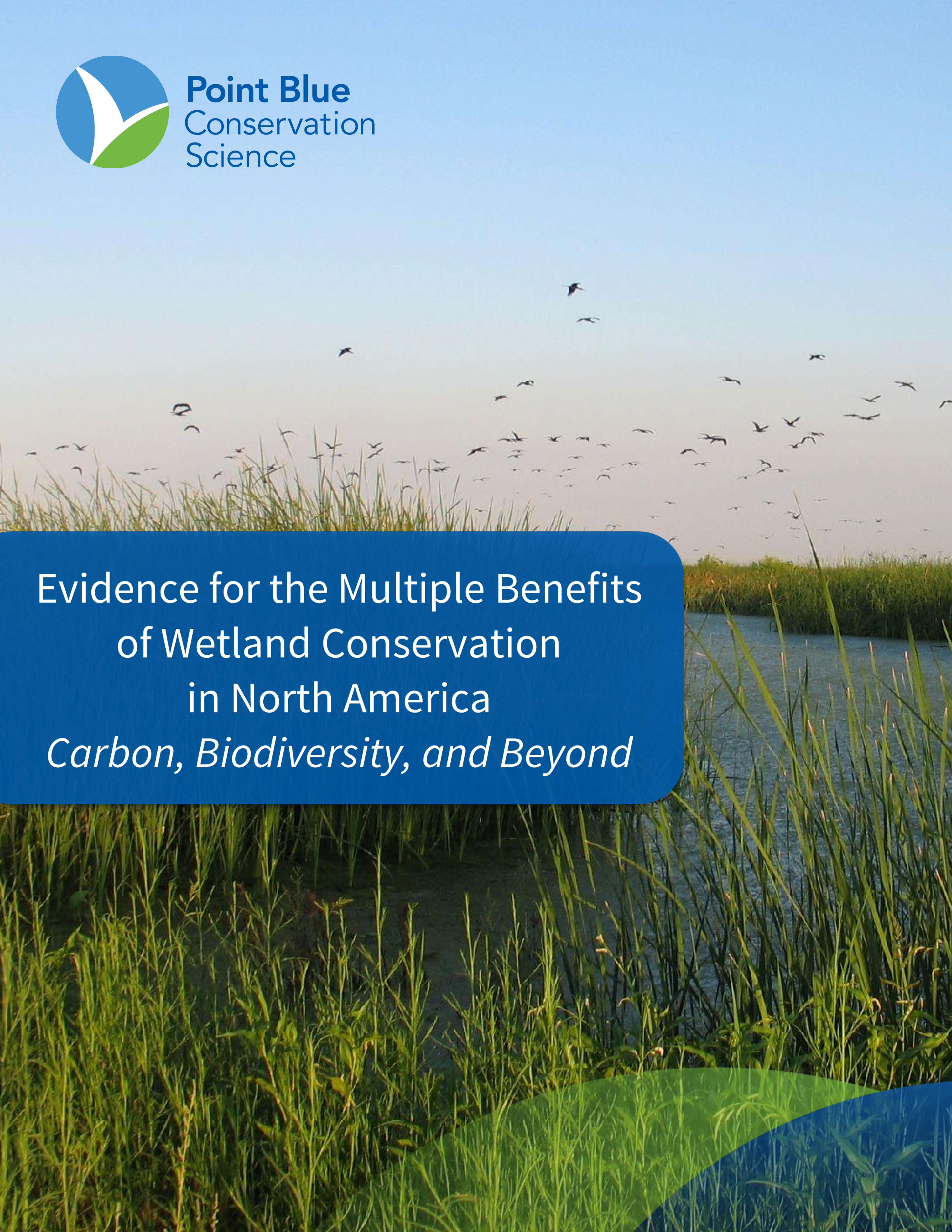

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Evidence for the Multiple Benefits of Wetland Conservation in North America

Carbon, Biodiversity, and Beyond

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Evidence for the Multiple Benefits of Wetland Conservation in North America: Carbon, Biodiversity, and Beyond

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Cover photo: Flock of White-faced Ibis flying over a managed wetland in the Tulare Basin (southern San Joaquin Valley) of California. Credit: Ryan DiGaudio/Point Blue

Point Blue Conservation Science – Point Blue's 160 scientists work to reduce the impacts of climate change, habitat loss, and other environmental threats while developing nature-based solutions to benefit both wildlife and people.

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Science Synthesis Overview

Bair Island Marine Park in Redwood City, CA
Photo by Tierney/Adobe Stock

Introduction

Wetlands currently make up approximately 5–8% of the global land surface (Mitsch and Gosselink 2007) and 5.5% of the land surface in the conterminous United States (Dahl 2011), but they play a disproportionately large role in the global carbon cycle (Bridgham et al. 2006; Poulter et al. 2021) and have one of the highest densities of ecosystem services of any terrestrial ecosystem on earth (Costanza et al. 1997). Wetlands are defined as any land that is saturated with water, at least intermittently (FGDC 2013), as the transitional area between true aquatic habitats and dry upland habitats, and as areas where the water table is at or near the surface (Cowardin 1979; Dahl 2011). They include a diverse array of geographic, hydrological, and climatological conditions from coastal mangroves and tidal marsh, to freshwater depressional wetlands and riparian floodplains, to montane meadows at high elevations. Despite their importance, an estimated 53-62% of wetlands in the conterminous United States have been lost, degraded, or converted to other land uses (Dahl and Johnson 1991; Bridgham et al. 2006), including over 20 million acres since the 1950s (Dahl 2011). Globally, estimates of wetland loss are more variable, ranging from 33% (Hu et al. 2017) to 87% (Davidson 2014), depending on wetland definitions and time periods examined. Remnant wetlands face increasing pressure from development, conversion to agricultural land, industrial pollution, and climate change (Dahl 2011; Wilson et al. 2022), highlighting the need for continued and expanded attention to wetland conservation and restoration efforts.

Global and national efforts to reach climate change mitigation and biodiversity conservation goals, such as the Paris Agreement (United Nations 2015) and America the Beautiful initiative (U.S. Department of the Interior et al. 2021), provide compelling motivation to advance efforts to protect and restore natural resources. To ensure wetlands are specifically and appropriately included in these global and national goals, we need a clear understanding of the scientific evidence for the broad array of wetland benefits. By synthesizing the scientific literature, these evidence assessments can inform policy decisions and implementation priorities for wetland conservation. Here, we summarize a subset of the vast scientific literature on wetland benefits by adapting a protocol for rapid evidence assessment (Varker et al. 2015). We used predefined search terms and screening criteria (described further in Appendix A) to identify studies that included data from wetlands in North America and addressed at least one of several broad categories of benefits we considered, including: water supply regulation, flood risk mitigation, water quality, soil health, biodiversity support, economic value, and carbon (Table 1). Our literature search and screening process narrowed down an initial set containing thousands of records to 662 for which we extracted detailed metadata, of which 457 specifically addressed carbon. Of these, we prioritized studies that provided representation across benefits categories and wetland classes, with a preference for empirical studies with large sample sizes, meta-analyses, and reviews. We ultimately identified 160 studies as high priority and most relevant to this synthesis, of which 79 addressed carbon and 81 addressed other benefits. We found that these high priority studies were frequently cited across the literature we reviewed, indicating that we successfully identified many of the most important studies. However, we also opportunistically added studies cited in the literature we reviewed or recommended to us during the development and review process, and we emphasize that our rapid evidence assessment was not exhaustive. We identify and discuss apparent gaps in the scientific evidence throughout. We also recognize that there are many other types of benefits wetlands can provide, including cultural, recreational, educational, and other factors related to quality of life (e.g., Pedersen et al. 2019), but these were beyond the scope of this synthesis.

Table 1. Multiple Benefits Examined in this Synthesis

Benefit Category	Description	Example
Water supply regulation	Improving and protecting water supply, such as by storing surface water, recharging groundwater aquifers, or regulating the release of water downstream	Montane meadow restoration increased summer stream baseflow volumes by 5-12 times over pre-restoration volumes, contributing to greater water supply to downstream communities, even in drought years (Hunt et al. 2018).
Flood risk mitigation	Protecting communities from flood and storm damage, such as by storing and slowly releasing floodwaters, reducing wave heights, or dissipating wave energy	Coastal wetlands, especially salt marshes, can reduce wave heights and provide shoreline protection during storm surges (Narayan et al. 2016).
Water quality	Removing pollutants from surface waters, such as by trapping sediments, removing nutrients, and detoxifying chemicals	Vegetated riparian buffers with widths of 11-15m can remove >=75% of nutrients and sediment in surface water runoff (Lind et al. 2019).
Soil health	Improving soil health through processes such as retention of sediments, nutrients, and soil moisture, erosion control, and carbon storage	Vegetated buffers improve soil quality in riparian zones compared to adjacent agricultural production areas characterized by increased carbon and nitrogen storage and increased soil moisture content (Rahe et al. 2015).
Biodiversity support	Providing essential habitat for a diverse array of plant and wildlife species	Intact, natural forested wetlands and recently restored emergent wetlands within the Grays River estuary provide off-channel rearing opportunities for migrant juvenile Coho salmon (Craig et al. 2014).
Economic value	Supporting local economies by providing benefits that have an associated monetary value in addition to their intrinsic value, such as by directly providing valuable resources or in terms of avoided costs	The capitalized value that an acre of salt marsh provides for recreational fishing is estimated as $6,471/acre (Barbier et al. 2011).
Carbon	Sequestering and storing more carbon than is released into the atmosphere	Of the studies we reviewed, net ecosystem exchange rates in tidal marsh ranged from 221 g C/m^2/yr (Taillardat et al. 2020) to 534 g C/m^2/yr (Lu et al. 2017)..

In this report, we synthesized the results of our rapid evidence assessment for a wide range of benefits associated with wetland conservation and restoration. Our synthesis emphasized data from North America and especially the United States, although many of the high priority meta-analyses and reviews we incorporated were global in scope, and we reference these data as well. Where data were available, we also discussed how benefits varied by wetland class (see next section), factors contributing to variation in the magnitude of each benefit, and evidence for the effectiveness of wetland restoration in recovering wetland benefits. We give special attention to evidence for the carbon benefits of wetlands, given the growing interest in the net impacts of wetlands on greenhouse gas emissions and global warming potential, and the breadth of the scientific literature. In addition, the role of beaver as an ecosystem engineer emerged as an important topic in the literature we reviewed, and we summarized their impacts on multiple benefits in a special topic section. Finally, we also assembled several case studies as examples of wetland conservation projects designed to protect and restore multiple benefits, which may serve as good models for future projects.

Wetland Classification

Developing a comprehensive understanding of evidence for wetland benefits is complicated by the diversity of wetlands and the many ways they may be classified in the scientific literature. For example, wetlands may be classified by any combination of hydrology (frequency of inundation, seasonality, salinity, or water source), vegetation type (herbaceous or woody), geography (region, latitude, or elevation), soil type (mineral or peat), and level of anthropogenic disturbance. Because definitions and delineations among wetland classes vary, we focused our synthesis on four major hydrological classes of wetlands, as defined below (see also Table 2). However, we also relied on authors' descriptions of the wetlands included in their studies, which did not always clearly fall into one of these classes. Thus, we reported data on additional wetland classes and subclasses where necessary.

Coastal wetlands are wetlands that occur at the interface between land and sea, especially in estuaries, lagoons, and along the leeward side of barrier islands, including **mudflats**, **salt marsh**, **mangroves**, and **seagrass meadows**. Their water is sourced primarily from the ocean, but also includes freshwater runoff from the land, and thus ranges from saline to brackish. They may also accumulate organic matter and sediment that is transported downstream from uplands or by wave action from ocean sources. Salt marshes, also called **estuarine emergent wetlands**, are dominated by herbaceous, salt-tolerant marsh plants, and are found in intertidal zones worldwide and on every coast of the United States. Mangroves, sometimes also called **estuarine woody wetlands**, are similarly found in the intertidal zone, but are instead dominated by salt-tolerant trees; in the United States they are limited to the coast of the Gulf of Mexico and the southern tip of Florida. Seagrass meadows range from intertidal to subtidal, and are formed by aquatic plants in shallow coastal areas, lagoons, and estuaries worldwide. Coastal wetlands are sensitive to coastal erosion and saltwater inundation from extreme events such as hurricanes, but also the long-term increases in water depth from sea level rise that can effectively drown wetland vegetation. Restoration of coastal wetlands often involves strategies to shelter shorelines from wave action, improve hydrological connection, trap sediment to build up new intertidal areas, and restore the upland transition zone to facilitate landward migration in response to sea level rise. Coastal wetlands are especially important to the global carbon cycle and the collective carbon stored in mudflats, salt marshes, mangroves, and sea grasses is often referred to as **blue carbon**.

Freshwater depressional wetlands are isolated freshwater wetlands formed primarily by precipitation, groundwater, or local runoff, and they are surrounded by uplands on all sides. Because they are hydrologically isolated, the organic matter and sediment they accumulate is entirely from the local basin. We include in this wetland class **prairie potholes**, **pocosins**, **vernal pools**, and **playa wetlands**. Prairie potholes are depressional wetlands found primarily in the upper midwest of the United States, that fill with snowmelt and precipitation, and are dominated by marsh vegetation. Pocosins are found along the Atlantic coast, and are formed by poorly-drained basins that are seasonally saturated; they often feature deep peat soils and are dominated by shrubby vegetation. Vernal pools and playa wetlands are ephemeral depressional wetlands found on the Pacific coast, southern high plains, and in the northeast, characterized by bedrock or clay substrate that collect seasonal precipitation; they are shallowly flooded during part of the year and may be completely dry at other times. Freshwater depressional wetlands are sensitive to disturbances including burning, drainage, and invasive species, and because they are among the lowest points in their watersheds, they can be at risk for sediment and pollution accumulation. Restoration of freshwater depressional wetlands often includes strategies to minimize disturbance, reduce nutrient load and sedimentation from surrounding land uses, and plug drains.

Riverine wetlands are wetlands that depend on the flow of water through river or stream channels, and include wetlands that form on the banks, terraces, and adjacent floodplains. Their water is primarily from overbank flooding or subsurface (underground) water flows from the stream channel, but they also intercept runoff from adjacent uplands. Similarly, they may accumulate deposits of organic matter and sediment from overbank flooding or runoff. Notably, riverine wetlands may also flood regularly with the tides, especially in the farthest downstream reaches near estuaries, and thus may receive organic matter and sediment from downstream. Riverine wetlands may also be called **riparian** or **floodplain wetlands**, and are variable in appearance and vegetation structure. For example, floodplain wetlands may appear like depressional wetlands except for the riverine source of their water supply, while riparian wetlands often include a diversity of tree and shrub species that form a corridor along the stream channel and may also extend into the floodplain. Many riverine wetlands have been heavily altered by the construction of dams and levees that limit the frequency and extent of overbank flows and contribute to reduced subsurface flows. Floodplains have also frequently been targets of conversion to agriculture because of their rich soils. Restoration of riverine wetlands typically includes restoration of hydrological connectivity and may include planting woody vegetation. An emerging strategy for some riverine wetlands is the installation of beaver dam analogs, or the restoration of beaver themselves; beaver have been extirpated from many areas and are ecosystem engineers that can have a dramatic impact on hydrology and the multiple benefits of riverine wetlands.

Montane meadows may be riverine or depressional wetlands, but they are distinctively higher elevation wetlands primarily fed by snowmelt. Also called **wet meadows**, they typically have more pronounced seasonal dynamics that affect rates of carbon accumulation and other benefits, including a short growing season, low temperatures, and snow cover for long periods of the year. As with other freshwater depressional wetlands and riverine wetlands described above, they are highly sensitive to disturbance and changes in hydrology, resulting in incision of stream channels, reduced connection to the floodplain, increased discharge of water from meadows to streams, and lowered water tables. Restoration of riverine montane meadows often focuses on elevating the water table and restoring connectivity between the stream and surrounding meadow floodplain, including by installation of beaver dam analogs or restoration of beaver themselves, as described above.

In addition to the four major wetland classes we examined, we also highlight a cross-cutting subclass of **peatlands**, which may be defined as wetlands that have soils with at least 30% (dry mass) of dead organic material (Joosten and Clarke 2002). Also called **bogs** or **fens**, depending on whether or not the water source is primarily precipitation, respectively, peatlands can form wherever consistently saturated soils prevent the full decomposition of organic material, which then quickly accumulates. Thus, peatlands may be found in any climate, associated with either fresh or saline water, and any of the four major wetland classes described above can also be a peatland. We highlight peatlands because their unique soil properties have a direct influence on their carbon benefits, and because some studies did not identify whether the peatlands they examined were coastal, freshwater, riverine, or montane meadows. Like any wetland, peatlands are sensitive to disturbance and alterations in hydrology, and due to their soil qualities, peatlands may be particularly targeted for conversion to agriculture or excavated for fuel. Peatland restoration typically focuses on restoring site hydrology which, in turn, protects stored carbon.

Finally, in the context of the highly altered nature of many of the landscapes in which wetlands occur, we also consider the evidence for carbon and co-benefits of **managed wetlands**, sometimes also called **constructed wetlands**. When possible, we compare the benefits of managed wetlands to natural ("reference") wetlands of similar type. In many regions, loss and conversion of natural wetlands has been partly mitigated by the construction of highly-managed wetlands with infrastructure to control the conveyance of water to and from each wetland. These wetlands are managed to meet specific objectives, such as improving water quality or providing habitat for wildlife, and thus the management of vegetation and the timing and depth of applied water are optimized with these goals in mind. In addition, they may or may not be constructed in areas that historically were wetlands, and may or may not function similarly to natural wetlands. Thus, the extent of carbon and co-benefits obtained from managed wetlands depends heavily on the management goals.

We do not consider here other wetlands included in some classification schemes, such as urban and industrial ponds, salt ponds, aquaculture ponds, or deep permanent waters such as rivers, lakes, and reservoirs.

Table 2. Primary wetland classification used in this synthesis.

Wetland Class	Description	Hydrological Features
Coastal wetlands	Wetlands that occur at the interface between land and sea, especially along shorelines protected from wave action such as in estuaries, lagoons, and along the leeward side of barrier islands, including **mudflats**, **salt marsh**, **mangroves**, and **seagrass meadows**.	Water is sourced primarily from the ocean, and organic matter and sediment is accumulated through wave action. However, freshwater runoff may also contribute to the supply of water, organic matter, and sediment, and the water may range from saline to brackish.
Freshwater depressional wetlands	Hydrologically isolated wetlands that are surrounded by uplands on all sides, including **prairie potholes**, **pocosins**, and ephemeral **vernal pools** and **playa wetlands**.	Water is sourced primarily by precipitation, groundwater, or local runoff. Organic matter and sediment are accumulated entirely from the local basin.
Riverine wetlands	Also called **riparian** or **floodplain wetlands**, these are wetlands that depend on the flow of water through river or stream channels, and include wetlands that form on the banks, terraces, and adjacent floodplains.	Water is sourced from overbank flooding or subsurface flows from the stream channel. Organic matter and sediment accumulates from overbank flooding or runoff from adjacent uplands.
Montane meadows	Also called **wet meadows**, these are riverine or depressional wetlands at higher elevations, typically with pronounced seasonal dynamics, including a short growing season, low temperatures, and snow cover for long periods of the year.	Water is sourced primarily from snowmelt. Sources of organic matter and sediment depend on whether the meadow is riverine or depressional.
Peatlands	A cross-cutting subclass including any wetlands with soils that have at least 30% (dry mass) of dead organic material. Peatlands can be found wherever consistent flooding prevents the decomposition of organic matter, including in any climate.	Water may be saline or fresh, and sources of water, organic matter, and sediment depend on which of the other four classes the peatland falls into. They may also be called **bogs** where water is sourced primarily from precipitation, or otherwise **fens**.
Managed wetlands	Wetlands that are actively managed to meet specific objectives, that may not function similarly to natural wetlands and may not be constructed in areas that historically were wetlands.	Water may be from any source, and is typically conveyed through infrastructure that controls the timing and extent of flooding.

Summary of Evidence for Multiple Benefits

Across all wetland classes in this synthesis, we found evidence for a broad array of benefits (see examples in Table 3). Overall, some key takeaways can be summarized from our our rapid evidence assessment:

- **Coastal wetlands** provide significant flood risk mitigation benefits in the form of storm surge protection, as well as water quality and biodiversity support benefits. Each of these benefits represent significant economic value from avoided damages to property and infrastructure, avoided cost of wastewater treatment, and providing habitat to species with commercial and recreational value. In addition, coastal wetlands are typically net carbon sinks with relatively low methane emissions and relatively high soil carbon storage; mangroves also store substantial amounts of carbon in aboveground biomass.

- **Freshwater depressional wetlands** provide important habitat for diverse and unique plants and animals often specialized for ephemeral water availability or conditions. These wetlands also provide benefits to soil health, water quality, and water supply regulation, in the form of surface water storage, and their restoration is expected to provide substantial economic value. Carbon storage and flux estimates in freshwater wetlands are highly variable and difficult to generalize, and their net carbon benefit is highly dependent on the rate of methane emissions, but many are likely old enough to be providing net carbon benefits.

