Policies. Ramsar handbooks for the wise use of wetlands, 4th edition, vol. 2. Gland, Switzerland: Ramsar Convention Secretariat.

Ramsar Convention Secretariat. (2012). Briefing Note 4. Retrieved from The benefits of wetland restoration: https://www.ramsar.org/sites/default/files/documents/library/bn4-en.pdf

Ramsar Convention Secretariat. (2015). Resolution XII.2 The Ramsar Strategic Plan 2016-2024. Retrieved from 12th Meeting of the Conference of the Parties to the Convention on Wetlands (Ramsar, Iran, 1971): https://www.ramsar.org/sites/default/files/documents/library/cop12_res02_strategic_plan_e_0.pdf

Ramsar Convention Secretariat. (2016). An Introduction to the Ramsar Convention on Wetlands, 7th ed. . Gland, Switzerland: Ramsar Convention Secretariat.

Ramsar Convention Secretariat. (2018). Briefing Note 10. Retrieved from Wetland Restoration for Climate Change Resilience: https://www.ramsar.org/sites/default/files/documents/library/bn10_restoration_climate_change_e.pdf

Ramsar Convention website. (2022, February 28). Contracting Parties to the Ramsar Convention. Retrieved from Ramsar Country Profiles: http://www.ramsar.org/sites/default/files/documents/library/annotated_contracting_parties_list_e.pdf

Ran, L., Lu, X. X., Richey, J. E., Sun, H., Han, J., Yu, R., . . . Yi, Q. (2015). Long-term spatial and temporal variation of CO2 partial pressure in the Yellow River, China. Biogeosciences, 12, 921–932.

Raymond, P., Hartmann, J., Lauerwald, R. et al. (2013). Global carbon dioxide emissions from inland waters. Nature, 503(7476), 355-359.

Renou-Wilson, F., Moser, G., Fallon, D., Farrell, C. A., Müller, C., & Wilson, D. (2019). Rewetting degraded peatlands for climate and biodiversity benefits: Results from two raised bogs. Ecological Engineering, 127, 547-560.

Richards, B., & Craft, C. B. (2015). Greenhouse gas fluxes from restored agricultural wetlands and natural wetlands, Northwestern Indiana. In The role of natural and constructed wetlands in nutrient cycling and retention on the landscape (pp. 17-32). Cham: Springer.

Rieley, J., & Page, S. (2016). Tropical peatland of the world. In In Tropical peatland ecosystems (pp. 3-32). Tokyo: Springer.

Robertson, A. I., Bunn, S. E., Boon, P. I., & Walker, K. F. (1999). Sources, sinks and transformations of organic carbon in Australian floodplain rivers. Marine and Freshwater Research, 50(8), 813-829.

Rozenberg, J., & Fay, M. (2019). Beyond the gap: How countries can afford the infrastructure they need while protecting the planet. Washington, DC: World Bank.

Russell, M., & Greening, H. (2015). Estimating benefits in a recovering estuary: Tampa Bay, Florida. Estuaries and Coasts, 38(1), 9-18.

Ruwaimana, M., Anshari, G. Z., & Gavin, D. G. (2020). Environmental archives and carbon storage from Inland Tropical Peat in West Kalimantan, Indonesia. AGU Fall Meeting Abstracts Vol. 2020 (pp. EP042-04). American Geophysical Union.

Saderne, V., Geraldi, N. R., Macreadie, P. I., Maher, D. T., Middelburg, J. J., Serrano, O., . . . Fourqurean, J. W. (2019). Role of carbonate burial in Blue Carbon budgets. Nature communications, 10(1), 1-9.

Salimi, S., Almuktar, S. A., & Scholz, M. (2021). (2021). Impact of climate change on wetland ecosystems: A critical review of experimental wetlands. Journal of Environmental Management, 286, 112160.

Sanches, L. F., Guenet, B., Marinho, C. C., Barros, N., & de Assis Esteves, F. (2019). Global regulation of methane emission from natural lakes. Scientific reports, 9(1), 1-10.

Sanders, et al. (2017). Carbon accumulation in Amazonian floodplain lakes: A significant component of Amazon budgets? Limnology and Oceanography 2, 29–35.

Sanon, S., Hein, T., Douven, W., & Winkler, P. (2012). Quantifying ecosystem service trade-offs: The case of an urban floodplain in Vienna, Austria. Journal of environmental management, 111, 159-172.

Saunois, et al. (2020). The global methane budget 2000–2017. Earth System Science Data, 12(3), 1561-1623.