- **Riverine wetlands** are biodiversity hotspots that can provide significant water supply regulation, flood protection, and water quality benefits provided their hydrological connectivity to adjacent rivers and streams is maintained. Vegetated riparian buffers can also contribute substantially to water quality and biodiversity benefits, provided they are of sufficient width. The economic value associated with flood protection benefits in particular may be substantial. As with other freshwater wetlands, carbon storage and flux estimates are highly variable in riverine wetlands. Riparian wetlands have the potential for substantial amounts of additional carbon stored in aboveground biomass, but we found few estimates of methane emissions.

- **Montane meadows** are also hotspots of biodiversity that can provide an array of benefits, especially by storing and slowly releasing surface water that improves the resilience of water supplies downstream. The capacity of montane meadows to provide these benefits depends on the degree of disturbance, degradation, and loss of hydrological function, but restoration and enhancement of hydrological function can improve these benefits. Carbon storage and fluxes appear to be less well-documented for montane meadows than other wetland classes, and fluxes are likely to be highly seasonal due to the short growing season and extended periods of snow cover.

- **Peatlands** are a cross-cutting wetland subclass notable for their very high amounts of carbon storage, which also makes them highly vulnerable to becoming large sources of carbon emissions if their soils are disturbed. Paired with carbon sequestration rates that appear to be relatively slow, any carbon emitted as a result of disturbance in peatlands will take a very long time to recover.

Table 3. Examples of evidence found by wetland class and benefit category. Also shown are the number of studies we synthesized for each (total n = 159), and a subset of the relevant references providing evidence for each benefit category. Note that some papers were relevant across multiple benefit categories and multiple wetland classes.

Benefit Category	**Wetland Class**			
	Coastal wetlands mudflats, salt marsh, mangrove, seagrass meadows (n = 15)	**Freshwater depressional** prairie potholes, pocosins, vernal pools, playas (n = 8)	**Riverine wetlands** floodplain, riparian (n = 46)	**Montane meadows** (n = 13)
Water supply regulation (n = 20)	See flood risk mitigation	Water storage in playas (Daniel et al. 2015), vernal pools (Duffy and Kahara 2011), and Prairie Potholes (Gleason et al. 2011); Role of clay subsoil thickness (Doherty et al. 2014)	Capacity to store flood water (Kousky and Walls 2014; Watson et al. 2016); Effects of levee setback (Singh et al. 2018); Role of beaver & beaver dam analogs (Weber et al. 2017; Larsen et al. 2021)	Role in baseflow, groundwater storage, and drought resilience (Hammersmark et al. 2008; Ramstead et al. 2012; Hunt et al. 2018); Efficacy of low-tech restoration techniques (Silverman et al. 2019); Impacts of livestock grazing (Vernon et al. 2022).
Flood risk mitigation (n = 22)	Storm surge protection (Shepard et al. 2011; Barbier 2013; Salgado and Martinez 2017; Van Coppenolle et al. 2018); Relative effectiveness of mangroves, salt marshes, coral reefs, and seagrass (Narayan et al. 2016)	Reducing the frequency and magnitude of floods (Kadykalo and Findlay 2016); Stormwater retention (Doherty et al. 2014); flood storage capacity (Duffy and Kahara 2011)	Reducing the frequency and magnitude of floods and flood damage (McAlpin et al. 2013; Kadykalo and Findlay 2016; Watson et al. 2016); Impacts of levees (Kousky and Walls 2014); Role of riparian forests (Rood et al. 2015); Role of beaver (Larsen et al 2021)	Effectiveness of restoration in restoring flood mitigation benefits (Hammersmark et al. 2008, Ramstead et al. 2012)
Water quality (n = 19)	Nutrient removal (Russell and Greening 2015); Removal of protozoal pathogens (Daniels et al. 2014)	National nitrogen removal capacity (Cheng et al. 2020); Influence of spatial configuration (Qiu and Turner 2015); Impacts of wet-dry cycles and vegetation on nutrient removal (Doherty et al. 2014)	Impacts of flow and vegetation on nutrient removal (Mitsch et al. 2012, 2014; Gordon et al. 2020); Benefits of vegetated riparian buffers (Hancock et al. 2019; Stutter et al. 2019; Valkama et al. 2019)	Benefits of beaver presence to water quality (Wegener et al. 2017) and role of beaver dam analogs in retaining sediment (Scamardo and Wohl 2020); Impacts of livestock grazing (Vernon et al. 2022)
Soil health (n = 9)	N/A	Effectiveness of restoration (Marton et al. 2015)	Benefits of vegetated riparian buffers (Rahe et al. 2015); Influence of vegetation community (Buchanan et al. 2020)	Impacts of livestock grazing on soil nutrients and cycling (Ramstead et al. 2012; Vernon et al. 2022)

	Wetland Class			
Benefit Category	**Coastal wetlands** mudflats, salt marsh, mangrove, seagrass meadows (n = 15)	**Freshwater depressional** prairie potholes, pocosins, vernal pools, playas (n = 8)	**Riverine wetlands** floodplain, riparian (n = 46)	**Montane meadows** (n = 13)
Biodiversity support (n = 34)	Habitat for coho salmon in tidal estuaries (Craig et al. 2014), birds in tidal marsh (Veloz et al. 2013), and shrimp and blue crab in Gulf Coast salt marsh (Barbier et al. 2011; Minello et al. 2012)	Critical habitat for birds (Gleason et al. 2011; Behney 2021); Restoration benefits insect pollinators (Begosh et al. 2020)	Habitat for birds, small mammals, insects and herpetofauna (Golet et al. 2008; Banville and Bateman 2012); Benefits to fish (Cross et al. 2013); Benefits of riparian buffers (Guzy et al. 2019); Role of beaver (Weber et al. 2017; Fairfax and Whittle 2020)	Habitat for fish (Moyle et al. 2008), birds (Campos et al. 2020; Loffland et al. 2022), and mammals and amphibians (Ramstead et al. 2012); Role of beaver and beaver dam analogs (Silverman et al. 2019)
Economic value (n = 14)	Avoided cost of water treatment (Barbier 2013; Russell and Greening 2015) and damage from floods and storm surges (Martinez et al. 2011; Barbier 2013; Menéndez et al. 2020); Shrimp production (Minello et al. 2012); Fisheries, recreation, and tourism in mangroves (Salem and Mercer 2012)	Value of carbon sequestration, soil health, and waterfowl production in Prairie Potholes (Gascoigne et al. 2011)	Avoided costs of flood damage (Kousky and Walls 2014; Watson et al. 2016); Recreational and commercial value of fish production (Ogston et al. 2015; Trebitz and Hoffman 2015)	N/A
Carbon (total: n = 79; across wetland classes: n = 21)	**Coastal wetlands** (n = 22)	**Freshwater** (n = 11)	**Riverine wetlands** (n = 6)	**Montane meadows** (n = 12)
	Soil carbon storage (Bridgham et al. 2006; Hinson et al. 2017; Alongi 2020); Aboveground carbon storage in mangroves (Villa and Bernal 2018; Kauffman et al. 2020a, 2020b); Sequestration (Mcleod et al. 2011; Lu et al. 2017); Methane emissions (Poffenbarger et al. 2011; Kroeger et al. 2017); see cross-wetland comparisons in Nahlik and Fennessy (2016) and Taillardat et al. (2020)	Soil carbon storage (Bridgham et al. 2006; Mazurczyk and Brooks 2018; Tangen and Bansal 2020); Aboveground carbon storage (Buffam et al. 2011); Carbon sequestration (Villa and Bernal 2018; Loder and Finkelstein 2020; Tangen and Bansal 2020); Methane emissions (Treat et al. 2018; Knox et al. 2019); Role of beaver (Mazurczyk and Brooks 2018)	Riparian soil and biomass carbon storage (Matzek et al. 2015, 2020; Sutfin et al. 2016; Dybala et al. 2019a, 2019b); Floodplain soil carbon sequestration (Mitsch et al. 2014; Loder and Finkelstein 2020); Methane emissions (Tan et al. 2020; and cross-wetland comparison Rosentreter et al. 2021)	Soil carbon storage (Norton et al. 2014; Reed et al. 2021); Soil carbon sequestration (Arnold et al. 2014; Tucker et al. 2014; Knowles et al. 2016; Blackburn et al. 2021; Reed et al. 2021); Methane emissions (Blankinship and Hart 2014; Reed et al. 2021); Role of beaver (Johnston 2014)

Wetland Conservation Recommendations

Based on our rapid evidence assessment, in this section we recommend priority actions to ensure wetland conservation policies and strategies are most effective in protecting and restoring the multiple benefits provided by wetlands.

Protect the multiple benefits of all existing wetlands

To maximize all wetland benefits, protection of all existing wetlands is the highest priority. Conserving wetlands of any class will preserve a broad array of benefits essential to human communities (Table 3), many of which also have substantial economic value in terms of directly providing valuable resources (e.g., fisheries) and in terms of avoided costs (e.g., flood damage and water treatment). In addition, wetlands across all classes store large amounts of carbon in the soil, and in the case of mangroves and riparian wetlands, in aboveground biomass as well. When wetlands are disturbed, the rich carbon stores in the soil immediately begin to be released, and it can take a very long time (ranging decades to millennia) for restored wetlands to accrue the carbon benefits of natural wetlands (Taillardat et al. 2020; Figure 7.9). Likewise, many other benefits are also reduced when wetlands are disturbed or degraded, requiring time to recover, and potentially incurring a high economic cost (e.g., Salem and Mercer 2012; Narayan et al. 2016; Watson et al. 2016). No wetland class clearly provided more benefits than the others (Table 3), and there was substantial overlap and variation in the carbon storage estimates (Figures 7.2–7.3), suggesting that no wetland class clearly stores more carbon than the others. The exception is peatlands, the cross-cutting subclass of wetlands within each wetland class, which by definition store very high amounts of carbon in their soils. Therefore, to protect existing carbon stocks from disturbance and release into the atmosphere, and to preserve the numerous other valuable benefits wetlands provide, wetlands of all classes should be protected, with particular attention paid to protecting wetlands with peat soils.

We emphasize that protecting existing wetlands requires not only protection from conversion to other land uses, but increasingly requires consideration of climate change, which is an important contributor to recent losses of coastal wetlands (Dahl 2011). Sea level rise will shift the intertidal zone where salt marshes and mangroves occur, and although some coastal wetlands may be resilient to sea level rise (Wang et al. 2019; Herbert et al. 2021), others may have insufficient sediment accumulation and/or insufficient upland space suitable for wetlands, resulting in increasing damage by erosion or drowning, and a net carbon loss (Herbert et al. 2021). Similarly, freshwater wetlands that rely on river flows, precipitation, and groundwater will be impacted by increasing frequency of drought and growing demands on freshwater resources. Long-term monitoring through variable climatic conditions would help identify the impacts of this hydrological variability on wetland carbon and co-benefits, including the potential for wetlands to store and gradually release water downstream through drought periods (Barksdale et al. 2014).

Restore wetlands to ensure long-term benefits

Given sufficient time, restored wetlands of any class can improve, if not fully recover, numerous benefits. While some benefits may begin to accrue immediately, such as colonization by diverse wildlife taxa including insects, bats, rodents, and birds (Golet et al. 2008), it can take decades or more to realize the full value of wetland restoration efforts. In terms of climate change mitigation benefits, restored wetlands of any class will also eventually provide a net cooling effect, but in the short-term there was a clear difference between coastal wetlands and all types of freshwater wetlands due to differences in the rates of methane emissions (Figure 7.8). Methane emissions are reduced in coastal wetlands, such that restoration of coastal wetlands can provide net cooling benefits after a much shorter period of time compared to restoration of freshwater wetlands, which may take decades to millennia to produce a net cooling effect (Figure 7.5). Therefore, to maximize immediate climate change mitigation benefits, protection of existing wetlands should be prioritized as described above, followed by restoration of coastal wetlands. However, to maximize long-term benefits and address long-term global declines in wetland extent, wetlands of all classes should be restored. Wetland restoration is particularly important given tremendous wetland losses over the past 100-200 years; namely, 33-87% of global historical wetlands (Davidson 2014; Hu et al. 2017) and up to 53-62% loss of U.S. wetlands have been lost, degraded, or converted to other land uses (Dahl and Johnson 1991; Bridgham et al. 2006). In addition, it is important to recognize that wetland restoration alone will not be sufficient to address climate change. Large-scale wetland conservation and restoration efforts should be paired with policies to reduce fossil fuel emissions and advance other nature-based climate solutions.

The effectiveness of restoration depends heavily on the site-specific context of the wetland, and strategically selecting restoration locations can help maximize the target benefits. For example, wetland restoration in agricultural watersheds with high nutrient loads can increase the magnitude of the water quality benefits (Cheng et al. 2020). In addition, there are several strategies for improving the recovery rates in restored wetlands: recovery rates may be improved in large wetland areas (>100 ha), in warmer climates, and with restored hydrological connectivity, planting effort, and effective management (Gleason et al. 2011; Moreno-Mateos et al. 2012; Yu et al. 2017; Dybala et al. 2019a; Su et al. 2021). Process-based restoration approaches designed to restore natural hydrological function can be highly effective, including relatively low-tech approaches such as installing beaver dam analogs, and a network of dams can substantially increase water storage capacity and wetted area (Weber et al. 2017; Karran et al. 2018; Larsen et al. 2021). In addition, the installation of vegetated buffer zones can increase biodiversity support and water quality benefits, especially when they are of sufficient width (Stutter et al. 2019).

As with natural wetlands, restored wetlands will also have to contend with the impacts of climate change, sea level rise, and increasing frequency of drought, and we recommend designing **climate-smart restoration** projects that are explicitly designed to be resilient to a range of projected future conditions and help mitigate the impacts of climate change (Simonson et al. 2021; Dunwiddie et al. 2009). For example, riparian restoration projects that incorporate a greater diversity of trees, including heat- and drought-tolerant species and varieties may be more resilient to more extreme climate conditions, while also helping to mitigate the impacts of heat and drought on wildlife by continuing to provide shade and resources (Seavy et al. 2009; Perry et al. 2015). Similarly, restored coastal wetlands, and particularly salt marshes, can provide protection to coastal ecosystems and properties from rising sea levels and storm events, if their location is carefully selected to ensure sufficient upland space for them to migrate and sufficient rates of sediment accumulation so they can build elevation and adapt to changing sea levels (Martinez et al. 2011; Stralberg et al. 2011; Narayan et al. 2016; Salgado and Martinez 2017; Van Coppenolle and Temmerman 2019).

Minimize methane emissions in restored wetlands

Restoration across all wetland classes is an essential strategy to recover the benefits lost from long-term global declines in wetland extent, but it is also important to recognize that freshwater wetlands emit significant amounts of methane (Rosentreter et al. 2021; Figure 7.8). Thus, newly restored wetlands represent a short-term trade-off in methane emissions for long-term gains in multiple benefits. However, managing the hydrology and nutrient inputs of wetlands can help minimize methane emissions. For example, methane emissions can be suppressed by restoring tidal flows and salinity in coastal wetlands, where appropriate, such as by converting rice paddies and aquaculture ponds to tidal salt marsh or mangroves (Kroeger et al. 2017; Rosentreter et al. 2021). Another strategy is to reduce the input of nutrients to freshwater wetlands, and to lakes, reservoirs, and rivers upstream, which contribute to both eutrophication and methane emissions in wetlands and estuaries downstream (Rosentreter et al. 2021); the addition of nitrogen to freshwater wetlands significantly increased greenhouse gas emissions (Chen et al. 2020). Installing vegetated riparian buffer strips is a proven strategy for intercepting run-off and capturing nutrients before they enter waterways, and further research into strategies for improving the efficacy of riparian buffer strips has been called for (Stutter et al. 2019). In addition, establishing more consistent and extensive monitoring of the extent of eutrophication in coastal wetlands would help identify where nutrient management is most needed (Oelsner and Stets 2019). Finally, there is some evidence that the presence of fish in wetlands can reduce methane production (Devlin et al. 2015) due to fish feeding on the zooplankton that would normally limit the population of methanotrophic bacteria that feed on dissolved methane.

Identify and address trade-offs among benefits

While we found widespread evidence that wetlands of all classes provide multiple benefits, it is important to recognize that wetland restoration and management strategies that target one benefit are not guaranteed to provide other benefits effectively. For example, there was evidence for a trade-off between plant biomass and several other metrics in experimental depressional wetlands, where a wetland dominated by cattail (*Typha* spp.) had very high net primary productivity but also less plant species richness, erosion resistance, and water quality benefits compared to other wetlands (Doherty et al. 2014), while another study found a trade-off between plant diversity and water quality benefits (Jessop et al. 2015). Similarly, an analysis of riparian forest restoration sites found that while carbon storage and bird abundances were both higher in restoration sites compared to an adjacent unrestored site, there was evidence for a trade-off between biodiversity and carbon storage at the highest forest stand densities; while managing for a high density of trees increased the amount of carbon stored in the trees, these areas were also associated with the lowest bird densities and diversities (Dybala et al. 2019b). Therefore, a focus on maximizing one benefit may come at the cost of other benefits.

We recommend adopting **Multiple-Benefit Conservation** approaches, defined as *efforts designed to simultaneously benefit local communities of people, enhance ecological function, and improve habitat quality for fish and wildlife* (Gardali et al. 2021). These efforts would explicitly define multiple goals at the outset of the project, inclusive of a wide range of community values, to allow identification of trade-offs among benefits and adjusting the design of restoration and management plans to achieve multiple goals simultaneously. While the economic value of these benefits can be a consideration (Brander et al. 2013), it is not required and indeed may not be appropriate (Gardali et al. 2021). It is also important to note that while trade-offs may occur within the footprint of a wetland itself, broader watershed-scale evaluations are also warranted to consider trade-offs that may occur elsewhere within the system (Stutter et al. 2019). Similarly, land use and management decisions elsewhere in the system, such as development or drainage, may incur trade-offs within wetlands (Barksdale et al. 2014). To improve the outcomes of wetland conservation policy and practices, we recommend supporting interdisciplinary research to support Multiple-Benefit Conservation approaches, including the relationships between benefits, how they are influenced by restoration design and management practices, and how they interact across spatial and temporal scales.

Science Synthesis Results

Photo credit: Jak Wonderly Photography

1. Water Supply Regulation

Across wetland classes, many of the important benefits provided by wetlands are related to hydrological processes. Because wetland soils and dense wetland vegetation can absorb water during wet periods and release water downstream during dry periods, wetlands are often described as “acting like sponges.” Wetlands of all classes can reduce the impact of floodwaters and surface water runoff, recharge and discharge groundwater, store surface water and regulate downstream flows. These hydrological dynamics are necessarily influenced by wetland water regimes (i.e., the timing and duration of inundation) and connectivity between wetland basins and surface-groundwater interactions, but may also be influenced by the composition and configuration of land covers in a watershed (Qiu and Turner 2015). Wetland hydrology, in turn, can affect other wetland benefits including water quality, flood risk mitigation, and biodiversity support (Euliss et al. 2004; Adame et al. 2019), discussed further below.

Coastal Wetlands

The majority of the studies in our evidence assessment around regulation of hydrological processes in coastal wetlands was centered on flood risk mitigation, a benefit for which there has been considerable effort to evaluate. Therefore, this topic is covered separately (see Chapter 2).

Freshwater Depressional Wetlands

Freshwater depressional wetlands provide surface water storage that can result in other benefits such as flood risk mitigation and groundwater recharge, and conservation and restoration programs can improve benefits related to water supply regulation (Gleason et al. 2011). For example, wetland playas enrolled in NRCS Wetland Reserve Programs (NRCS 2010) had more than five times the water volume of cropland playas, and were comparable to reference playas, suggesting these programs help restore baseline hydrologic conditions in playa wetlands (Daniel et al. 2015). Similarly, water storage capacity in vernal pools and other seasonal wetlands in California’s Central Valley increased in the last decade when these wetlands were enrolled in USDA conservation programs (Duffy and Kahara 2011). However, water storage capacity and the speed with which ponded water drains depends on the type and thickness of subsoils, which in turn, can influence other wetland benefits. While in other ecosystems (e.g. grasslands) there is evidence that managing for net primary productivity may be correlated with additional benefits like plant species richness, the co-occurence of these benefits in wetlands may follow a different pattern. An investigation of multiple ecosystem services in three experimental wetlands demonstrated that the fast-draining wetlands (with the thinnest clay subsoil) were linked to the broadest suite of ecosystem services overall, including flow attenuation, stormwater retention, diversity support, erosion resistance, and water quality improvement, but not net primary productivity (Doherty et al. 2014). This apparent trade-off aligned with other studies finding that in wetland systems, managing for net primary productivity may mean sacrificing other benefits (Doherty et al. 2014; Jessop et al. 2015). Thus, important water supply regulation and other wetland co-benefits may not be well preserved by managing for net primary productivity alone, suggesting that wetland managers should clearly define their goals when developing management and restoration plans.

Riverine Wetlands

Riverine wetlands can provide multiple critical benefits related to water supply regulation, including surface water storage, flood risk mitigation, groundwater recharge. These benefits are affected by changes to the hydrological connectivity between rivers and their floodplains, such as the presence of beaver dams that promote floodplain connectivity (see Chapter 8) or the installation, size, and placement of levees and other barriers. For example, during restoration of the Yakima River floodplain (Yakima Basin, WA), an area that has experienced more than 50 floods since 1894, levees were setback (moved away from the river) ~200m and ~300m from old levee locations on either side of the river (Singh et al. 2018). Levee setback is a restoration technique intended to improve the floodplain ecosystem, including surface-groundwater connection. In a groundwater model evaluating the effects of levee setback restoration on floodplain hydrological services, the authors found that setbacks improved flow through the hyporheic zone (the sediments underlying a stream channel where surface water and groundwater interact)k (Singh et al. 2018). Thus, levee setbacks increased the interaction between surface and groundwater, providing a greater total area for microbial activity and nutrient retention. The authors concluded improved hyporheic flow and floodplain reconnection should also support more opportunities for enhanced biogeochemical processing, improved water quality, and increased habitat value (Singh et al. 2018).

Montane Meadows

Montane meadows provide a wide range of water supply regulation benefits, including flood attenuation, groundwater storage, and extended dry season base flows (Ramstead et al. 2012; Hunt et al. 2018; Vernon et al. 2022), which in turn create additional benefits (described further below). Conversely, meadows subject to human disturbance, overgrazing, and erosion often have severely incised channels, reducing the hydrological connectivity between streams and their floodplains, degrading meadow function and threatening these benefits (Campos et al. 2020). Restoration of meadow hydrology can restore essential meadow functions and resilience, including increased groundwater recharge (Hammersmark et al. 2008; Ramstead et al. 2012; Hunt et al. 2018) and increased drought resilience in California's Sierra Nevada (Hunt et al. 2018). Evidence for changes in summer outflows following restoration was mixed. Hunt et al. (2018) found an increase of 5% more than inflow to 35-95% more than inflow, even during drought conditions. Nash et al. (2018) did not detect a change, but concluded that late summer flows could still be maximized by promoting conditions to encourage lateral water flows.

While engineered approaches to restoring incised meadows can be expensive and beyond the scope of some landowners, low-tech restoration practices have also been shown to be effective (and cost-effective) in restoring meadow hydrologic connectivity. Examples of restoration of hydrologic connectivity include: grazing management to reduce further disturbance, the installation of Zeedyk structures (primarily hand-built simple rock and wood structures; Maestas et al. 2018), or the construction of beaver dam analogs (see Chapter 8) to slow and disperse the flow of water on to the surrounding meadow (Silverman et al. 2019). Specifically, the installation of one thousand Zeedyk structures in the Upper Gunnison Basin of Colorado increased productivity by 24%, attributed to a raising of the water table. In another example, ten years after restoring a meadow that had been heavily grazed, vegetation productivity no longer fluctuated with precipitation, suggesting that the meadow had developed greater resilience to drought and climate variability (Silverman et al. 2019). While restoring hydrologic connectivity can be good for grazing, livestock grazing can be a frequent contributor to meadow degradation. The majority of 47 studies in the Sierra Nevada found negative, neutral, or mixed negative and neutral associations between livestock grazing and meadow ecosystem function (Vernon et al. 2022), and the combined action of grazing removal with additional restoration in meadows may increase the potential to restore meadow function and benefits (Ramstead et al. 2012; Vernon et al. 2022).

2. Flood Risk Mitigation

Due to its importance for protecting human lives, economic interests and infrastructure, the flood risk mitigation capacity of wetlands is well-represented in the literature. Floods are one of the most damaging natural forces, contributing to loss of life and property and damage to infrastructure (Kousky and Walls 2014; Narayan et al. 2016; Salgado and Martinez 2017). By holding floodwaters and slowing the rate and energy of water flow, wetlands of all classes can provide substantial flood protection benefits.

Coastal Wetlands

Coastal wetlands have the capacity to attenuate wave energy and dissipate the impacts of flood waters associated with rising seas and storm surges, especially during flood events (Van Coppenolle et al. 2018), preserving significant economic value to human infrastructure (Salgado and Martinez 2017). Coastal wetlands, including mangroves, salt marshes, coral reefs and seagrass/kelp beds, can reduce wave heights by 35% to 71%, with Narayan et al. (2016) estimating coral reefs and salt marshes have the highest potential to mitigate risk, and mangroves and seagrass beds about half as effective. However, in a global analysis examining the natural defense capabilities of mangroves, Menéndez et al. (2020) found significant coastal community protection benefits from mangroves, with North America projected to see some of the highest levels of those benefits in the world. Many other studies have also found that salt marshes have significant potential to attenuate wave energy and protect against storm surges (Martinez et al. 2011; Shepard et al. 2011; Barbier 2013; Salgado and Martinez 2017; Van Coppenolle et al. 2018).

Martinez et al. (2011) concluded that there was little data to support the widespread anecdotal evidence that coastal ecosystems provide protection against natural disasters, but that conclusion was rarely replicated in our evidence review. For example, in evaluating the protection value of coastal marshes in Southeast Louisiana (United States), Barbier (2013) estimated that the height of damaging storm surges can be reduced by 1 meter for each 9.4 to 12.6 km of additional coastal wetland. There is also evidence that protecting and restoring vegetated coastal wetlands can improve the resilience of coastal communities and ecosystems to impacts of sea level rise, storm surge and other climate change related hazards, even if hard infrastructure may also be necessary (Salgado and Martinez 2017). Factors influencing the effectiveness of the flood abatement benefits provided by salt marshes included the ratio of vegetation height to water depth (Narayan et al. 2016), as well as vegetation density, vegetation stiffness, and marsh width (Shepard et al. 2011). Similarly, the factors most frequently associated with stabilizing shorelines against erosion and wave action were vegetation species identity, height, and biomass production (Shepard et al. 2011).

Freshwater and Riverine Wetlands

Freshwater and riverine wetlands can decrease the frequency and magnitude of floods by storing water and slowing peak flow events (Kadykalo and Findlay 2016). However, the magnitude of these benefits is site- and context-specific, and difficult to generalize. As described in the previous section on water supply regulation, the capacity of freshwater depressional wetlands to store surface water and the speed with which ponded water drains depends on the type and thickness of subsoils. For riverine wetlands, their capacity to provide flood attenuation benefits is associated with preservation of natural geomorphology, sediment dynamics, biological structure, and hydrological connectivity (McAlpin et al. 2013; Kousky and Walls 2014; Watson et al. 2016). For example, levee construction and channelization of waterways should be avoided to allow floodplains to store water during flood events (Kousky and Walls 2014), although isolated wetlands within leveed systems still have flood protection value (Theiling and Burant 2013). Maintaining and restoring riparian forests can help prevent erosion, maintain hydrological dynamics, and improve flood resilience; tree cover may provide more resistance than grasslands to flood-associated bank erosion in riparian zones, especially for big flows (Rood et al. 2015).

Montane Meadows

Montane meadows provide flood attenuation benefits, and as described in the previous section on water supply regulation, these benefits can be threatened by disturbances that reduce the hydrological connectivity between streams and their floodplains. However, restoration of degraded meadow hydrology can restore these benefits. A review of the efficacy of montane meadow restoration from studies across the Southwestern United States concluded that there was substantial evidence that pond and plug methods, which involve excavation and construction aimed at restoring floodplain activity by intermittently damming incised channels, were successful at allowing meadows to regain their flood resilience benefits (Ramstead et al. 2012). Similarly, using a hydrological model of a 230-ha mountain meadow along a 3.6 km restored reach in California, one study found increased frequency and duration of floodplain inundation and decreased magnitude of flood peak flows (Hammersmark et al. 2008) which can be related to flood events.

3. Water Quality

Pollution of surface and groundwater from nutrients, sediments, pesticides, pathogens and other pollutants is one of the most pressing issues on the planet, leading to both issues of water supply reliability and prevalence of waterborne disease (UNESCO 2021). Wetlands have a well-demonstrated capacity to improve water quality through pollution retention and removal, including in agricultural landscapes where nutrient and pesticide deposition into wetland systems can be high (Verhoeven et al. 2006; Duffy and Kahara 2011; Karpuzcu et al. 2013; Oelsner and Stets 2019; Cheng et al. 2020; Gordon et al. 2020).

One of the major ways wetlands provide water quality benefits is through reducing nutrients in waterways. In the absence of effective nutrient reduction, eutrophication leads to harmful algal blooms and hypoxic regions known as "dead zones," where a cascade of events initiated by nutrient loading leads to die-offs of fish and other aquatic life. Common nutrients are nitrogen (primarily as nitrate, NO_3^-), and phosphorus (primarily as particulate phosphorus and soluble reactive phosphorus [$H_2PO_4^-$, HPO_4^{2-}]). Nitrogen and phosphorus flowing into a wetland as runoff from adjacent uplands or from upstream in a waterway can be taken up by wetland plants and cycled through the system through decomposition and nutrient cycling processes. Phosphorus is removed primarily by deposition and through sorption and accumulation in sediments. Nitrates entering wetland anaerobic soil zones can also be permanently removed through a microbial-mediated process of denitrification, a process that likely occurs at relatively higher rates in smaller, shallow wetlands with larger amounts of dissolved organic carbon, especially when oxygen content is low (Hansen et al. 2021). Denitrification also inevitably produces some N_2O (Firestone 1982), a potent greenhouse gas, suggesting a potential trade-off between water quality and climate change mitigation benefits, particularly under conditions of excess nutrient loading, low pH, and/or low soil moisture that amplify the production of N_2O.

Overall, we found evidence that wetlands in the U.S. have significant capacity to remove nitrogen pollution, and that these water quality benefits could be increased by prioritizing wetland restoration efforts in regions with a high nitrogen surplus. Recent estimates of the current total nitrogen removal capacity of U.S. wetlands are ~860 ± 160 kilotons of nitrogen per year (approximately 8% of the current U.S. nitrogen surplus), with the highest removal rates where high wetland density and high nitrogen surpluses overlap (Cheng et al. 2020). This estimate is an order of magnitude less than some previous estimates, but reflects a more refined approach that accounts for the 38% of HUC-8 watersheds across the U.S. with a relatively low density of wetlands and a high nitrogen surplus (i.e., areas of intense agricultural activity). Thus, this spatial disconnect likely limits the current nitrogen removal capacity of wetlands in the U.S., but also indicates significant potential for strategically-located wetland restoration efforts to result in significantly increased nitrogen removal in the future: a 10% increase in wetland area (equal to 5.1 million hectares, or approximately 5 times the wetland restoration area currently supported by the USDA Wetland Reserve Program) was projected to result in a 90% increase in nitrogen removal above current levels, or an additional 809 ± 395 kilotons of nitrogen removal across the contiguous U.S. (Cheng et al. 2020).

Coastal Wetlands

Coastal wetlands provide water quality benefits through the trapping and deposition of suspended particles, assimilation and uptake of nutrients by plants and algae (Barbier et al. 2011), and microbial-mediated removal via denitrification (Russell and Greening 2015). Rates of denitrification were estimated as 4 g N/m^2/yr in saltwater marshes, 1 g N/m^2/yr in mangroves, and 9 g N/m^2/yr in seagrass beds (Russell and Greening 2015). Barbier et al. (2011) also described the water quality benefits of seagrass beds as “dramatic” (without providing quantitative estimates).

To understand how water quality in coastal ecosystems is changing in the U.S., Oelsner and Stets (2019) analyzed a total of 95 sites draining into 56 unique estuaries and 295 recent (2002–2012) nutrient and sediment trends. For a majority of the rivers and streams flowing into estuaries considered most vulnerable to eutrophication, nitrate yields decreased over this time period, where yield is calculated by dividing the flow-normalized nitrate load by the watershed area, and approximately half had decreasing phosphorus yields. Nutrient inputs were attributed primarily to nonpoint agricultural sources. While overall nitrogen, phosphorus and sediment inputs decreased into coastal watersheds, continued reductions of N and P loadings into waterways would benefit coastal wetlands (Oelsner and Stets 2019) and help prevent or reduce the risk of eutrophication.

Coastal wetlands also provide water quality benefits through the removal of protozoal pathogens that pose a public health threat, including *Cryptosporidium parvum*, *Toxoplasma gondii*, and *Giardia lamblia*. In the Monterey Bay region of California, protozoal removal rates increased with water-vegetation contact time, suggesting that planting emergent vegetation across a wetland channel would improve pathogen removal rates (Daniels et al. 2014).

Freshwater and Riverine Wetlands

A wide body of literature provides evidence that freshwater and riverine wetlands provide water quality benefits through the removal of nutrients and other pollutants. In a review of 23 years of literature evaluating ecosystem services for aquatic habitats, “water quality” was the most recurrent concept in riverine and lacustrine studies (D’Alelio et al. 2021). Another review by Vidon et al. (2019, as cited in Stutter et al. 2019) found that riparian zones act as sinks for nitrogen in subsurface flow and total phosphorus in surface flow. The capacity of freshwater and riverine wetlands to remove pollutants is affected by landscape composition and the spatial configuration of wetlands relative to sources of pollutants in the landscape (Qiu and Turner 2015; Cheng et al. 2020). In an evaluation of 100 subwatersheds in Wisconsin, water quality was lowest in subwatersheds with more cropland and highest in subwatersheds with more wetlands, grassland, and forest, but the correlation was nonlinear such that water quality was highest where cropland covered <60% and/or wetlands covered >6% of the subwatershed (Qiu and Turner. 2015). As described above, Cheng et al. (2020) concluded that the nitrogen removal capacity of wetlands is increased when high density wetlands overlap with nitrogen hotspots (i.e., areas of intense agricultural activity).

The nutrient removal capacity of freshwater wetlands is also influenced by the duration of inundation and water management regime. In experimental riverine wetlands in Ohio, pulsed water flows appeared to favor greater N-removal than steady flows (Mitsch et al. 2012), while wet-dry cycles were associated with greater removal of N and P, as well as greater plant diversity in constructed depressional wetlands in Wisconsin (Doherty et al. 2014). Wet-dry cycles can also help promote the biodegradation of organophosphate-based insecticides in riverine and depressional wetlands in agricultural watersheds in California (Karpuzcu et al. 2013).

The protection or restoration of freshwater and riverine wetlands can also significantly improve the water quality benefits of freshwater and riverine wetlands (Duffy and Kahara 2011; Mitsch et al. 2012, 2014; Cheng et al. 2020; Gordon et al. 2020), depending on planting regime and hydrological connectivity, and given sufficient time since restoration. For example, for a pair of 1-ha created riverine marshes (one planted, one allowed to naturally colonize) in Ohio studied over 20 years, total phosphorus retention was higher in the planted wetland compared to the wetland that was naturally colonizing (44.3 ± 4.4% vs. 38.8 ± 5.3% respectively; $p = 0.059$) while total nitrogen retention was significantly higher in the naturally colonizing wetland compared to the planted wetland (32.1 ± 2.0% vs. 28.4 ± 2.6% respectively; $p = 0.000085$; Mitsch et al. 2014). Sedimentation rates also tended to increase over time after wetland restoration, with carbon and nitrogen both increasing in soil cores over time, suggesting that nutrient retention in wetlands could be sustainable for decades (Mitsch et al. 2014). Wetlands could consistently retain phosphorus in amounts of 0.5–5 g $P/m^2/yr$ and nitrogen in amounts of about 10–40 g $N/m^2/yr$ (Mitsch et al. 2000, as cited in Mitsch et al. 2014).

A global synthesis of field studies in riverine wetlands concluded that restored and reconnected floodplains can remove significant masses of nutrients from their associated waterways, particularly when coupled with restoration of dense wetland vegetation (Gordon et al. 2020). Removal rates for nitrate-N (NO_3−N) averaged 200 kg/ha/yr, (range: 2.35 to 962 kg-N/ha/yr) and removal rates for particulate P averaged 21.0 kg-P/ha/yr (range: net release of 14.6 to net retention of 130 kg-P/ha/yr). In-stream deposition of insecticides at vegetated riparian buffers was also reduced by an average of 96% compared with non-vegetated sites along agricultural ditches, suggesting that riparian buffers can also substantially reduce pesticide loading into streams (Hancock et al. 2019).

To increase water quality benefits, Gordon et al. (2020) specifically recommended: (1) Engineer the floodplain to optimize hydraulic load; (2) Incorporate a permanently inundated wetland in the floodplain area to improve NO3−N removal; and (3) Restore floodplains along waterways with higher concentrations of nutrients to increase the load of nutrients into the floodplain, similar to recommendations by Cheng et al. (2020). By improving floodplain connectivity and slowing the flow of water, the presence of beaver dams or installation of beaver dam analogs may also improve water quality benefits (see Chapter 8). Another recommended design solution, with the potential to more than double the total daily nitrate removal (110% increase) is to construct small floodplains adjacent to the main stream channels to spread water flows over wider, vegetated areas (Speir et al. 2020). However, these benefits took time to develop, with denitrification in the floodplain undetectable for up to one year post-restoration (Speir et al. 2020). Finally, in an optimization analysis, the construction of wide, slow-flowing, vegetated water bodies within the riverine corridor were the single-most cost-effective management action to reduce both nitrate and sediment loads in intensively managed agricultural landscapes (Hansen et al. 2021).

The installation of vegetated riparian buffer strips is another widespread approach to improving water quality, but with varying efficacy. In a meta-analysis of 46 studies published between 1980 and 2017, buffers consistently reduced total N in surface runoff by 57% (-68 to -43%), nitrates in surface runoff by 33% (-48 to -17%), and nitrates in groundwater by 70% (-78 to -62%,) compared with no-buffer controls (Valkama et al. 2019). Buffers were most likely to be effective at reducing impacts of nonpoint source pollution if they are designed to intercept contaminants and enhance diversity in surface topography (Stutter et al. 2019). Specifically, Stutter et al. (2019) recommend: (1) Near headwaters, variable-width buffers can promote pollution removal from diffuse sources and should include water channels, such as engineered ditches (2) Near crop fields, saturated buffers, tree zones and mini-wetlands can alleviate pollution pressures; and (3) Within floodplains, target restoration to increase wetland connectivity. In addition, water quality benefits generally increase with buffer width, with a minimum width of 5 meters recommended (data from Dal Ferro et al. 2019; Jaynes and Isenhart 2019; as cited in Stutter et al. 2019). Ramesh et al. (2021)found the highest sediment reduction efficiency associated with the widest buffers (10-20 meters wide). Based on a meta-analysis of all relevant publications since 1984, Lind et al. (2019) provided values for riparian buffer zone width to maximize support for species and nutrient removal benefits (Figure 3.1). However, saturated buffers can also promote conditions favorable to microbial denitrification and uptake of nitrates by buffer plants (Stutter et al. 2019), suggesting that narrower buffer widths could still be effective if they are saturated.

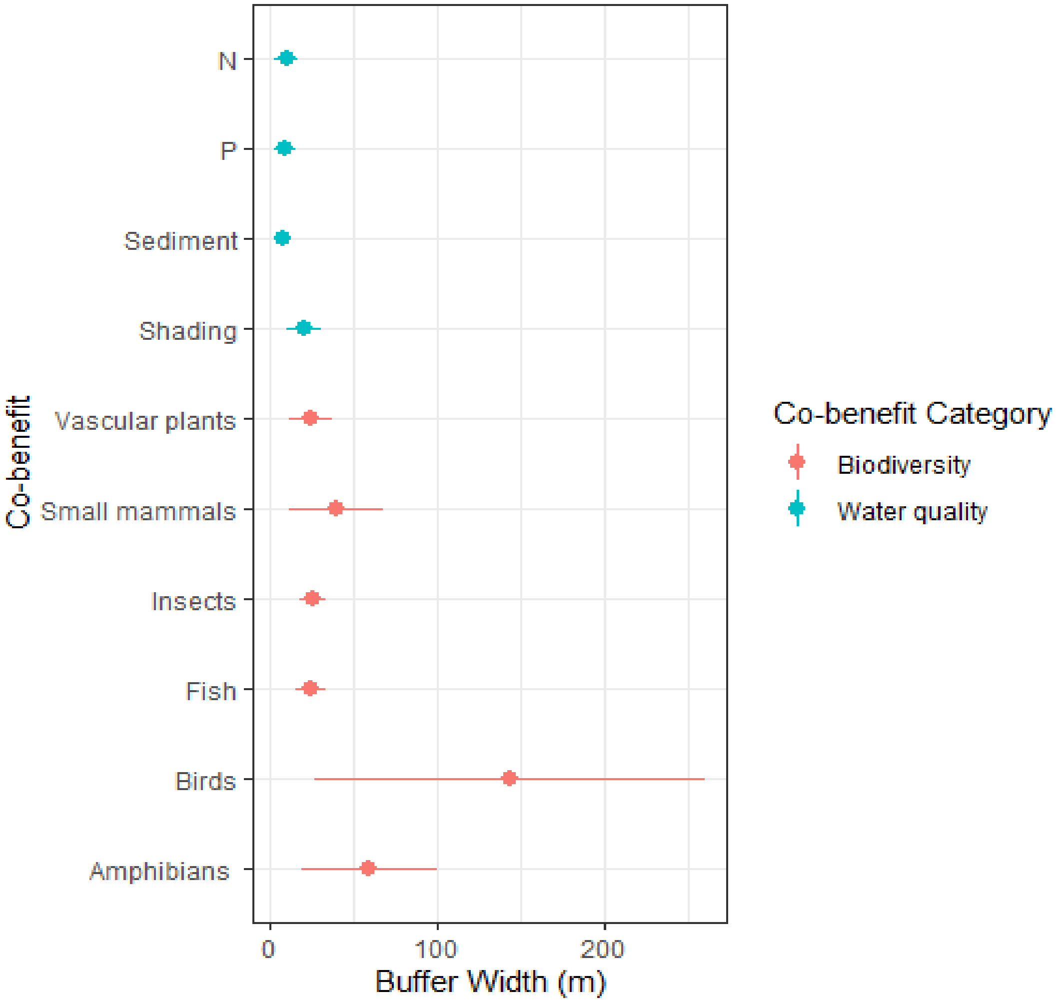

Figure 3.1. Mean riparian zone width in meters needed to provide biodiversity support and water quality benefits, based on quantitatively derived width recommendations from 134 data points from 43 peer-reviewed studies. Data are means ± SD; where error bars are not visible they are too small to discern. Figure modified from Lind et al. (2019).