Sawakuchi, H. O., Bastviken, D., Sawakuchi, A. O., Ward, N. D., Borges, C. D., et al. (2016). Oxidative mitigation of aquatic methane emissions in large Amazonian rivers. Global change biology, 22(3), 1075-1085.

Scott, D. N., & Wohl, E. E. (2018). Geomorphic regulation of floodplain soil organic carbon concentration in watersheds of the Rocky and Cascade Mountains, USA. Earth Surface Dynamics, 6(4), 1101-1114.

Segarra, K. E. A., Schubotz, F., Samarkin, V., Yoshinaga, M. Y., et al. (2015). High rates of anaerobic methane oxidation in freshwater wetlands reduce potential atmospheric methane emissions. Nature Communications, 6(1), 1-8.

Seifollahi-Aghmiuni, S., Nockrach, M., & Kalantari, Z. (2019). The potential of wetlands in achieving the sustainable development goals of the 2030 Agenda. Water, 11(3), 609.

Seitzinger , S. P., Kroeze, C., & Styles, R. V. (2000). Global distribution of N2O emissions from aquatic systems: natural emissions and anthropogenic effects. Chemosphere-Glob. Change Sci. 2, 267–279.

Serrano, O., Gómez-López , D. I., Sánchez-Valencia, L., Acosta-Chaparro , A., Navas-Camacho, R., González-Corredor, J., . . . Marbà, N. (2021). Seagrass blue carbon stocks and sequestration rates in the Colombian Caribbean. Scientific reports, 11(1), 1-12.

Sheehan, L., Crooks, S., Tomasko, D., Robison, O., Connell, H., & Quinton, B. (2017). Tampa Bay Blue Carbon Assessment: Summary of Findings. Tampa Bay Estuary Program Technical Report #07-16 & #07a- 16. St. Petersburg, FL: Tampa Bay Estuary Program.

Shen, Z., Rosenheim, B. E., Törnqvist, T. E., & Lang, A. (2021). Engineered Continental-Scale Rivers Can Drive Changes in the Carbon Cycle. AGU Advances, 2(1), e2020AV000273.

Sherwood, E. T., Greening, H. S., Janicki, A. J., & Karlen, D. J. (2016). Tampa Bay estuary: Monitoring long-term recovery through regional partnerships. Regional Studies in Marine Science, 4, 1-11.

Sherwood, E. T., Greening, H. S., Johansson, J. R., Kaufman, K., & Raulerson, G. E. (2017). Tampa Bay (Florida, USA) documenting seagrass recovery since the 1980's and reviewing the benefits. Southeastern Geographer, 57(3), 294-319.

Silva, J. P., Canchala, T. R., Lubberding, H. J., Peña, E. J., & Gijzen, H. J. (2016). Greenhouse gas emissions from a tropical eutrophic freshwater wetland. International. International Journal of Environmental and Ecological Engineering, 10(5), 541-547.

Sinha, E., Michalak, A. M., Calvin, K. V., & Lawrence, P. J. (2019). Societal decisions about climate mitigation will have dramatic impacts on eutrophication in the 21st century. Nature communications, 10(1), 1-11.

Smaczniak, K. (2018). National Wetland Policy: USA. In F. C. (eds), In The Wetland Book. Springer, Dordrecht.

Sorensen, K. W. (1993). Indonesian peat swamp forests and their role as a carbon sink. Chemosphere, 27(6), 1065-1082.

Spalding, Mark D and Leal, Maricé (editors). (2021). The State of the World's Mangroves 2021. Global Mangrove Alliance.

Stanley, E. H., Casson, N. J., Christel, S. T., Crawford , J. T., Loken, L. C., & Oliver, S. K. (2016). Stanley, E. H., Casson, N. J., Christel, S. T., Crawford, J. T., Loken, L. C., & Oliver, S. K. (2016). The ecology of methane in streams and rivers: patterns, controls, and global significance. Ecological Monographs, 86(2), 146-171.

Stirling, E., Fitzpatrick, R. W., & Mosley, L. M. (2020). (2020). Drought effects on wet soils in inland wetlands and peatlands. Earth-Science Reviews, 210, 103387.

Ström, L., Mastepanov, M., & Christensen, T. R. (2005). Species-specific effects of vascular plants on carbon turnover and methane emissions from wetlands. Biogeochemistry, 75, 65– 82.

Su, H., Chen, J., Wu, Y., Chen, J.,et al. (2019). Morphological traits of submerged macrophytes reveal specific positive feedbacks to water clarity in freshwater ecosystems. Science of the Total Environment, 684, 578-586.