Montane Meadows

As with other wetlands, intact montane meadows can provide water quality benefits through trapping and deposition of sediment and nutrients. In addition, the presence of beaver in montane meadows increased nutrient removal capacity, indicating a positive impact on water quality (Wegener et al. 2017; see Chapter 8). Conversely, degradation can reduce water quality in montane meadows. In a review of ten studies examining the impacts of livestock grazing on Sierra meadows, Vernon et al. (2022) found negative impacts of grazing on water quality including increased pathogen concentrations, increased algal biomass (and thus eutrophication potential), elevated fecal matter, and increased stream turbidity, sediment levels, and temperatures. Therefore, considering grazing regimes when managing montane meadows will likely help preserve water quality.

4. Soil Health

Soil health is essential to life on earth, and yet soils face global degradation (Kraamwinkel et al. 2021). Indicators of soil health include soil moisture, total organic carbon (C), and total nitrogen (N) content, as well as soil bulk density, an indicator of soil compaction. These soil properties can be highly variable within wetlands, and are directly related to other wetland benefits. For example, the structure and composition of the vegetation community, and thus the degree of biodiversity support in wetlands, can influence soil health (Rahe et al. 2015; Buchanan et al. 2020), while soil health in turn can affect water holding capacity, and thus water supply regulation benefits (Smith 2018). In addition, soil properties can be a primary driver of rates of nutrient cycling, and thus water quality benefits (Marton et al. 2015). Efforts to build soil organic matter can help restore soil health, but most metrics of soil health generally improve slowly, on a timescale of decades (Wolf et al. 2011; Rahe et al. 2015). Restoration of soil health can, in turn, enhance denitrification rates and help protect water quality (Marton et al. 2015; Rahe et al. 2015), albeit with a corresponding increase in N_2O emissions (see Chapter 3), and newer wetlands are likely to have lower rates of nutrient cycling (Wolf et al. 2011). Thus, newly created wetlands will take time to match the soil health benefits of intact wetlands.

Coastal Wetlands

Our rapid evidence assessment did not produce papers focused specifically addressing soil health in coastal wetlands in North America. Papers related to nutrient cycling in coastal wetlands were focused primarily on water quality, as described above (see Chapter 3).

Freshwater Depressional Wetlands

Soil health benefits in freshwater depressional wetlands can be highly variable within and among wetlands, and a comparison of natural and 10-year-old restored freshwater depressional wetlands found several differences (Marton et al. 2015). Natural wetlands had greater soil moisture, plant-available nitrogen, organic carbon, and total nitrogen, as well as lower bulk density than restored wetlands, all indicating better soil health. Restored wetlands had higher spatial variability in these indicators, as well as higher denitrification rates overall, suggesting greater water quality benefits, although the mechanism for this surprising result was ultimately unclear. While more study is needed to understand this variability and improve wetland restoration efforts, the authors concluded wetland restoration can support soil health and related benefits (Marton et al. 2015).

Riverine Wetlands

Soil health benefits in riverine wetlands may be influenced by multiple factors, including the composition of the vegetation community. For example, the installation of forested riparian buffers resulted in multiple improved indicators of soil health compared to adjacent agricultural fields, including greater soil total C, total N, and moisture levels, attributed to increases in organic matter inputs (Rahe et al. 2015). Further, soil total C and N were positively related to age of restoration, suggesting continued improvement of soil health over time (Rahe et al. 2015). Herbaceous plant communities in riparian buffers also matter, with lower rates of N mineralization and CO_2 efflux found in fast-growing herbaceous communities with high functional diversity (as opposed to taxonomic diversity), and reduced N runoff (Buchanan et al. 2020). Finally, soil organic matter may be higher where groundwater levels are higher (Barksdale et al. 2014).

Montane Meadows

In our evidence assessment, we found a substantial focus on the impacts of grazing to the soil health properties of montane meadows, including soil organic matter content, water holding capacity, nutrient cycling, and soil bulk density. Some authors have found an increase in soil nutrients associated with grazing (Blank et al. 2006, as cited in Ramstead et al. 2012), while a recent review that included five studies evaluating the impacts of grazing on soils in Sierra Nevada meadows found that livestock grazing can affect soil nutrient availability and cycling (Vernon et al. 2022). The intensity of grazing may also play a role in the degree of impact, with low intensity grazing having less impact on soil nutrients, though still impacting meadow hydrology (Vernon et al. 2022). Similarly, other authors reported significant impacts to soil properties from grazing but also rapid recovery once meadow restoration occurred (Wheeler et al. 2002, as cited in Ramstead et al. 2012). Thus, the soil health benefits of montane meadows can be influenced by management decisions about grazing intensity and restoration.

5. Biodiversity Support

Wetlands are well-recognized as biodiversity hotspots, with up to 40% of the world's species living in or breeding in wetlands (Convention on Wetlands 2021). Their role as interfaces between aquatic and terrestrial ecosystems, together with the structural habitat diversity they provide and their high nutrient inputs and primary productivity contribute to supporting biodiversity (Barbier et al. 2011; Ramstead et al. 2012; Craig et al. 2014). In turn, increased biodiversity contributes to a number of other wetland benefits such as food and fiber provisioning, pollination, and pest control, as well as water quality (see Chapter 3), soil health (see Chapter 4), and carbon (see Chapter 7). However, more than 25% of wetland plants and animals are at risk of extinction (Convention on Wetlands 2021), threatening all of these benefits. The loss and degradation of wetland habitat is a major driver of this global biodiversity loss. Fortunately, restored, managed, and created wetlands can all successfully create wetland habitat for many species (Golet et al. 2008; Gleason et al. 2011; DiGaudio et al. 2015; Campos et al. 2020).

For wildlife in the conterminous United States, wetland cover was significantly correlated with species richness within taxonomic groups, including amphibians, birds, mammals, reptiles, and endemic terrestrial species (Dertien et al. 2020). The sign of the correlations varied by region, with positive correlations across much of the country for amphibians, birds, and endemic species groups, and positive correlations for reptiles and mammals in specific regions. Certain regions stood out as hotspots of positive correlations, including the Great Plains, the Southeastern Plains and Piedmont of the eastern U.S., and surprisingly the Mojave and Sonoran Desert Basins, where the authors estimated that even a small increase in wetland cover could have a major impact on species presence or absence (Dertien et al. 2020). Thus, while wetland conservation throughout the U.S. is likely to benefit many wildlife species across taxa, collaborative efforts on ecoregional scales that work across state lines, such as Migratory Bird Joint Ventures (www.mbjv.org), may be important to establishing effective wetland conservation initiatives, priorities, and policies.

Coastal Wetlands

Estuaries are important nurseries for a breadth of species (Barbier et al. 2011), including fish and birds. For threatened lower Columbia River (USA) coho salmon (*Oncorhynchus kisutch*), subyearling, and to a lesser extent, yearling *O. kisutch* utilized the tidal freshwater estuary habitats, including emergent and forested wetlands, supporting the importance of these wetlands to the population (Craig et al. 2014). In the San Francisco Estuary (California, USA), an analysis based on long-term monitoring concluded that spatially-targeted tidal marsh restoration benefits a suite of bird species, making their populations more resilient to sea level rise (Veloz et al. 2013).

Wetlands also provide biodiversity support for economically important species. Salt marshes may account for 66% of the shrimp and 25% of the blue crab production in the Gulf of Mexico (data from Zimmerman et al. 2000, as cited in Barbier et al. 2011). Densities of economically important brown (*Farfantepenaeus aztecus*) and white (*Litopenaeus setiferus*) shrimp peaked at 13.4 and 8.9 shrimp/m^2 (respectively) at vegetated salt marsh edges in Galveston Bay, Texas (Minello et al. 2012), supporting the concept that marsh edge ecotones support the highest shrimp populations. Annual total shrimp production ranged from 90 to 146 kg/acre in constructed salt marshes (Minello et al. 2012). The highest shrimp production was seen in projects that included terracing and island construction, and the lowest where construction included only dredging (Minello et al. 2012).

Freshwater Depressional Wetlands

Freshwater depressional wetlands provide important habitat for diverse and unique plants and animals often specialized for ephemeral water availability or conditions (Gleason et al. 2011). For example, Prairie Potholes represent critical waterbird breeding habitat, along with important habitat for dozens of at-risk species (Gleason et al. 2011), and playa wetlands store water in highly arid parts of the U.S., which gives them a major role for biodiversity preservation as critical habitat for an array of wildlife (Daniel et al. 2015), including breeding birds (Behney 2021). Restoration of freshwater depressional wetlands can improve these benefits, with restored wetlands in the Prairie Pothole region supporting similar invertebrate, mammal, and amphibian populations as native prairie wetlands (Gleason et al. 2011). In playa wetlands, restoration has been shown to promote insect pollinator conservation (Begosh et al. 2020), and playas with vegetated buffers in the High Plains supported substantially greater abundances of both wetland-dependent and total birds during the breeding season compared to unbuffered playas (Behney 2021).

Riverine Wetlands

The evidence for biodiversity support in riverine wetlands primarily focused on riparian forests, which are well-recognized hotspots for biodiversity, particularly in the arid western United States (Knopf et al. 1988; Knopf and Samson 1994). Riparian forests support a rich diversity of organisms including birds (Golet et al. 2008; Heath et al. 2017; Dybala et al. 2019b), herpetofauna (Banville and Bateman 2012; Guzy et al. 2019), small mammals, and insects (Golet et al. 2008). Riparian forests are also important for providing shade that maintains cool water temperatures for fish, such as Brook Trout in Wisconsin streams (Cross et al. 2013). The abundance and diversity of species supported by riparian forests is likely to be influenced by vegetation structure, composition, and heterogeneity. For example, a study in the Sierra Nevada of California found that riparian bird occupancy increased with more willow (*Salix* spp.) cover and less overstory cover, while bumble bee occupancy increased with greater flowering plant richness, more forb cover, and less shrub cover (Cole et al. 2019). Similarly, structural diversity and woody debris may support more herpetofauna (Banville and Bateman 2012), while very high forest stand densities may negatively impact bird abundance and diversity (Dybala et al. 2019b).

Restoration of riparian vegetation does successfully provide habitat for many taxa, including birds, bats, rodents, and insects (Golet et al. 2008), with benefits to birds increasing with time since restoration (Gardali et al. 2006; DiGaudio et al. 2015; Dybala et al. 2018). Restoration of urban riparian ecosystems also provides habitat, with six times the abundance of herpetofauna found in restored compared to disturbed urban reaches (Banville and Bateman 2012). Further, the installation of vegetated riparian buffers can also provide habitat for many taxa, with occupancy and species richness consistently increasing with buffer width; optimal widths ranged from 25m for insects and fish, to 35m for reptiles, 55m for amphibians, and 140m for birds (Figure 3.1; Guzy et al. 2019, Lind et al. 2019).

In addition to restoring vegetation, restoring the natural hydrology and connectivity between rivers and their floodplains can also be important for improving multiple ecosystem functions, including vegetation recruitment, sediment and nutrient deposition, and groundwater recharge (Swenson et al. 2003; Yoder 2018). Restored hydrology has been associated with improved vegetation productivity and resilience (Silverman et al. 2019) and supporting the species diversity, relative abundance, and growth of riverine and anadromous fishes (Beechie et al. 2013; Phelps et al. 2015). In addition, periodic flooding in restored floodplains can help control populations of pest mammal species (Golet et al. 2013).

The methods used and the site-specific context of a riverine wetland restoration may have important influences on the extent of biodiversity support and interactions with other wetland benefits. For example, comparing two restored riverine marshes, Mitsch et al. (2014) found that the marsh in which vegetation naturally recruited was more quickly dominated by cattail (*Typha* spp.) than the marsh that was planted with a more diverse set of species, suggesting a benefit of investing in planting effort. However, the authors attributed the eventual cattail monoculture in both marshes to the high nutrient load from adjacent agriculture, but noted that disturbances by wildlife (muskrat, beaver, geese) or changes in water level (e.g., through dam removal) could reduce their dominance, indicating the importance of factors beyond the immediate restoration site and the importance of ongoing adaptive management decisions to maintain wetland benefits. Similarly, a study in California found that riparian forest restoration can provide more habitat to birds while also storing more carbon than unrestored sites, whether restored through planting or natural recruitment of vegetation, but also found that where the restored forest was densest, carbon stocks were highest while bird abundance and diversity was lowest, suggesting a trade-off between carbon and biodiversity support benefits (Dybala et al. 2019b). Thus, restoration decisions about planting density or management decisions about subsequent forest management can influence the magnitude of wetland benefits and how they interact with each other.

Montane Meadows

Montane meadows are also well-recognized biodiversity hotspots that serve as important habitat for an array of fish and wildlife species (Patton and Judd 1970; Graber 1996; Kattelman and Embury 1996; Siegel and DeSante 1999), including large ungulates (Patton and Judd 1970), small mammals (NFWF 2010) and many listed species including the Yosemite toad (*Bufo canoru*), the mountain yellow–legged frog (*Rana muscosa*), and endemic populations of Apache trout (*Onocorhynchus gilae var. apache*) (AGFD 2001, as cited in Ramstead et al. 2012). Meadows also provide important habitat for fish and birds, serving as critical habitat for eight native trout species in California's Sierra Nevada (Moyle et al. 2008) and having been described as the single most important habitat requirement for some migratory landbird species in the Sierra Nevada (Siegel & DeSante 1999).

Widespread disturbance and degradation of montane meadows has contributed to endangered, threatened, and declining vertebrate populations, but efforts to restore meadows have been successful in improving habitat quality for birds, bats, small mammals, fish, and benthic macroinvertebrates (Ramstead et al. 2012; Campos et al. 2020; Loffland et al. 2022). Across 31 hydrologic restoration projects using pond-and-plug techniques that ranged 1–18 years old, Campos et al. (2020) found an increase in abundance for 6 of 12 bird species evaluated with time since restoration, with no species responding negatively and the strongest positive responses for rarer species that were largely missing from the youngest restoration sites. In alignment with these results, Ramstead et al. (2012) reported higher nest success for ground nesting birds in a restored portion of montane meadow compared to the unrestored, grazed portion, and Loffland et al. (2022) reported significantly slower declines in abundance at restored meadows in the Sierra

Nevada for willow flycatcher (*Empidonax traillii*), a California endangered species with widespread population declines over a two-decade period.

As with riverine wetlands described above, the restoration methods used and the site-specific context of montane meadow restoration may have important influences on the extent of biodiversity support. Restoring the natural hydrology of montane meadows, such as through pond-and-plug techniques or low-tech practices including beaver dam analogs (see Chapter 8), is associated with revegetation success and vegetation resilience (Ramstead et al. 2012; Silverman et al. 2019), and in turn, the response of wildlife. Reducing disturbance through robust grazing management strategies may also reduce the impacts to meadow fish and wildlife (including birds, amphibians, aquatic macroinvertebrates, fish, and mammals), such as: excluding livestock from sensitive areas like fens and riparian zones, reducing stocking levels and/or grazing duration, using rest-rotation grazing regimes, and avoiding impacts during the breeding season (Vernon et al. 2022). However, restoration of highly degraded meadows is not guaranteed to improve habitat conditions beyond the average condition of unrestored meadows (Pope et al. 2015), and Campos et al. (2020) found that half of the bird species evaluated did not significantly increase in abundance following restoration. Thus, improvements in restoration methods, designs, and priorities may be needed. For example, prioritizing restoration in geographies where target bird species tend to occur in higher abundances, and planting dense vegetation at the time of restoration may accelerate the response of bird populations (Campos et al. 2020). Avoiding the establishment of non-native species, such as green sunfish (*Lepomis cyanellus*) and bullfrog (*Rana catesbeiana*) may require managing wet meadows so that they do not have permanent ponding (Ramstead et al. 2012).

Managed Wetlands

Our evidence assessment also found that biodiversity support is a common goal of managed wetlands. By managing the vegetation cover and the timing and depth of water application, managed wetlands can provide valuable habitat for target species. In California's Central Valley, over 90% of the historical wetland extent has been converted to other land uses and most of the flooded habitat currently available is a combination of managed wetlands and irrigated agriculture, yet the Central Valley still supports hundreds of bird species representing millions of individual birds (CVJV 2020). In addition, Central Valley private lands that have been restored, enhanced, and managed as wetlands through voluntary easements and incentive programs supported 181 bird species during the breeding season and 88 species during fall migration, including special status species targeted for conservation (DiGaudio et al. 2015). In West Virginia, managed wetlands supported 3,348 waterbirds representing 27 total species over the winter non-breeding season, and differences among managed wetlands suggested wetland size, depth, and topography influenced the composition of species present (Clipp et al. 2017).

6. Economic Value

Many of the benefits wetlands provide also represent significant economic value to human communities, especially flood risk mitigation benefits, water quality benefits, and the commercial and recreational value of biodiversity support benefits. The studies in our evidence assessment frequently included an evaluation of the economic value of these benefits to illustrate the cost of wetland loss and degradation, the potential value of wetland restoration, and/or the value of wetlands relative to alternative infrastructure (e.g., in the case of flood risk mitigation and water quality benefits). Although we note that these estimates of economic value are necessarily incomplete, due to the difficulty of representing several dimensions of value in economic terms (e.g., cultural and aesthetic values) and that we have not adjusted these estimates for inflation since the studies were completed, we found evidence that wetlands provide significant economic benefits. For example, in a meta-analysis of the economic benefits for wetlands of all types in agricultural landscapes (i.e., within 5km), Brander et al. (2013) estimated an average value of $3,389 per hectare per year for water supply regulation benefits, $6,923 per hectare per year for flood risk mitigation benefits, and $5,788 per hectare per year for water quality benefits. Across a conservatively estimated 59 million hectares of wetlands in agricultural landscapes in the U.S., the authors estimated a total economic value for just these three benefits of $1.8 billion USD per year (95% confidence interval: $1.3 – 61.3 billion USD per year). In addition, strategic placement of wetland restoration could substantially increase these values, such as locating wetlands near property at highest risk of damage from flooding (Kousky and Walls 2014) or near nitrogen surpluses to increase water quality benefits (Cheng et al. 2020, as discussed in Chapter 3).

Coastal Wetlands: storm surge protection, water quality benefits, and fisheries

One of the most significant economic benefits provided by coastal wetlands may be the protection they provide from storm surges (Barbier 2013). Globally, severe storms including hurricanes and tropical storms have resulted in an estimated loss of $17.9 billion in property and infrastructure damages since 1900 (Costanza et al. 2008), but by attenuating wave energy and dissipating the impacts of flood waters associated with rising seas and storm surges (see Chapter 2), coastal wetlands may save billions of dollars in avoided damages. For example, Barbier et al. (2011) estimated that salt marshes reduced economic damages from hurricanes by $8,236 per hectare per year, while Salem and Mercer (2012) valued the protection provided by mangroves at an average of $3,116 per hectare per year. In southeastern Louisiana, the value of the protection provided by coastal marshes was equivalent to saving up to five residential properties per storm event, based on mean property values (Barbier 2013). The total value of the protection provided by coastal herbaceous wetlands along the U.S. Atlantic coast was estimated as $23.2 billion per year (Martinez et al. 2011). Menéndez et al. (2020) projected that the annual economic value of mangroves for protecting U.S. coastal cities is more than $250 million.

Where coastal wetlands have been fragmented or degraded, even small improvements in wetland continuity and roughness (a measure related to marsh vegetation) could reduce flood damages associated with storm surges in the range of hundreds of thousands of dollars (Barbier 2013). Where coastal wetlands have been entirely lost, wetland restoration can still be a cost-effective approach to providing storm surge protection, particularly for mangroves and salt marshes. For example, Narayan et al. (2016) estimated that restoring both mangroves and salt marshes can be 2–5 times less expensive than deploying engineered options for wave heights up to half a meter, estimating the cost per m^2 restored at $0.10 for mangroves (range: $0.05–$6.43) and $1.11 for salt marsh (range: $0.01–$33.00) (Narayan et al. 2016). In addition, Martinez et al. (2011) noted that wetlands provide a number of other valuable benefits beyond flood protection that cannot be provided by engineered options, estimated at $11,700 per hectare per year (in 2004 USD) and expected to increase in value over time.

Coastal wetlands also provide economic benefits by improving water quality (see Chapter 3) and in some cases avoiding the cost of a traditional wastewater treatment plant. For example, in the absence of functioning wetlands, the cost of removing a kilogram of nitrogen with an engineered option (like a wastewater treatment facility) ranged from $2.71 to as high as $1,885, depending on the quality of the treatment facility and the difficulty of routing water to treatment areas (Russell and Greening 2015). Thus, efforts to expand seagrass, coastal marsh, and mangroves in Tampa Bay (Florida, United States) are estimated to have resulted in nutrient reductions equivalent to $24 million per year in avoided cost, a value expected to grow with an increasing population (Russell and Greening 2015). Other estimates of avoided wastewater treatment cost varied from $1,940–$37,066 per hectare ($785 to $15,000 per acre) in capitalized costs in salt marsh swamps (Barbier et al. 2011), while Salem and Mercer (2012) estimated a mean value for mangroves of $44 per hectare in nutrient retention and $4,748 per hectare in water and air purification (Salem and Mercer 2012).

Coastal wetlands also provide substantial economic benefits by supporting biodiversity that has commercial and recreational value. For example, a majority of the fish and shellfish harvested both commercially and recreationally in the U.S. are supported by wetland ecosystems, including two of the most valuable fisheries in the United States: brown shrimp (*Farfantepenaeus aztecus*) and white shrimp (*Litopenaeus setiferus*). Minello et al. (2012) estimated the value of shrimp production from nine constructed salt marshes in Galveston Bay, Texas (United States) at $425–$690 per hectare per year. In mangroves, Salem and Mercer (2012) estimated a mean valuation of $23,613 per hectare per year from fisheries, but also $37,927 per hectare per year for the value of recreation and tourism, and $38,115 per hectare per year for timber and forestry products.

Finally, coastal wetlands can also provide economic benefits by sequestering carbon. Salem and Mercer (2012) estimated a mean valuation of $967 per hectare per year in mangroves, based on global carbon prices. Using carbon sequestration rates from Mcleod et al. (2011) and a conservative estimate of the total social cost of carbon, Russell and Greening (2015) estimated the combined value of carbon sequestration in Tampa Bay coastal wetlands (seagrass, salt marsh, and mangroves, also known as “blue carbon”) as close to a quarter of a million dollars annually.

Freshwater Depressional Wetlands: carbon, soil health, and waterfowl

Our rapid evidence assessment identified only one study that evaluated the economic benefits provided by freshwater depressional wetlands, which estimated the change in economic value for scenarios of land use change in the Prairie Potholes Region of the Dakotas (Gascoigne et al. 2011). The authors estimated that an aggressive conservation scenario in which there is a 50% increase in the area of cropland enrolled in conservation programs (Conservation Reserve Program and Wetlands Reserve Program) with no loss native prairie would produce a net economic benefit of $3.6 million per ha per year, considering the value of increased carbon sequestration, increased soil health by reducing erosion, and biodiversity support by increasing waterfowl reproduction for recreational hunting, as well as accounting for the loss of crop market value. However, the loss of native prairie was projected to incur a greater total cost than the gains achieved by enrolling cropland in conservation programs, suggesting the importance of conserving existing native prairie in the region.

Riverine Wetlands: flood protection, property values, and fisheries

Like coastal wetlands, one of the most significant economic benefits provided by riverine wetlands may be the protection they provide from flood damage. Where floodplains are hydrologically connected to river and stream channels, they can store floodwater and slow peak flow events, protecting property and infrastructure downstream. For example, in Middlebury, Vermont, the value of flood risk mitigation benefits from a network of riverine wetlands was estimated at $126,000–$459,000 per year in avoided property damage alone, sufficient to cover more than a quarter of the estimated cost of wetland conservation in the watershed (Watson et al. 2016). Similarly, in St. Louis County, Missouri, the value of flood risk mitigation benefits from the Meramec Greenway, a collection of protected parcels in the Meramec River floodplain, was estimated as $7.7 million per year in avoided damages, and emphasized that this value could be improved by targeting parcels at highest risk of damage (Kousky and Walls 2014). While this total value was not as high as the alternative of having single-family homes on the same parcels ($17.2 million), this difference was more than overcome when the increase in home property values near the Greenway was accounted for ($24 million), suggesting the high aesthetic and recreational value of living near a protected area. A study of levee districts along the Lower Illinois River demonstrated that the agricultural profits in many cases were outweighed by the estimated wetland benefits from reconnecting floodplains (Guida et al. 2016).

Riverine wetlands also provide economic benefits by supporting biodiversity with recreational and commercial value. Of the fishery harvest in the Great Lakes, an estimated 50% of the commercial harvest and 80% of recreational harvest comprises species that rely on the riverine and lacustrine wetlands along their coasts (Trebitz and Hoffman 2015). Similarly, in British Columbia, floodplain restoration sites were estimated to have directly contributed to the production of 27%–34% of the out-migrating wild coho (*Onchorhynchus kisutch*) smolts, at a cost that was comparable to hatchery production (Ogston et al. 2015).

7. Carbon

In this section, we synthesize the extensive scientific literature for the carbon benefits of wetlands, with an in-depth examination of estimates for carbon storage in wetland soils and aboveground biomass, followed by estimates of carbon fluxes between wetlands and the atmosphere, and evidence for the impacts of disturbance on carbon storage and fluxes and the effectiveness of wetland restoration in recovering carbon benefits. However, we begin with essential background information on the role of wetlands in the global carbon cycle, important factors influencing carbon storage and fluxes in wetlands, and an overview of carbon accounting methods and essential terms used in the scientific literature.

7.1 Background

Wetlands in the global carbon cycle

The carbon cycle refers to the movement of carbon between different carbon pools. Any location where carbon is stored can be called a reservoir or pool. At the broadest scale, carbon pools include the atmosphere, oceans, soils and sediments, and living organisms, but each of these pools can also be more finely subdivided, such as the carbon stored in live trees, leaf litter, or roots. Fluxes refer to the processes by which carbon moves between pools, and the rates and magnitude of these fluxes influence the longevity of carbon storage within a pool. For example, through the process of photosynthesis, plants remove carbon dioxide (CO_2) from the atmosphere and store the carbon in their tissues (assimilation), which then become the food source for nearly all other life on Earth. As animals and microbes break down this organic matter, they release carbon dioxide back to the atmosphere (respiration) while also transferring some of the organic carbon into long-term soil and sediment pools. (We distinguish soils as resulting from rock weathering in a site, typically facilitated by organic growth, and sediments as resulting from erosional transport of material away from a weathering site and deposition in a new location.) Thus, when assimilation outpaces respiration, more carbon is accumulating in living organisms and dead organic matter than is being released to the atmosphere, resulting in a net carbon sink. To understand the net carbon benefits of wetland conservation and restoration, it is essential to estimate both the magnitude of the existing carbon pools and the carbon fluxes into or out of these pools (Howard et al. 2017).

Wetlands excel at removing CO_2 from the atmosphere. To date, CO_2 is the primary anthropogenic greenhouse gas (GHG), accounting for approximately 80% of all GHG emissions from human activities. It is also long-lived (multi-century lifespan; Inman 2008), and global atmospheric concentrations continue to climb, now exceeding 417 ppm (Dlugokencky and Tans). One major reason wetlands are so effective at sequestering CO_2 is that microbial respiration requires oxygen to efficiently decompose organic matter in the soil, and flooded soils are lacking in oxygen (they experience sub- or anoxic conditions). Thus, organic matter does not break down as rapidly, and the carbon it contains has the opportunity to accumulate in wetland soils rather than being released back into the atmosphere as CO_2. However, understanding the net carbon benefits of wetlands also requires accounting for the release of methane (CH_4) by specialized microbes that can metabolize carbon in the absence of oxygen (methanogenesis). Methane is the second most important GHG relevant to warming. While much less prevalent (only 1.89 ppm as of 2020) and short-lived (12-year atmospheric lifespan) compared to CO_2, it is also a more potent

GHG (Schiermeier 2020 Jul 14). Potency refers to the ability of the molecule to absorb energy in the atmosphere, and a common and widely used metric for comparing the relative contributions of different molecules to absorb energy is to estimate its global warming potential (GWP). GWP integrates the radiative forcing (i.e., energy absorbed, measured in Watts/m^2) of a mass of GHG emissions relative to the same mass of CO_2 over a specified time frame, where the GWP of CO_2 is, by definition, always 1 (IPCC 2013). Across a 20-year time frame, the GWP of methane is about 84 times more than CO_2, but over a 100-year time frame, by which point much of the methane will have already degraded, its GWP is reduced to an estimated 28-34 times more than CO_2 (IPCC 2013).

While it is well-understood that wetlands emit significant quantities of methane, the exact magnitude of those emissions is unclear, with different modeling approaches arriving at different estimates. For example, a bottom-up process incorporating estimates of land surface emissions, atmospheric chemistry, and data-driven extrapolations, produced a higher estimate than previously reported for aquatic ecosystems at 431 Tg/yr (95% confidence interval: 343–519), representing 41% (median) to 53% (mean) of the global total (Rosentreter et al. 2021). However, bottom-up processes typically yield higher estimates than top-down studies using atmospheric observations within an atmospheric inverse-modeling framework and can be 60% lower (149 Tg/yr; Saunois et al. 2020). In either case, even though wetlands excel at removing CO_2 from the atmosphere, methane emissions will at least partially offset those benefits, particularly over shorter time frames. In addition, the rate of methane emissions varies by wetland class, with a higher rate within freshwater and brackish wetlands (salinity less than 10-18 parts per thousand), than in coastal wetlands, where high sulfate in sea water suppresses methanogenesis (Figure 7.1a, c).

Another important factor influencing the net carbon benefits of wetlands is disturbance of the existing carbon pools. Carbon can remain stored in soils and woody biomass over very long time-frames until a disturbance releases it. For example, fires and disease can release carbon stored in trees, and landslides can expose buried organic matter, making it available for microbial respiration in the presence of oxygen. A similar process occurs when previously inundated areas like wetlands become dry, such as temporarily during a drought, or permanently due to intentional drainage (Figure 7.1b, d). Upon being exposed to oxygen after draining, previously anoxic soils can lose stored carbon. The release of carbon from drained wetlands can be extensive, especially when coupled with fire (Marengo et al. 2021), earth moving (Oikawa et al. 2017), or even wind. Thus, carbon that took centuries to build up in various pools can take days to decades to be released.

Beyond these two major factors - salinity and disturbance - carbon storage and fluxes in wetlands can vary as a function of several other factors (Table 7.1). These include broad factors such as soil type (e.g. peat versus mineral soil) and region (and related climate), as well as factors that can change rapidly across space (i.e. from one end of a wetland to the other) or over time (i.e. the course of a year, season, or day), such as hydrology, salinity, vegetation input, and nutrient load. They can also be influenced by the presence of fauna. For example, fish that feed on zooplankton can increase the presence of methanotrophic bacteria (bacteria that metabolize methane), reducing methane emissions (Devlin et al. 2015). In addition, studies may produce different estimates of carbon storage and flux depending on the methodology used (discussed further in the next section).

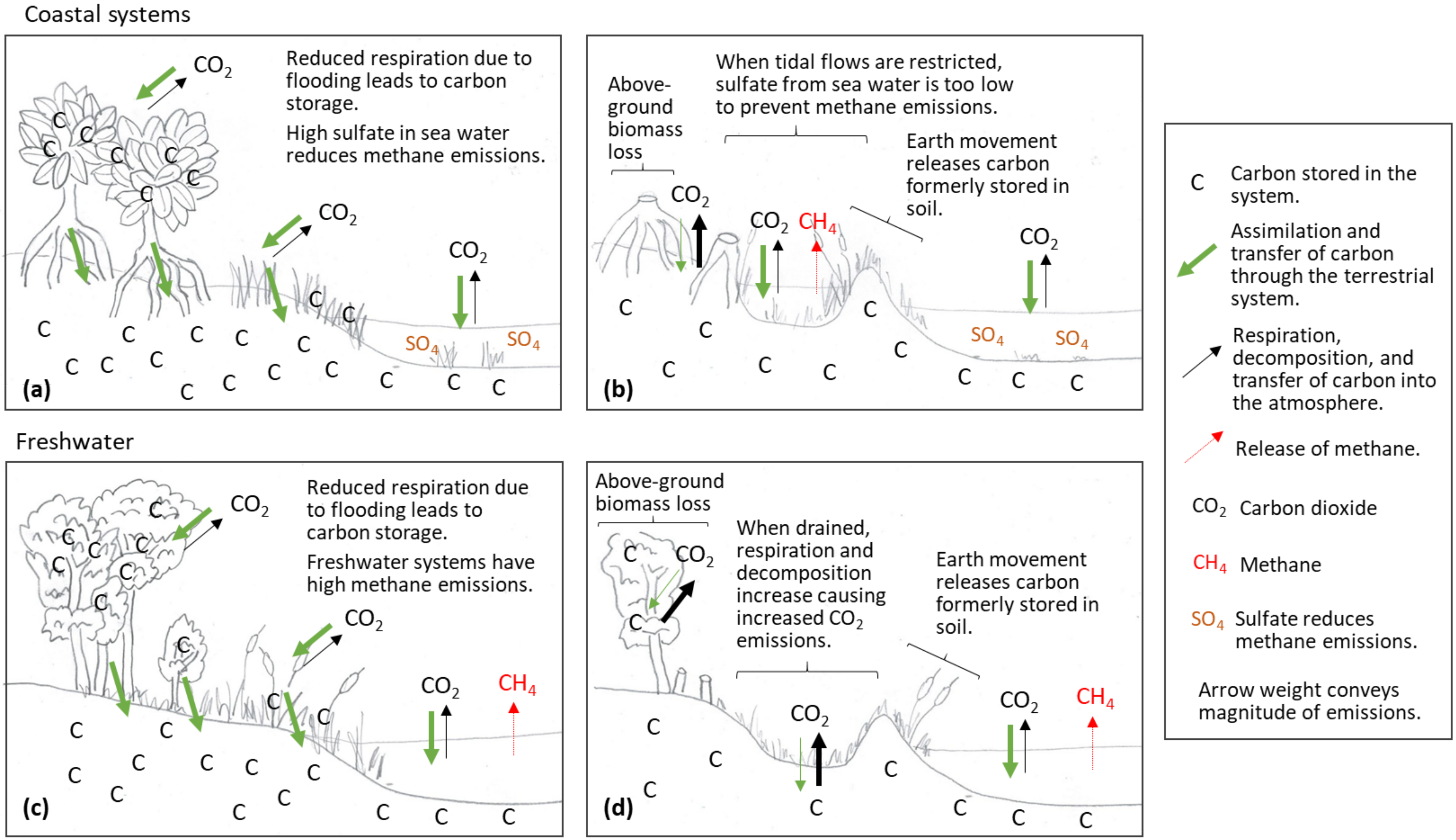

Figure 7.1. Carbon storage and fluxes in intact and disturbed wetlands. Carbon is exchanged between the atmosphere, the terrestrial biosphere, and water bodies in (a) intact coastal wetlands, (b) damaged and degraded coastal wetlands, (c) intact freshwater wetlands, and (d) damaged and degraded freshwater wetlands.

Table 7.1. Wetland covariates and their expected influence on carbon flux and storage.

Covariate	Influence on Flux	Influence on Pool
Hydrology	Consistently inundated soils have lower CO_2 respiration (Wilson et al. 2016; Amendola et al. 2018; Tangen and Bansal 2020) and higher methane emissions than seasonally or perennially dry soils, depending on salinity level. There is often a lag in the onset of flooding and increased carbon storage (Tangen and Bansal 2019) and increased methane emissions (Chang et al. 2021). In permanently inundated wetlands, depth can influence methane flux, where methane is typically produced in anoxic sediments/soil and deep wetlands can be thermally stratified with reduced methane diffusion or ebullition to the surface (Bastviken et al. 2004).	Locations that have been consistently inundated are likely to contain high soil carbon (Roche et al. 2014; Tangen and Bansal 2020). Over longer time frames (decades to centuries), the warming effects of the methane emitted by freshwater wetlands can be more than compensated for by high carbon storage rates and high total cumulative carbon stored (Taillardat et al. 2020).
Salinity	Seawater with salinity greater than 10-18 ppt typically coincides with high sulfate, which inhibits microbial methane production, decreasing methane emissions (Poffenbarger et al. 2011; Kroeger et al. 2017; Rosentreter et al. 2021). An exception to this rule are salt ponds where very high salinity can lead to a microbial community with high methane production (Zhou et al. 2022).	"Blue carbon" systems are frequently celebrated for their high soil carbon storage and low methane emissions (Howard et al. 2017).
Region	Soil carbon accumulation rates vary by region (Ouyang and Lee 2014) due primarily to underlying geology and nutrient transport, vegetation and productivity, and also due to climate.	Irrespective of climate, certain regions can store more carbon due to nutrient transfer across water bodies. In the U.S., this leads to different carbon storage in different locations (Nahlik and Fennessy 2016).
Nutrient Loads	Increased nutrient loading increases both plant growth and respiration. One meta-analysis found that nitrogen addition significantly increased CO_2 emissions across climate regions, with the exception of decreased CO_2 emissions in temperate continental regions (Chen et al. 2020). Another study found fertilization increased aboveground biomass and elevation gain (due to trapped sediment) on the North Carolina coast (Davis et al. 2017). Whereas, in coastal South Carolina, fertilization led to greater respiration and net emissions (Morris and Bradley 1999). Emissions of methane and N_2O, potent greenhouse gasses, also increase with increasing nutrient inputs (Rosentreter et al. 2021).	A global meta-analysis across ecosystems found increasing soil organic carbon with increasing nutrients (Xu et al. 2021). This result was not significant when wetlands were analyzed alone, however the authors acknowledged a lack of wetland observations.
Geochemistry	Methane production (methanogenesis) requires saturated or nearly saturated soil conditions and the absence of competing, more electronegative electron acceptors (NO_3^-, SO_4^{-2}, Fe^+) (Valentine et al. 1994).	Amendola et al. 2018 found that soil carbon storage varied with geochemical properties, having a positive correlation with aluminum and iron content and a negative correlation with pH and clay content.

Covariate	Influence on Flux	Influence on Pool
Flora and Fauna	Types of vegetation and their influence on nutrients (Moyes and Bowling 2016; Hinson et al. 2019; Kauffman et al. 2020a) and lignin content can affect the cycling of carbon through the system. Sphagnum mosses are associated with methanotrophs (Kostka et al. 2016). Zooplankton consuming fish can increase the presence of methanotrophic bacteria (Devlin et al. 2015).	Carbon in woody plant tissue is likely to be stored for longer periods of time. Further, different vegetation types will have different above:belowground carbon storage ratios. Biologically diverse mangroves in Indonesia have higher carbon storage (Rosentreter et al. 2018).
Climate	Provided adequate moisture, increasing temperature increases plant productivity and respiration. Provided adequate warmth, increasing precipitation increases plant productivity and respiration. Depending on level of inundation, wetlands may or may not be sensitive to the additional water provided by precipitation. The resulting difference between plant growth and respiration is typically small but increasing with increasing temperature or precipitation (Chen et al. 2019). Yet, despite regional differences in carbon storage that suggest an influence of climate influence, climate is not nearly as important as hydrology. In contrast, methane emissions typically increase with increasing temperature, with implications for climate change (Zhu et al. 2020).	Warming is expected to decrease soil carbon (Villa and Bernal 2018), but the effect is weak (Hinson et al. 2019).
Disturbance	Decadal carbon flux estimates must also account for periodic disturbances and interannual variability (Braswell et al. 1997). For example, periodic drying may release much of the carbon stored over years of reduced respiration due to inundated soils. Fires and beaver dams are other disturbances that can affect carbon cycling.	After a disturbance, carbon that has taken centuries to accumulate can be lost in a manner of years and decades.
Methodology	Carbon cycling measurements are influenced by how they are measured (e.g., chamber vs. eddy flux; Baldocchi 2003; Knox et al. 2019), soil depth (Nahlik and Fennessy 2016), how the disparate carbon pools are combined to get landscape measurements (Baldocchi et al. 2003), whether measurements are taken over long- versus short-time spans and how wetlands are classified (Loder and Finkelstein 2020). Finally, obtaining wetland soil/sediment samples can be challenging due to wetland soil compaction. Thus, making bulk density measurements precise enough to capture changes in carbon stocks over time is difficult.	While there are fewer methodological issues with measuring carbon storage, some still apply, including: soil depth (Nahlik and Fennessy 2016) and how wetlands are classified (Loder and Finkelstein 2020). Further, obtaining samples can be challenging in wetlands, especially in the presence of soil compaction.

Carbon accounting methods & terminology

One approach to estimating the net carbon flux of a study area is by measuring the change in carbon stored in different carbon pools over time. For example, incremental changes in tree biomass using allometric equations that relate biomass to diameter and height measurements can be used to estimate the change in the aboveground woody biomass pool. Similarly, the accumulation of carbon in the soil can be documented through change over time or inferred from aging soil cores. This approach represents the long-term change in carbon storage, with measurements potentially separated by years. However, it does not allow for understanding how the rates of change may vary over time or in response to specific environmental conditions, and can be difficult to apply over short time frames or to estimate small changes in carbon storage (Loder and Finkelstein 2020).