Suárez-Delucchi, A. (2018). National Wetland Policy: Chile. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Sun, R., Chen, A., Chen, L., & Lü, Y. (2012). Cooling effects of wetlands in an urban region: The case of Beijing. Ecological Indicators, 20, 57-64.

Sutfin, N. A., & Wohl, E. (2017). Substantial soil organic carbon retention along floodplains of mountain streams. Journal of Geophysical Research. Earth Surface, 122(7), 1325-1338.

Sutfin, N. A., Wohl, E. E., & Dwire, K. A. (2016). Banking carbon: a review of organic carbon storage and physical factors influencing retention in floodplains and riparian ecosystems. Earth Surface Processes and Landforms, 41(1), 38-60.

Tabari, H. (2020). Climate change impact on flood and extreme precipitation increases with water availability. Scientific reports, 10(1), 1-10.

Tan, L., Ge, Z., Zhou, X., Li, S., Li, X., & Tang, J. (2020). Conversion of coastal wetlands, riparian wetlands, and peatlands increases greenhouse gas emissions: A global meta-analysis. Global change biology, 26(3), 1638-1653.

Tangen, B. A., & Bansal, S. (2020). Soil organic carbon stocks and sequestration rates of inland, freshwater wetlands: Sources of variability and uncertainty. Science of The Total Environment, 749, 141444.

Tanneberger, F., Tegetmeyer, C., Busse, S., Barthelmes, A., et al. (2017). The peatland map of Europe. Mires Peat, 19, 1–17.

Tete, E., Viaud, V., & Walter, C. (2015). Organic carbon and nitrogen mineralization in a poorly-drained mineral soil under transient waterlogged conditions: an incubation experiment. European Journal of Soil Science, 66(3), 427-437.

Tian, H., Xu, R., Canadell, J. G., Thompson, R. L., et al. (2020). A comprehensive quantification of global nitrous oxide sources and sinks. Nature, 586(7828), 248-256.

Tomasko et al. (2015). Quantification of Carbon Sequestration Rates for Seagrass. Submitted to Restore America's Estuaries and Tampa Bay Estuary Program.

Tomscha, S. A., Bentley, S., Platzer, E., Jackson, B., de Roiste, M., Hartley, S., . . . Deslippe, J. R. (2021). Multiple methods confirm wetland restoration improves ecosystem services. Ecosystems and People, 17(1), 25-40.

Tomscha, S. A., Gergel, S. E., & Tomlinson, M. J. (2017). The spatial organization of ecosystem services in river-floodplains. Ecosphere, 8(3), e01728.

Tong, C., Wang, W. Q., Huang, J. F., Gauci, V., et al. (2012). Invasive alien plants increase CH4 emissions from a subtropical tidal estuarine wetland. Biogeochemistry, 111, 677– 693.

Tremblay, A., Varfalvy, L., Garneau, M., & Roehm, C. (2005). Greenhouse gas emissions-fluxes and processes: hydroelectric reservoirs and natural environments. Berlin: Springer Science & Business Media.

UNEP & IUCN. (2021). Nature-based solutions for climate change mitigation. Nairobi and Gland: United Nations Environment Programme (UNEP), Nairobi and International Union for Conservation of Nature (IUCN), Gland.

UNEP. (2014). Green Infrastructure Guide for Water Management: Ecosystem-based management approaches for water-related infrastructure projects. United Nations Environment Programme (UNEP).

UNEP. (2020). Out of the blue: The value of seagrasses to the environment and to people. Nairobi: United Nations Environment Programme.

UNESCO & UN-Water. (2020). United Nations World Water Development Report 2020: Water and Climate Change. Paris, France: UNESCO.

UNESCO/IHP. (2008). Scoping paper on Assessment of the GHG Status of Freshwater Reservoirs. The United Nations Educational, Scientific and Cultural Organization (UNESCO) & International Hydrological Programme (IHP).

Unsworth, R. K., Nordlund, L. M., & Cullen-Unsworth, L. C. (2018). Seagrass meadows support global fisheries production. Conservation Letters.

Unsworth, R.K.F., McKenzie, L.J., Collier, C.J. et al. (2019). Global challenges for seagrass conservation. Ambio 48, 801–815.

Urák, I., Hartel, T., Gallé, R., & Balog, A. (2017). Worldwide peatland degradations and the related carbon dioxide emissions: the importance of policy regulations. Environmental Science & Policy, 69, 57-64.

Vanselow-Algan, M., Schmidt, S. R., Greven,, M., Fiencke, C., Kutzbach, L., & Pfeiffer, E. M. (2015). High methane emissions dominated annual greenhouse gas balances 30 years after bog rewetting. Biogeosciences, 12(14), 4361-4371.