A second method is through the use of leaf cuvettes or plant and soil chambers (Baldocchi 2003), which measure fluxes between very specific carbon pools. These chambers are typically small, measuring fluxes on a small spatial footprint, which are then scaled up to make landscape-level estimates. They are useful for determining the mechanisms behind fluxes and can measure variation in fluxes over fine spatial and temporal scales to identify “hot spots” and "hot moments". However, this approach can introduce bias in the estimates if the sampling does not represent this spatial and temporal variation accurately, such as sampling only a small spatial area or a short time period. Measuring fluxes throughout the year can be particularly important given that microbial respiration occurs year-round (Baldocchi and Penuelas 2019); if measurements are only taken during a “growing season”, when photosynthesis is high, and respiration in the winter or dry season is neglected, the net difference between photosynthesis and respiration will be biased toward overestimating carbon storage.

Since the late 1990s and early 2000s, eddy covariance techniques have emerged as a third approach to assessing ecosystem carbon exchange between the atmosphere and the canopy of the vegetation below (Baldocchi 2003). Eddy covariance works by continuously measuring (every second across years) the difference between carbon dioxide concentrations in air moving upward versus downward. The resulting measurements are similar to a plant and soil chamber, but with a whole-canopy footprint estimate of gas emissions. Eddy covariance works best over flat terrain, under stable environmental conditions, and where the underlying vegetation extends upward for an extended distance. A drawback of eddy covariance methods is that they do not measure highly resolved fluxes between smaller carbon pools (e.g. “hot spots” of carbon flux within larger footprints).

In the sections below, we address both estimates of carbon storage and carbon fluxes in wetlands, synthesized from all types of measurements. To focus on long-term, large-scale carbon benefits of wetlands, we did not summarize estimates of carbon flux rates over short time frames or between fine-resolution carbon pools (e.g. roots and soil). Instead, we focused on annual estimates of carbon accumulation or flux rates between atmospheric carbon and soil or biomass carbon, and we sought to ensure that measurements were directly comparable. In addition, studies typically reported only one of several similar metrics used by atmospheric chemists and ecosystem ecologists to describe carbon fluxes, with slightly different meanings: net ecosystem productivity (often abbreviated NEP), net ecosystem carbon balance (NECB), and net ecosystem exchange (NEE). Chapin et al. (2006) defined and clarified the differences among these terms (Table 7.2), and we were careful to keep these values separate so that comparisons can be made across studies and wetland classes.

Table 7.2. Carbon flux terminology and definitions.

Key Term	Abbreviation	Definition
Net ecosystem productivity	NEP	Gross primary production (carbon assimilation) minus ecosystem respiration. When production outpaces respiration, carbon is added to ecosystems and the sign of NEP is positive. Approaches NECB when primary production and ecosystem respiration are the dominant fluxes and there is no lateral carbon transport.
Net ecosystem carbon balance	NECB	The net rate of carbon accumulation in specific ecosystems (or local footprints) accounting for all sources and sinks, including net ecosystem exchange of CO_2, methane emissions, and lateral movement of soil or biomass, such as through erosion or harvest.
Net ecosystem exchange	NEE	The net CO_2 flux from the ecosystem to the atmosphere, where the input of CO_2 to the ecosystem (and loss from the atmosphere) has a negative sign. Approaches -NECB when CO_2 is the dominant flux, and over short time scales, NEE is approximately equal to -NEP.

7.2 Carbon Storage

Wetlands accumulate **soil carbon** where organic matter is deposited, potentially buried by additional deposits of organic matter and sediments, and not rapidly decomposed, due to saturated soils that lack oxygen and suppress microbial decomposition. Both the concentration of the carbon in soil as well as the depth of this organic-rich layer contribute to the soil carbon storage of a particular wetland site. The organic matter may arrive in the form of roots, leaf litter, or other plant tissues that grew on site, or it may be transported and deposited in a wetland by water movement. Thus, soil carbon is influenced by local vegetation growth, hydrological connectivity, and spatial configuration relative to other sources of organic matter. In addition to carbon in the soils, wetlands may have substantial amounts of **biomass carbon** stored in the plants growing on site, particularly the long-lived woody biomass of trees. Biomass carbon estimates may or may not include the carbon stored in roots, and thus may also be referred to as **aboveground carbon**. Similarly, estimates of soil carbon may or may not include roots, and may also be referred to as **belowground carbon**.

In this section, we synthesized estimates of carbon storage from the literature in our rapid evidence assessment, focusing on carbon stored in the soil (Figure 7.2) and biomass (Figure 7.3). We reviewed 22 papers explicitly about coastal wetlands, 11 papers on freshwater wetlands, six papers on riparian systems, 12 papers on montane meadows, six papers on beaver wetlands, and another 21 papers that included a comparison across wetland classes. These papers included both carbon storage and emissions data in disturbed and natural systems, with most papers focused only on natural systems. Some papers focused on individual sites (all the montane meadow papers) whereas some papers represented meta-analyses and syntheses over broad geographic areas. Thus, individual papers contained different amounts of evidence. In each section below we provide a relative ranking of the amount of evidence we reviewed for each wetland class (low, medium, or high), reflecting the relative confidence we have in the overall assessment. While this ranking is subjective, the underlying data are available for further examination (Conlisk et al. 2022).

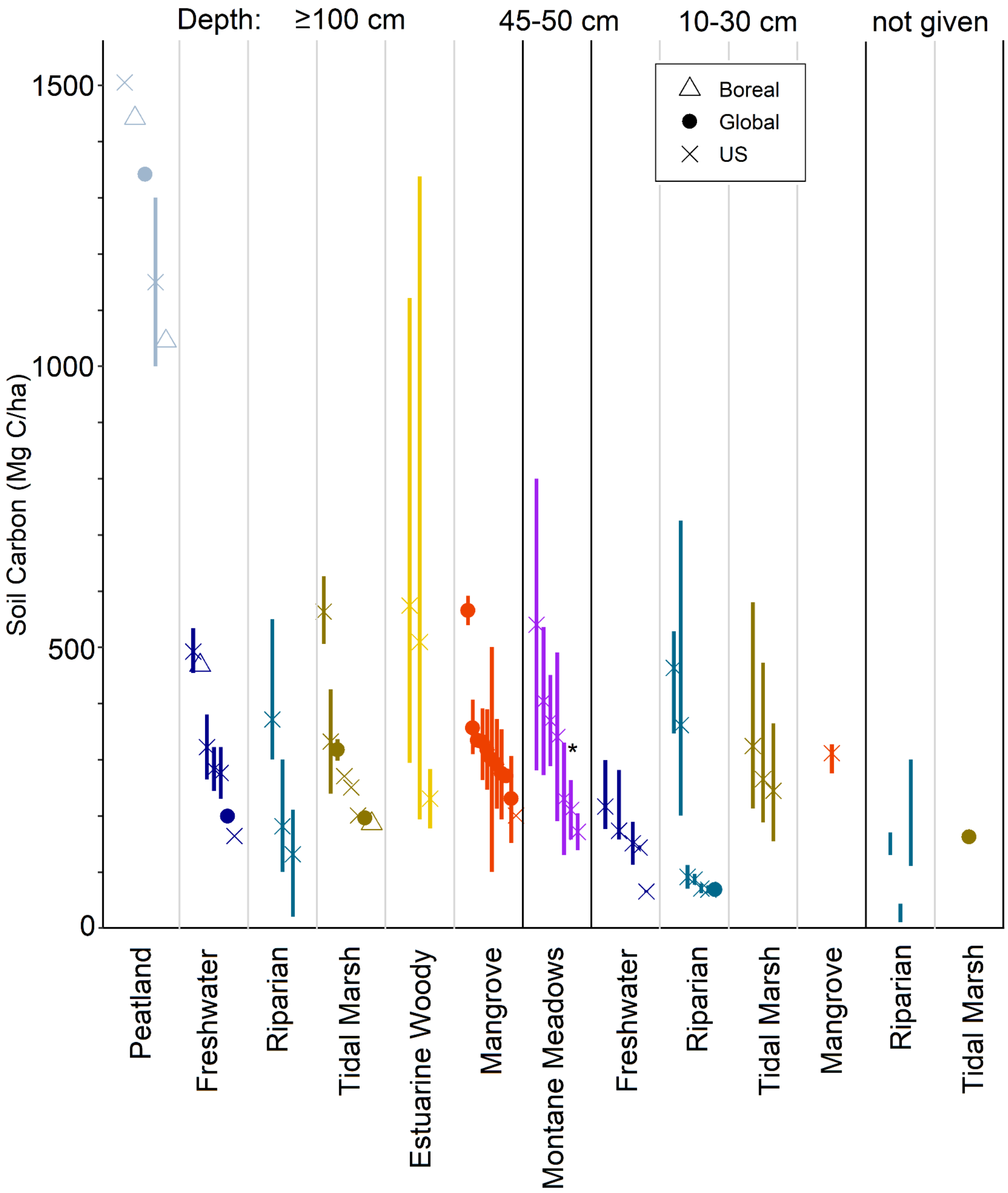

Figure 7.2. Soil carbon storage estimates by wetland class. Each point represents a reviewed study's median or mean value (whichever was reported) of soil carbon in metric tons per hectare (Mg/ha), with point shape indicating the geographic scope of the study as shown in the legend. Values are grouped by soil core depth: greater than or equal to 100 cm, between 45-50 cm (with the one exception under the * that was a study conducted at 75 cm), 10-30 cm, and not given. Bars represent the variation in the estimate, typically the interquartile range, standard deviation, or, less commonly, the minimum and maximum. Where there is no point, there was no median or mean given, only a range; where there is no bar, there was no range given. Values of individual points come primarily from global meta-analyses with a few points coming from studies across small spatial extents. Wetland classification was based on information given in the individual studies, with some combining of like habitat (e.g. "salt marsh" and "tidal marsh" were combined; see Conlisk et al. 2022 for additional information for individual studies). Some studies identified "woody" and "shrub" habitats without specifying the constituent woody species (e.g. not all woody estuarine systems are mangroves), thus we retained the classification "woody" for these studies.

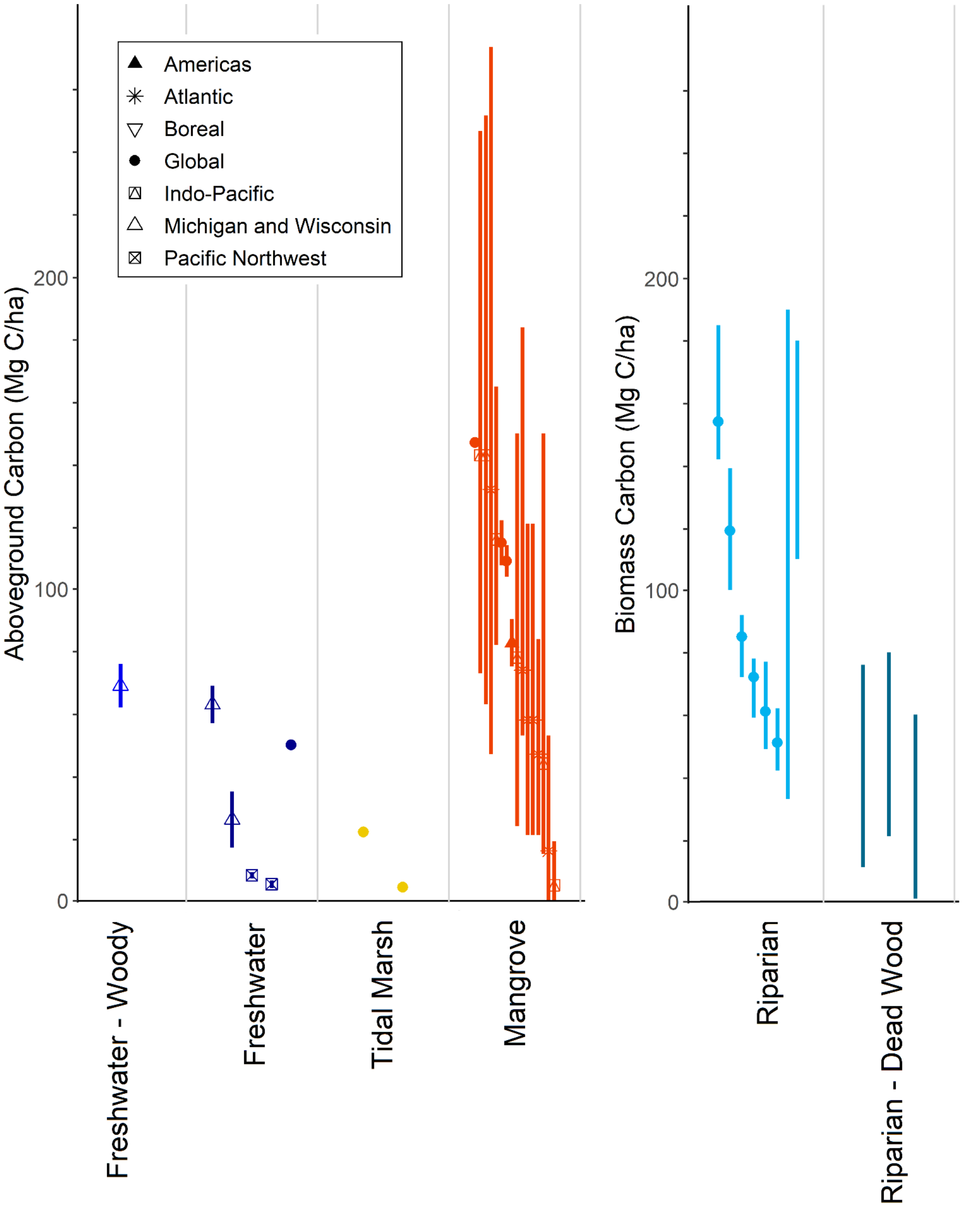

Figure 7.3. Biomass carbon storage estimates by wetland class, grouped into estimates for aboveground (left) and total biomass (right, including aboveground and roots). Each point represents a reviewed study's median or mean value (whichever was reported) in metric tons per hectare (Mg/ha), with point shape indicating the geographic scope of the study as shown in the legend. Bars represent the variation in the estimate, typically the interquartile range, standard deviation, or less commonly, the minimum and maximum. Where there is no point, there was no median or mean given, only a range; where there is no bar, there was no range given. Wetland classification was based on information given in the individual studies, with some combining of like habitat (e.g. "salt marsh" and "tidal marsh" were combined; see Conlisk et al. 2022 for additional information for individual studies). Some studies identified "woody" wetlands without additional detail about the wetland class (e.g. not all freshwater woody wetlands are riparian), thus we retained the classification "woody" for these studies.

Based on this rapid evidence assessment, we summarized the general patterns of carbon storage within each wetland class and relative to the other classes, to facilitate comparisons and potentially inform wetland restoration and conservation priorities:

- **Peatlands had the highest estimates of soil carbon storage** (Figure 7.2). As a cross-cutting subclass of wetlands defined by having at least 30% organic material in the soils, high soil carbon storage in peatlands is expected regardless of wetland class. Conservation of existing peatlands is highly advisable to prevent the loss and emissions of these carbon stocks.

- **Coastal wetlands generally had the highest soil carbon storage after peatlands.** High soil carbon storage in coastal wetlands (including mangroves, estuarine woody wetlands, and tidal marsh) is well-described in the literature and supported by our evidence assessment, particularly for estuarine woody wetlands (Figure 7.2). Mangroves have the additional benefit of substantial carbon storage aboveground in their trees (Figure 7.3).

- **Freshwater wetlands are a heterogenous group with high variability in soil carbon storage.** Some freshwater wetlands have comparable storage to coastal systems while other wetlands have much lower values. On average, the freshwater wetland papers we reviewed showed lower aboveground carbon storage than mangroves and riparian wetlands.

- **Riparian wetlands had variable soil carbon storage and high aboveground carbon storage.** The high variability in soil carbon depends on hydrological connectivity to sources of organic matter, and the geometry and fluvial setting among the channel, bank, and floodplain. Riparian wetlands can store similar amounts of aboveground carbon as mangroves (Figure 7.3).

- **Montane meadows are less well-documented than other wetland classes.** Soil carbon storage is highly dependent on seasonal hydrology and snowpack affecting rates of soil microbial respiration.

Finally, we found that **conservation of all wetland classes should be a high priority to protect existing carbon stocks.** Wetlands of all classes store large amounts of carbon in their soils and biomass, with substantial overlap across wetland classes and high variability among sites within each wetland class. Thus, other than perhaps the peatlands within each wetland class, no individual wetland class clearly stores more carbon than the others. Given the ongoing threats to existing wetlands, and the high risk of loss of these carbon stocks if wetlands are disturbed (see section 7.4), conservation of all wetland classes is advisable.

Peatlands: cross-cutting wetland subclass with the most soil carbon, by definition

Across centuries and millennia, peatlands across the globe have accumulated high densities of soil carbon (by definition greater than 30% dry mass of dead organic material). High carbon density makes peatlands especially attractive for conversion to agriculture or excavation for fuel, where such disturbances result in large carbon losses. In this synthesis, we found few soil carbon estimates from freshwater carbon studies and meta-analyses (Figure 7.2). Specifically, Bridgham et al. (2006) reported between 1045 and 1142 Mg C/ha in boreal permafrost and non-permafrost peatlands, respectively (Figure 7.2). In the North Highlands Lake District of Michigan and Wisconsin, Buffam et al. (2011) found 1150 Mg C/ha. Scaling up the soil carbon storage estimates to the continental U.S., Bridgham et al. (2006) estimated a total of 14 Pg C stored in peatlands. Relative to the estimated 69-75 Pg of carbon stored in the top meter of soil in the continental U.S. (Gonçalves et al. 2021), we estimated peatlands represent 18-20% despite occupying only 1.2% of the land area (Bridgham et al. 2006). We do not include any estimates of biomass carbon for peatlands (Figure 7.3) because they occur across all wetland classes with a wide range of vegetation types and thus highly variable biomass carbon stocks. Overall, we have **moderate confidence** in peatland soil carbon storage estimates given that we did not review papers that explicitly compared peatlands across different geographic and climatic regions. However, we have high confidence in the relative differences between peatlands and non-peatland wetlands.

Coastal wetlands: the highest soil carbon storage outside peatlands

Excluding peatlands, soil carbon stocks were generally higher in coastal wetlands (including tidal marsh, mangroves, and estuarine woody wetlands) compared to the other wetland classes we reviewed (Figure 7.2). However, there was substantial overlap across wetland classes. Mangroves also had substantial amounts of carbon stored aboveground in woody biomass, comparable to riverine wetlands (Figure 7.3), though these values were generally smaller than the soil carbon stocks. While variability in soil carbon stocks was a function of region, depth, vegetation, salinity, latitude, and precipitation, none of these predictors was highly influential on its own given the inherent variability in carbon storage across sites. Variability in carbon storage estimates in coastal systems was generally smaller compared to other systems reviewed, with the possible exception of high variability in estuarine woody systems.

In mangroves, soil carbon stock estimates ranged from a low of 200 Mg C/ha (Bridgham et al. 2006) to a global mean estimate of 565 +/- 26 Mg C/ha (median: 500.5 Mg/ha) in the first meter of soil (Alongi 2020). Estimates of tidal marsh soil carbon storage ranged from 162 Mg C/ha (Poulter et al. 2021, no range provided and no depth given) to 563 Mg C/ha in the first meter of soil (Hinson et al. 2017, specifically calling the ecosystem “estuarine emergents”). While the tidal marsh and mangrove studies reviewed here provided a similar range of carbon storage values (Figure 7.2), soil organic carbon is generally viewed as higher in tidal forested and mangrove systems compared to marshes (Hinson et al. 2019; Alongi 2020; Kauffman et al. 2020a).

Projecting the total amount of carbon stored in mangrove soils worldwide, Sanderman et al. (2018) estimated 6.4 Pg in the top meter of soil, representing roughly 0.26%–0.64% of the global soil organic carbon pool despite being only 0.11% of the land surface. While comparable global estimates of tidal marsh land area are underdeveloped, tidal wetlands comprise approximately 0.3% of the total area in the continental U.S., but their soils constitute up to

1.5-1.9% of soil carbon, or 1.15-1.36 Pg (Hinson et al. 2017) out of a total of 69-75 Pg C in all U.S. soils (Gonçalves et al. 2021), similar to the estimate of 57-73 Pg C in Lajtha et al. (2018).

Soil carbon storage in coastal wetlands varies regionally, as well as with depth, vegetation, salinity, latitude, and precipitation (Table 7.1). Regional variations have been identified across the U.S. (Nahlik and Fennessy 2016; Hinson et al. 2019), with the highest soil carbon densities in the Atchafalaya-Vermillion Bay complex in Louisiana followed by the Chesapeake Bay region (Hinson et al. 2017). By depth, nearly a quarter of total soil carbon (measured to 1.2 m) in estuarine emergent ecosystems and almost 10% in estuarine woody systems occurred in soil deeper than 90 cm (Nahlik and Fennessy 2016). In mangroves, lower soil carbon storage was found in those dominated by *Avicennia*, those with reduced precipitation, and those at higher latitudes (Kauffman et al. 2020a). Other studies similarly identified lower carbon storage in tidal wetlands with reduced precipitation (Hinson et al. 2019), and in salt marshes at higher latitudes (Ouyang and Lee 2020). Temperature was also a factor across the continental U.S., with lower soil carbon storage associated with higher temperatures in the Gulf of Mexico and west coast (Hinson et al. 2019).

In addition to soil carbon, aboveground carbon storage in mangroves ranged from 82 Mg C/ha (Kauffman et al. 2020a) to 147 Mg C/ha (Villa and Bernal 2018) (Figure 7.3). Consistent with their lower stature, salt marsh aboveground carbon storage was smaller, ranging from 4.3 +/-0.10 Mg C/ha (Alongi 2020) to 22 Mg C/ha (Villa and Bernal 2018, no range provided). Measuring total ecosystem carbon storage in the Pacific Northwest (from Northern California to Puget Sound), carbon storage increased along a transect from low marsh (low elevation/high salinity) to tidal forest (high elevation/low salinity) (Kauffman et al. 2020b); tidal forests had significantly more carbon aboveground (range: 74-395, mean: 220 Mg C/ha) compared to seagrass and marsh communities (<8.2 Mg/ha), resulting in soil carbon making up >98% of total ecosystem carbon in the seagrass and marsh communities and 78% in the tidal forest.

We have **high confidence** in the coastal wetlands carbon storage estimates. Coastal wetlands were well-represented in the papers we reviewed, both in the total number of papers and in the fact that many (roughly half) of these papers were syntheses, meta-analyses, or review papers.

Freshwater wetlands: high variability across ecosystems

Of the freshwater wetland papers we reviewed, we found substantial soil carbon storage in freshwater wetlands, but generally not as high as coastal wetlands (Figure 7.2). Part of the difficulty in summarizing freshwater wetlands was the variation in freshwater wetland classification. Wetlands could be categorized, for example, by latitude (e.g. tropical, temperate, boreal), vegetation (e.g. herbaceous, shrub, or forest), or region (e.g. Prairie Pothole Region), and not clearly fall into one of our classes. For example, estimates of soil carbon storage in freshwater wetlands with emergent vegetation and mineral soils may include freshwater depressional, riverine, and montane meadow wetlands. Similarly, aboveground carbon was highly variable and difficult to summarize because of the variability in vegetation types that might occur in a given wetland (Figure 7.3). Overall, consistent flooding and duration of flooding led to higher soil carbon storage and was the most important covariate for describing variability in soil carbon (Amendola et al. 2018; Tangen and Bansal 2020).

Soil carbon stock estimates in freshwater wetlands ranged from 163 Mg C/ha for the conterminous U.S., to 467 Mg C/ha in Alaska (Bridgham et al. 2006), and 492 Mg C/ha (interquartile range: 200-725) in soils within freshwater wetlands near the U.S. coast (Hinson et al. 2017). Soils in freshwater forested wetlands (which included palustrine, lacustrine, and riverine forested wetlands) contained 283 Mg C/ha (Nahlik and Fennessy 2016). In Figure 7.2, we presented only two estimates specific to freshwater depressional wetlands, with 151 Mg C/ha in Pennsylvania (Mazurczyk and Brooks 2018) and 65 Mg C/ha from the Prairie Potholes Region (Tangen and Bansal 2020). Additional estimates of soil carbon stocks in the Prairie Potholes ranged 140–180 Mg C/ha across 59 seasonal and 60 semi-permanent wetlands (Tangen et al. 2015), but these values were not included in Figure 7.2 because they included estimates from disturbed wetlands. Scaling up the per-ha estimates of soil carbon storage to wetlands across the entire continental U.S. yields an estimated 5.1 Pg C or 6.6%–8.4% of the total soil carbon stored in only 3.9% of the land area (Bridgham et al. 2006).

Soil carbon storage in freshwater wetlands is sensitive to periodic drying, where consistently inundated areas have higher carbon storage (Amendola et al. 2018; Tangen and Bansal 2020). In addition, soil carbon storage varied with other soil properties, having a positive correlation with aluminum and iron content and a negative correlation with pH and clay content, but little relationship with temperature (Amendola et al. 2018).

Aboveground carbon was variable in freshwater wetlands due to variation in vegetation composition (Figure 7.3). Excluding riparian wetlands, treated separately in the next section, the highest reported value was 69 +/- 7 Mg C/ha, described as from a freshwater woody system (Buffam et al. 2011).

We have **moderate-high confidence** in the freshwater wetlands carbon storage estimates synthesized here. Freshwater wetlands were well-represented in the papers we reviewed, and roughly a quarter of which were review or synthesis papers, but the lack of clarity on the exact types of freshwater wetlands included in some papers presents some uncertainty.

Riparian wetlands: variable soil and high aboveground carbon storage

Although incorporated into some of the estimates for freshwater wetlands described above, we found several papers specific to riparian wetlands and synthesized these separately. Riparian wetlands generally had lower soil carbon content than the other wetland classes we reviewed, but also the highest variability across studies as measured by the coefficient of variation (standard deviation/mean). This high variability depends on hydrology, geometry, and fluvial setting among the channel, bank, and floodplain (Matzek et al. 2020). In active channels, organic matter may be transported to floodplains and estuaries downstream instead of stored locally, while in meandering, wide floodplains with many obstructions to slow floodwaters, organic matter can be readily trapped and buried in floodplain soils. Thus, we might expect higher soil carbon in floodplains than channels. However, the frequency and duration of floodplain inundation, as well as the quantity of organic matter transported, will influence how much carbon is stored in riverine floodplains. In addition to soil carbon, riverine wetlands often support woody vegetation and thus substantial carbon storage in aboveground biomass, comparable in magnitude to mangroves and varying with forest age and climate.

At the lower end of soil carbon estimates in riverine wetlands, Mazurczyk and Brooks (2018) reported a range of 68.25–90.8 Mg C/ha in soil up to 20 cm deep in Pennsylvania, varying among upper perennial wetlands (within or near a channel with continuous flow) and headwater complexes (a mosaic of small streams, depressions and slopes with groundwater or intermittent surface flows), respectively. At the upper end of estimates, a global review of floodplains and riparian ecosystems found a wide range of values from 50 to 3000 Mg C/ha, with most falling in the 110-300 Mg C/ha range (Sutfin et al. 2016).

Carbon stored in standing biomass in riparian ecosystems ranged as high as 318-487 Mg C/ha for mature or old growth riparian forests, but with a global median value of 63 Mg C/ha (Dybala et al. 2019a). Accounting for forest age, climate, and restoration status, average values projected for a mature riparian forest ranged from 51 to 158 Mg C/ha (with confidence intervals ranging 42-185 Mg C/ha), with the highest values for old-growth forest in relatively warm and wet climates and the lowest values for relatively dry climates (Dybala et al. 2019a). These values are comparable to the range of 33-190 Mg C/ha in standing biomass from an earlier review (Sutfin et al. 2016), which also estimated another 21-80 Mg C/ha typically stored in large downed wood. These estimates place biomass carbon stocks for riparian forests on par with IPCC estimates for tropical forests (120 Mg C/ha), boreal forests (64 Mg C/ha), and temperate forests (44 Mg C/ha), and more than IPCC estimates for wetlands in general (29 Mg C/ha) or other non-forested biomes (Dybala et al. 2019a).

Summing across soil and biomass carbon pools, total riparian ecosystem carbon storage was estimated as averaging 202–386 Mg C/ha (Sutfin et al. 2016). Globally, riverine ecosystems could store as much as 16–125 Pg C, with 12–80 Pg C in the soil (Sutfin et al. 2016) and 13–31 Pg C in the standing biomass (Dybala et al. 2019a). Although there is considerable uncertainty in scaling these numbers globally, riparian ecosystems could account for 0.5 to 8% of global soil organic carbon storage, even though they cover only 0.5–1% of the land surface (Sutfin et al. 2016).

We have **moderate confidence** in the carbon storage estimates for riparian wetlands. While we did not review many papers specifically on riverine wetlands (n = 6), two were comprehensive review and synthesis papers.

Montane meadows: soil carbon storage uncertain and likely depends on hydrology

Montane meadows had some of the highest soil carbon measurements, however, the studies we reviewed were usually restricted to small spatial extents (i.e. single meadows) within the Sierra Nevada of California. Thus, without more evidence across montane ecosystems, the data we reviewed may not be broadly representative. Whether they are depressional or riverine meadows, seasonal hydrology and soil moisture driven by snowmelt are important factors affecting soil carbon storage in montane meadows. The duration of flooding influences if and when soils become dry enough that microbial respiration increases and soil carbon is lost, or whether soil carbon continues to be stored under floodwater and later snowpack. During the winter snow season, microbial respiration proceeds at very low levels relative to the warmer growing season.

Within montane meadows, soil carbon averaged 210 Mg C/ha for 13 meadows (Reed et al. 2021). Studies that examined variation with hydrology suggested that drier meadows stored less carbon (228–261 Mg C/ha) compared to wetter meadows (540 Mg C/ha) (Norton et al. 2014). Overall, we did not find an estimate of total carbon storage across the continental U.S. or globally that was specific to montane meadows.

We have relatively **low confidence** in the carbon storage estimates we report for montane meadows. Estimates for soil carbon storage in montane meadows were underrepresented in our initial literature search (Appendix A), and we intentionally included more of them in our list of high-priority papers to review, increasing the number of papers to 12. Still, most of these papers had restricted scope, with no reviews or meta-analyses found.

7.3 Carbon Fluxes

Accounting for emissions of carbon dioxide (CO_2) and methane (CH_4) from wetlands is important to understanding the net carbon benefits. As described above (see section 7.1), methane is a more potent greenhouse gas (GHG) than CO_2, such that the benefits of sequestering and storing carbon in wetland soils or biomass could be reduced or eliminated by methane emissions. However, because of its relatively short life-span in the atmosphere, the amount of methane emitted by wetlands can quickly (within decades) reach equilibrium, meaning that the rate of emissions matches the rate of degradation, at which point there is no net contribution to heating of the atmosphere. In contrast, CO_2 has a much longer life-span in the atmosphere, such that reaching equilibrium requires much longer time frames (millennia). Therefore over the long-term, steady incremental removal of CO_2 from the atmosphere, such as through sequestration in wetland soils and biomass, will continue to contribute to cooling for centuries, whereas the warming effect of steady incremental additions of methane will plateau within about 50 years. Thus, the net warming or cooling influence of an individual wetland in a given year depends on how long the wetland has been emitting methane and removing CO_2 from the atmosphere, and the ratio between the rates of these two fluxes, leading to non-linear relationships (Figure 7.4, reproduced from Taillardat et al. 2020).

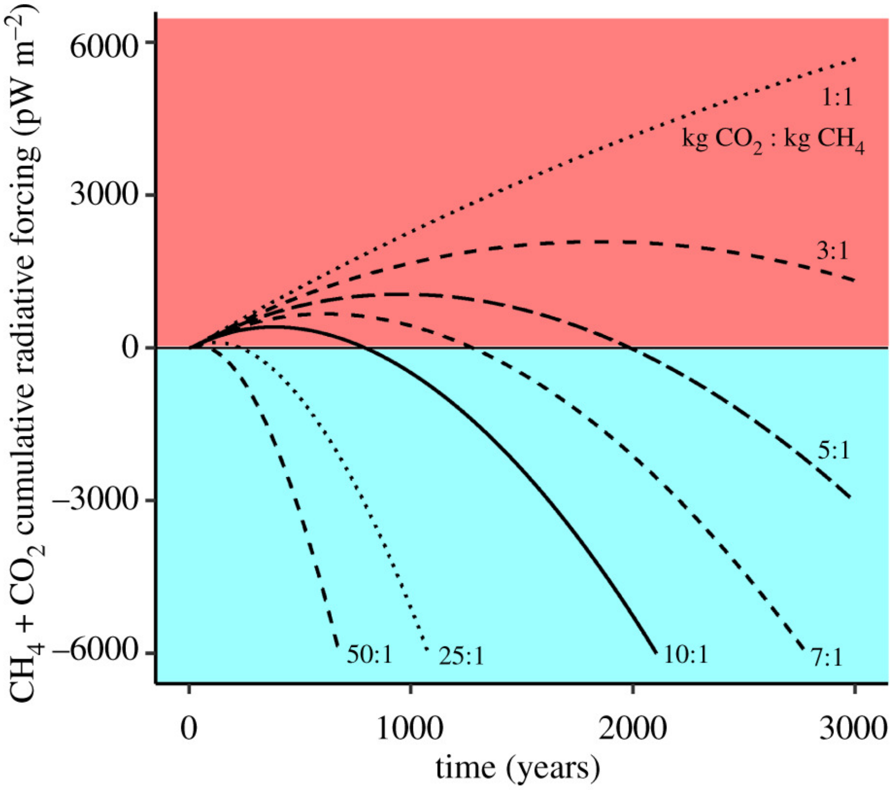

Figure 7.4. Estimated time to net cooling effect in wetlands. For varying ratios of methane emissions to net CO_2 uptake, wetlands initially have a net positive warming effect on the atmosphere (red background), but CO_2 uptake eventually offsets the methane emissions, resulting in a net negative warming effect (i.e., cooling effect; blue background). Figure reproduced from Taillardat et al. 2020, which was adapted from Neubauer and Verhoeven (2019).

Although few studies explicitly report the ratio of these two rates, they are useful for estimating a wetland's switchover time, or the age at which a wetland has sequestered enough carbon to have a net cooling effect on the atmosphere despite continued methane emissions. A comprehensive examination of switchover times across wetland classes showed a range from immediate or very short time frames to millennia (Figure 7.5). The highest median switchover times were in peatlands, with three of the 25 peatlands having switchover times greater than 3000 years. Freshwater systems also had long switchover times, with four wetlands having switchover times greater than 3000 years, and a broad range in switchover times, with one wetland having an immediate switchover time. The shortest switchover times were in coastal wetlands, which have the advantage of relatively low methane emissions. The sea water in coastal wetlands contains sulfate, allowing sulfate-reducing bacteria to outcompete the bacteria that produce methane. Thus, when sea water occurs at salinities of 10-18 ppt, typically there is enough sulfate to suppress methane emissions (Poffenbarger et al. 2011; Rosentreter et al. 2021). (We note that at very high salinities, as seen in industrial salt ponds, sulfate reducing bacteria are outcompeted by halophiles that produce large quantities of methane, Zhou et al. 2022; we do not address salt ponds in this synthesis.) However, many mangrove and salt marsh habitats occur in estuaries where salt and freshwater mix, such that when the influx of freshwater is high, there can be corresponding increases in emissions of methane; this temporal and spatial variation is likely to have been underestimated (Rosentreter et al. 2018, 2021).

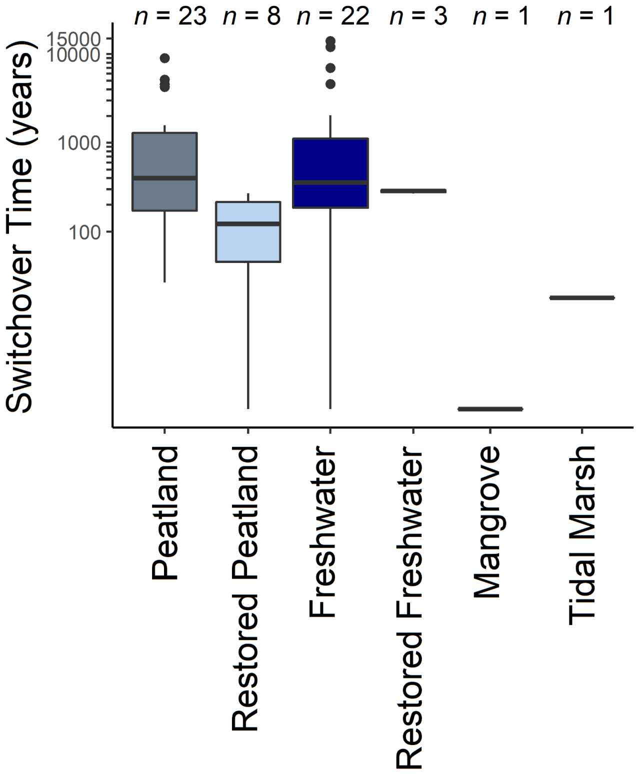

Figure 7.5. Estimated range of switchover times by wetland class. Switchover times represent the age at which a wetland changes from having a net warming to a net cooling effect. Note that locations with CO_2 emissions that exceed CO_2 uptake will never have a cooling effect and thus cannot be included in switchover time calculations. (Values from Neubauer 2014; Taillardat et al. 2020; Arias-Ortiz et al. 2021)

Because estimates of switchover times from individual wetlands were rare in the literature, we instead examined individual estimates of carbon fluxes, including rates of soil carbon sequestration (Figure 7.6), ecosystem flux (Figure 7.7), and methane emissions (Figure 7.8). We also distinguished between similar ecosystem flux values in Figure 7.7 because they represent slightly different definitions, as described above (see Table 7.2): Net Ecosystem Productivity (NEP), Net Ecosystem Carbon Balance (NECB), and Net Ecosystem Exchange (NEE). However, we also acknowledge that their values can be approximately equivalent under certain conditions. Note that we report *negative* NEE to reflect that NEP, NECB, and -NEE are all positive values when carbon is sequestered by the ecosystem.

We synthesized information on estimates of these fluxes from the studies included in our rapid evidence assessment, including 22 papers explicitly about coastal wetlands, 11 papers on freshwater wetlands, six papers on riparian systems, 12 papers on montane meadows, six papers on beaver wetlands, and another 21 papers that included a comparison across wetland classes (as described in section 7.2). Because each paper represented different amounts of evidence, we also provided a relative ranking of the amount of evidence we reviewed for each wetland class (low, medium, or high), reflecting the relative confidence we have in the overall assessment. While this ranking is subjective, the underlying data are available for further examination (Conlisk et al. 2022).

Soil carbon accumulation rates generally ranged up to ~300 g C/m^2/yr, although studies in montane meadows found high spatial and interannual variability (Figure 7.6), discussed further below. Methane emissions ranged up to ~100 g C/m^2/yr, but most estimates were less than 40 g C/m^2/yr (Figure 7.8). Carbon flux into the ecosystem (-NEE, NECB, or NEP) rates were generally positive for wetlands (meaning net addition of carbon to the ecosystem), and ranged over 1000 g C/m^2/yr. In contrast, drained wetlands and agricultural sites showed negative values (i.e., carbon was released to the atmosphere) ranging well below -1000 g C m^2/yr (Figure 7.7).

Although we did not attempt to assess the literature for carbon fluxes in agriculture, we reported values presented in studies that compared them to wetlands, and in Figure 7.7 these included values from rice, pasture, corn, aquaculture, and unspecified cropland (including three values from Tan et al. 2020, and one from Webb et al. 2019 that mixed pasture and sugarcane). Most agricultural sites emitted carbon into the atmosphere except for two rice studies (Knox et al. 2015; Petrescu et al. 2015), one pasture, one unspecified cropland, and one aquaculture site (Tan et al. 2020). Methane emissions from agricultural sites in Figure 7.8 came from rice, pasture, aquaculture and an unspecified cropland type (the three unspecified values are from Tan et al. 2020).

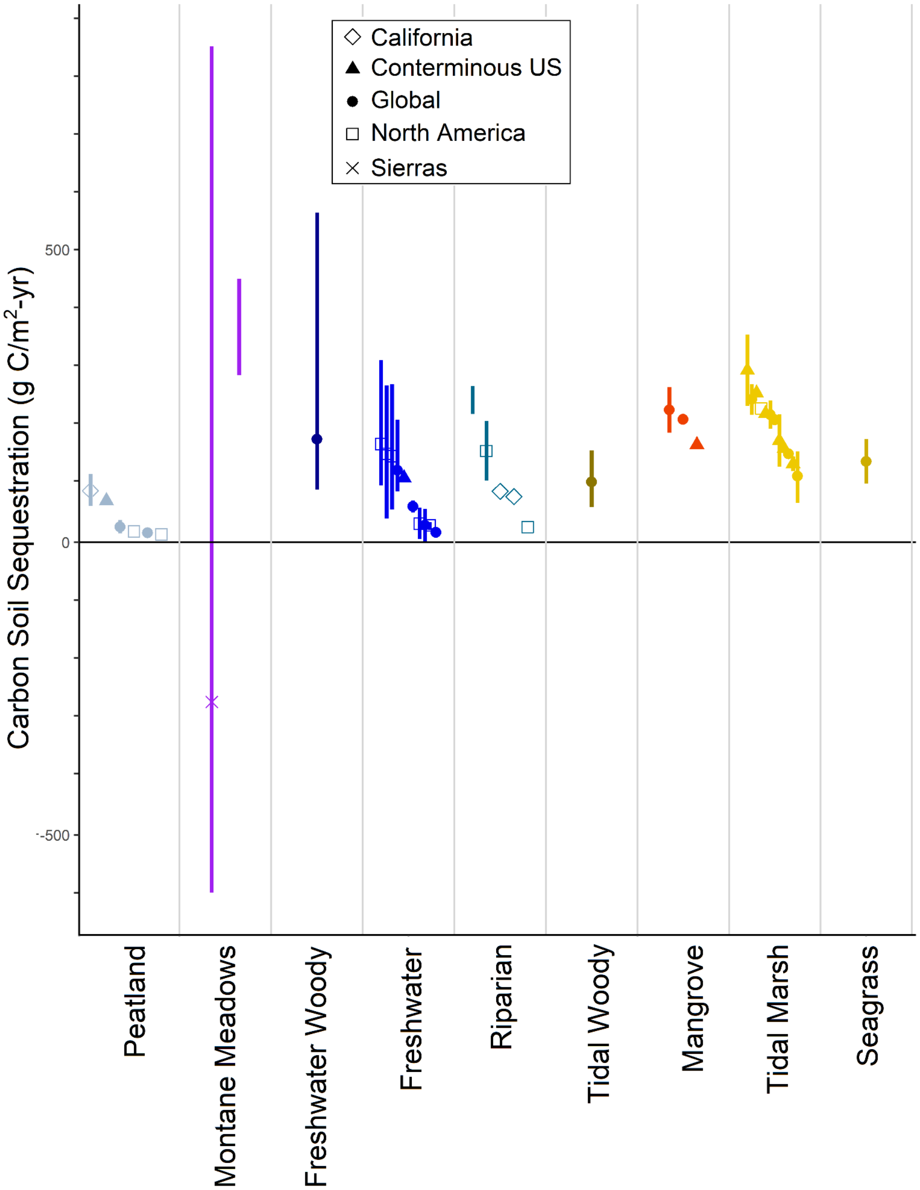

Figure 7.6. Soil carbon flux estimates by wetland class. Each point represents a reviewed study's median or mean value (whichever was reported) of soil carbon sequestration rates or fluxes in grams per m^2 per year, with point shape indicating the geographic scope of the study as shown in the legend. Positive values indicate carbon was added to the soil, and negative values indicate carbon was lost from the soil. Bars represent the variation in the estimates, typically the interquartile range, standard deviation, or, less commonly, the minimum and maximum. Where there is no point, there was no median or mean given, only a range; where there is no bar, there was no range given. Wetland classification was based on information given in the individual studies, with some combining of like habitat (e.g. "salt marsh" and "tidal marsh" were combined; see Appendix C for additional information for individual studies). Some studies identified "woody" and "shrub" habitats without specifying the constituent woody species (e.g. not all woody estuarine systems are mangroves), thus we retained the classification "woody" for these studies.