Verhoeven, J. T., & Setter, T. L. (2010). Agricultural use of wetlands: opportunities and limitations. Annals of Botany, 105(1), 155–163.

Verpoorter, C., Kutser, T., Seekell, D. A., & Tranvik, L. J. (2014). A global inventory of lakes based on high-resolution satellite imagery. Geophysical Research Letters, 41(18), 6396-6402.

Vigerstol, K., Abell, R., Brauman, K., Buytaert, W., & Vogl, A. (2021). Addressing water security through nature-based solutions. In In Nature-based Solutions and Water Security (pp. 37-62). Elsevier.

Vymazal, J. (2018). Constructed Wetlands for Water Quality Regulation. In F. C. (eds), In The Wetland Book. Dordrecht: Springer.

Waldo, S., Deemer, B. R., Bair, L. S., & Beaulieu, J. J. (2021). Greenhouse gas emissions from an arid-zone reservoir and their environmental policy significance: Results from existing global models and an exploratory dataset. Environmental Science & Policy, 120, 53-62.

Webb, J. R., Hayes, N. M., Simpson, G. L., Leavitt, P. R., Baulch, H. M., & Finlay, K. (2019). Widespread nitrous oxide undersaturation in farm waterbodies creates an unexpected greenhouse gas sink. Proceedings of the National Academy of Sciences, 116(20).

Weil, R. R., & Brady, N. C. (2016). The Nature and Properties of Soils. Upper Saddle River, NJ: Pearson.

Wetlands International. (2022, March 3). Retrieved from Wetlands International: wetlands.org

Wetzel, R. G. (2001). Limnology: lake and river ecosystems. Gulf Professional Publishing.

Wohl, E., Hall Jr, R. O., Lininger, K. B., Sutfin, N. A., & Walters, D. M. (2017). Carbon dynamics of river corridors and the effects of human alterations. Ecological Monographs, 87(3), 379-409.

World Bank. (2009). Convenient Solutions for an Inconvenient Truth: Ecosystem-based Approaches to Climate Change. The World Bank.

World Bank. (2012). Inclusive green growth: The pathway to sustainable development. Washington, DC: World Bank.

Xu, J., Morris, P. J., Liu, J., & Holden, J. (2018). PEATMAP: Refining estimates of global peatland distribution based on a meta-analysis. Catena, 160, 134-140.

Yan, X., Thieu, V., & Garnier, J. (2021). Long-Term Evolution of Greenhouse Gas Emissions From Global Reservoirs. Frontiers in Environmental Science, 289.

Yao, Y., Tian , H., Shi, H., Pan, S., Xu, R., Pan, N., & Canadell, J. (2020). Increased global nitrous oxide emissions from streams and rivers in the Anthropocene. Nature Climate Change 10, 138–142.

Yates, K., Greening, H., & Morrison, G. (2011). Integrating science and resource management in Tampa Bay, Florida. U.S. Geological Survey Circular 1348.

Yu, Z. C. (2012). Northern peatland carbon stocks and dynamics: a review. Biogeosciences, 9(10), 4071-4085.

Yuan, K., Zhu, Q., Li, F., Riley, W. J., Torn, M., Chu, H., ... & Jackson, R. (2022). Causality guided machine learning model on wetland CH4 emissions across global wetlands. Agricultural and Forest Meteorology, 324, 109115.

Zhang et al. (2020). Significant methane ebullition from alpine permafrost rivers on the East Qinghai–Tibet Plateau. Nature Geoscience, 13, 349–35.

Zhang, W., Li, H., Xiao, Q., & Li, X. (2021). Urban rivers are hotspots of riverine greenhouse gas (N2O, CH4, CO2) emissions in the mixed-landscape chaohu lake basin. Water Research 189.

Zhang, Y., Li, C., Trettin, C. C., Li, H., & Sun, G. (2002). An integrated model of soil, hydrology, and vegetation for carbon dynamics in wetland ecosystems. Global Biogeochemical Cycles, 16(4), 9-1.

Zhang, Z., Zimmermann, N. E., Kaplan, J. O., & Poulter, B. (2016). Modeling spatiotemporal dynamics of global wetlands: comprehensive evaluation of a new sub-grid TOPMODEL parameterization and uncertainties. Biogeosciences, 13(5), 1387-1408.

Zhao, J., Malone, S. L., Oberbauer, S. F., Olivas, P. C., et al. (2019). Intensified inundation shifts a freshwater wetland from a CO2 sink to a source. Global change biology, 25(10), 3319-3333.