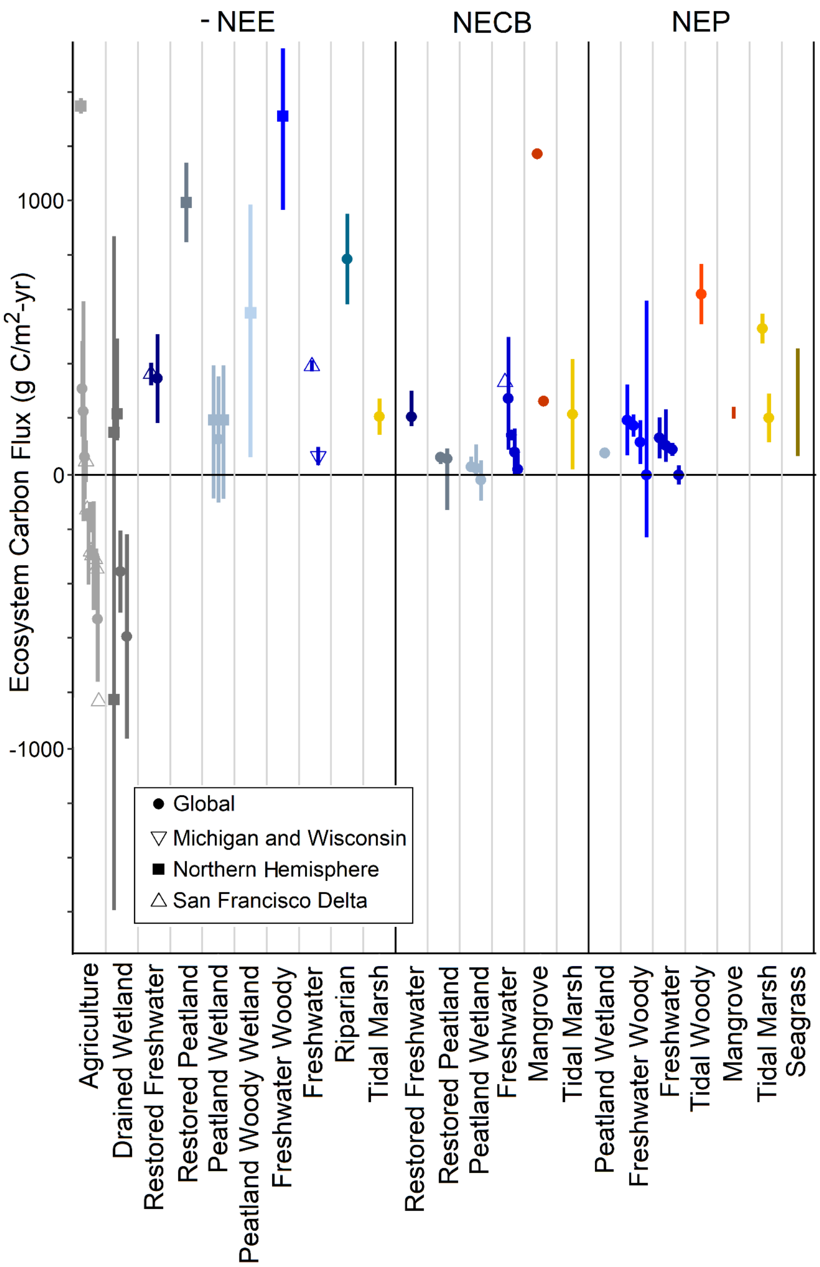

Figure 7.7. Ecosystem carbon flux estimates by wetland class. Each point represents a reviewed study's median or mean value (whichever was reported) of ecosystem carbon flux (-NEE, NECB, or NEP) in grams per m^2 per year, with point shape indicating the geographic scope of the study as shown in the legend. Positive values indicate carbon was added to the ecosystem, and negative values indicate carbon was lost from the ecosystem. Bars represent the variation in the estimate, typically the interquartile range, standard deviation, or, less commonly, the minimum and maximum. Where there is no point, there was no median or mean given, only a range; where there is no bar, there was no range given. Wetland classification was based on information given in the individual studies, with some combining of like habitat (e.g. "salt marsh" and "tidal marsh" were combined; see Conlisk et al. 2022 for additional information for individual studies). Some studies identified "woody" and "shrub" habitats without specifying the constituent woody species (e.g. not all woody estuarine systems are mangroves), thus we retained the classification "woody" for these studies.

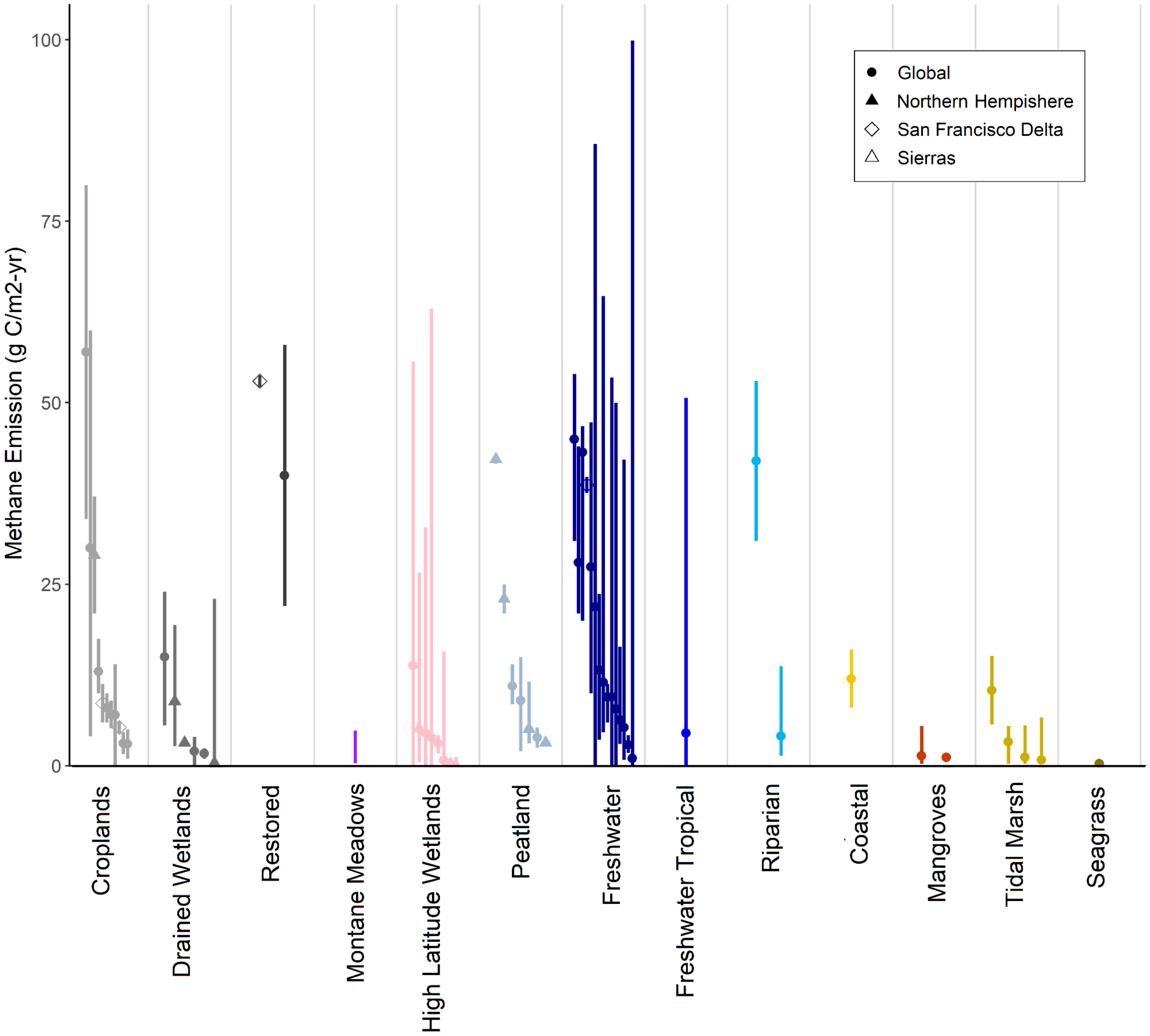

Figure 7.8. Estimates of methane emissions by wetland class. Each point represents a reviewed study's median or mean value (whichever was reported) of methane emissions in grams per m^2 per year, with point shape indicating the geographic scope of the study as shown in the legend. Bars represent the variation in the estimate, typically the interquartile range, standard deviation, or, less commonly, the minimum and maximum. Where there is no point, there was no median or mean given, only a range; where there is no bar, there was no range given. Wetland classification was based on information given in the individual studies, with some combining of like habitat (e.g. "salt marsh" and "tidal marsh" were combined; see Appendix C for additional information for individual studies).

Based on this rapid evidence assessment, we summarized the general patterns of carbon fluxes within each wetland class and relative to the other classes, to facilitate comparisons and potentially inform wetland restoration and conservation priorities:

- **Peatlands had relatively low rates of sequestration and high methane emissions.** Despite the very high amounts of soil carbon storage (Figure 7.2), the relatively slow rates of sequestration (Figure 7.6) indicate the long time frames required to build up these carbon stocks, and contribute to some of the longest estimates of switchover time (Figure 7.5). Any carbon emitted as a result of anthropogenic disturbance in peatlands will take a long time to recover.

- **Coastal wetlands are a particularly strong carbon sink, with high rates of sequestration and few methane emissions.** The high rates of soil carbon sequestration, particularly in mangroves and tidal marsh wetlands were consistent with high rates of soil carbon storage (Figure 7.2). Together with some of the lowest rates of methane emissions (Figure 7.8), coastal wetlands also had among the highest rates of ecosystem flux (Figure 7.7). Conservation *and* restoration of coastal wetlands will provide warming mitigation benefits almost immediately with very short switchover times (Figure 7.5). Further, keeping tidal inflow into freshwater wetlands can reduce methane emissions in estuaries.

- **Freshwater wetlands have high soil carbon sequestration, ecosystem flux, and methane emissions.** Freshwater wetlands have longer switchover times than coastal wetlands (Figure 7.5), but many are likely old enough to be providing net carbon benefits despite ongoing methane emissions (Taillardat et al. 2020). While restoration of freshwater wetlands will provide net carbon benefits over the long-term, along with numerous other benefits (Table 3), they are also likely to produce non-trivial methane emissions.

- **Riparian wetlands have high soil and biomass carbon sequestration rates, but few estimates of methane emissions.** Like other freshwater wetlands, riparian wetlands provide substantial benefits including soil (Figure 7.6) and biomass carbon sequestration, but also emit methane at rates that may be highly variable spatially and temporally. Thus, restoration of riparian wetlands is also likely to produce non-trivial methane emissions.

- **Montane meadows have seasonal emissions profiles that warrant further study**. Most of the studies on montane meadows came from papers that only measured carbon emissions in the summer growing season and only over a small spatial footprint, not accounting for fluxes that occur over the winter. High seasonal and inter-annual variability limit inference about carbon sequestration rates, but montane meadows may not be a significant source of methane.

Finally, we emphasize that because of methane emissions, **it may take decades, centuries, or millennia to see carbon storage benefits of freshwater wetland restoration.** Methane is a more potent GHG than CO_2 and when wetlands are restored, methane emissions increase. However, the net cooling effect of continued accumulation of carbon in freshwater wetlands eventually counteracts the net warming effect of methane over decades, centuries, or millenia, depending on the relative rates of CO_2 sequestration versus methane emission.

Peatlands: fluxes uncertain, but may have relatively low carbon sequestration and high methane emissions

Peatlands had some of the longest switchover times of any wetland class (Figure 7.5), with two of the 25 peatlands requiring over 5000 years to have a net carbon benefit (Taillardat et al. 2020). Because peatlands by definition also have the highest soil carbon stocks, conserving wetlands of any class with peat soils should be a high priority because restoring their net carbon benefits and recovering carbon losses will take an especially long time.

Rates of carbon accumulation in peatland soils were relatively slow (Buffam et al. 2011), ranging 25 g C/m^2/yr (Taillardat et al. 2020) to 130 +/-362 g C/m^2/yr (Tan et al. 2020). In the U.S., estimates ranged from 13 g C/m^2/yr in Alaskan permafrost peatlands to 94 g C/m^2/yr in San Francisco Delta peatlands in California (Callaway et al. 2012). Tropical peat swamps were the one freshwater wetland ecosystem with net CO_2 emissions into the atmosphere (Taillardat et al. 2020).

Overall, we have **low confidence** in the peatland carbon flux estimates. None of the papers we reviewed specifically compared carbon fluxes across peatlands. Further, we did not explicitly seek out studies comparing peatlands in fresh versus brackish waters, which is likely to impact flux estimates and switchover times.

Coastal wetlands: high carbon sequestration and low methane emissions

Coastal wetlands had some of the shortest switchover times (Taillardat et al. 2020, Figure 7.5), consistent with high rates of carbon sequestration (Figure 7.6) paired with low rates of methane emissions (Figure 7.8). Because of these characteristics, coastal wetlands are a particularly strong carbon sink, and restoration of coastal wetlands should contribute to climate change mitigation over policy-relevant time scales (years to decades). Drivers of the variation in flux rates include sulfate, which suppresses methane emissions, and vegetation, latitude, and region, which all influence net ecosystem productivity and respiration.

We found high rates of soil carbon sequestration in coastal wetlands, and evidence that they can accumulate almost 20 times more carbon than temperate, tropical, or boreal forests on a per area basis (Mcleod et al. 2011). Mudflats in Alaska and the continental U.S. can sequester soil carbon at a rate of 229 and 255 g C/m^2/yr, respectively (Bridgham et al. 2006). In salt marshes, rates ranged from 113 g C/m^2/yr from a global study of 453 intertidal measurements (interquartile range: 57-187, Villa and Bernal 2018) to 294 +/- 61 C/m^2/yr in the Gulf of Mexico and Southeast coast of the U.S. (Ouyang and Lee 2014). In mangroves, rates ranged from 167 g C/m^2/yr in the conterminous U.S. (Bridgham et al. 2006) to 226 +/- 39 g C/m^2/yr estimated as the global average (Mcleod et al. 2011). Mangroves also had some of the highest rates of net ecosystem exchange (Figure 7.7), ranging from 269 g C/m^2/yr (Taillardat et al. 2020) to 1170 g C/m^2/yr (Webb et al. 2019). Of the studies we reviewed, net ecosystem exchange rates in tidal marshes ranged from 221 g C/m^2/yr (Taillardat et al. 2020) to 534 g C/m^2/yr (Lu et al. 2017).

Increasing salinity, which correlates with methanogenesis-reducing sulfate in sea water, was related to decreasing methane emissions (Kroeger et al. 2017), consistent with previous studies (Poffenbarger et al. 2011). Methane emissions recorded as part of this study were lower in coastal versus freshwater systems (Figure 7.8). When accounting for methane emissions, mangroves and saltmarshes had a median net cooling effect over a 100-year time frame (removing 775 and 20 g C/m^2/yr CO_2 equivalents, respectively) and an almost immediate switchover time from net warming to net cooling (Taillardat et al. 2020).

Additional drivers of the variation in these rates include latitude, vegetation, and region. In salt marshes, lower rates of carbon accumulation were found above 68 degrees latitude (Ouyang and Lee 2014). Similarly in mangroves, lower rates of carbon burial in sediments were found at higher latitudes (Rosentreter et al. 2018), but soil carbon accumulation was constant (~100 g C/m^2/yr, range 100-600) with no clear temperature or precipitation dependence (Feher et al. 2017). More generally across salt marsh, mangrove, mixed, and salt flat sites in Eastern North America, Feher et al. (2017) found a sigmoidal relationship between aboveground biomass and temperature, a linear relationship between productivity (in Mg C/ha/yr) and temperature, and no relationship between soil carbon density and temperature. Vegetation diversity influenced soil carbon storage rates in mangroves in Indonesia, with the highest rates in the most diverse mangroves, related to varying rates of root decomposition (Rosentreter et al. 2018).

Overall, we have **high confidence** in the coastal carbon flux estimates. We reviewed numerous papers from these systems including review papers and syntheses.

Freshwater wetlands: high carbon sequestration and high methane emissions

Although freshwater wetlands store large amounts of carbon and have relatively high rates of carbon sequestration, freshwater wetlands are also a source of globally significant methane emissions. Because of these methane emissions, freshwater wetlands have longer switchover times than coastal wetlands on average, ranging 0–2000 years with a median of 360, and may have positive global warming potential particularly within the first 100 years of their establishment (Taillardat et al. 2020, Figure 7.5). Consequently, conserving existing freshwater wetlands, especially those over 100 years old, should be a high priority to preserve their net carbon benefits despite ongoing methane emissions. Creating new freshwater wetlands as mitigation for the destruction of existing wetlands will likely take centuries or even millennia to recover their net carbon benefits. Further, while additional restoration of freshwater wetlands is advisable, given extensive losses and their long-term carbon benefits in addition to multiple other benefits (Table 3), it is important to account for their warming potential in the near-term. Carbon sequestration and methane emissions in freshwater wetlands varied as a function of wetland class and other characteristics, including hydrology, geology, sediment transport, and vegetation.

Soil carbon sequestration rates in freshwater wetlands ranged from 17 g C/m^2/yr (Bridgham et al. 2006) to 147 g C/m^2/yr (median across four depressional sites with range 56-270 g C/m^2/yr; Loder and Finkelstein 2020). Synthesizing 35 papers, Loder and Finkelstein (2020) found short-term (over the last 50–100 years) median soil accumulation in lacustrine wetlands of 168 g C/m^2/yr (range: 97-311 g C/m^2/yr) and long-term (over centuries and millennia) accumulation rates of 29 g C/m^2/yr (range: 21-34 g C/m^2/yr). Tree-dominated wetlands had relatively high median soil carbon sequestration rates with 186 g C/m^2/yr (interquartile range: 90-563 g C/m^2/yr) (Villa and Bernal 2018). In the Prairie Pothole region, by regressing soil organic carbon estimates against restoration age, Tangen and Bansal (2020) estimated a sequestration rate of 35–110 g C/m^2/yr, where upslope wetlands store carbon more slowly than the inner, inundated area.

The net atmospheric cooling effect of soil carbon sequestration rates can be matched and exceeded by the warming influence of methane emissions from freshwater wetlands. Across a coordinated campaign of methane measurements using eddy flux towers from 60 sites (with few observations in tropical and mountainous regions), Knox et al. (2019) found methane emissions ranged from a median of 2.9 g C/m^2/yr in a tundra wetland (interquartile range: 1.8-6.2 g C/m^2/yr) to 114.9 ± 13.4 g C/m^2/yr for an estuarine freshwater marsh, and of the over 30 measurements reported for freshwater wetlands, more than half exceeded 40 g C/m^2/yr, a value that was rarely exceeded in other wetland classes (Figure 7.8). (Eddy flux measurements of wetland methane emissions in Knox et al. (2019) were typically lower than the chamber techniques used in Treat et al. (2018), pointing to the importance of measurement methodology.) The important drivers of variation in methane emissions differed between seasonally flooded wetlands, for which methane emissions rose with water table depth, and consistently flooded wetlands, for which methane emissions rose with temperature (Knox et al. 2021). However, it takes time for methane production to begin, lagging the onset of flooding by approximately 17 days and changes in air temperature by 8 days (Knox et al. 2021), similar to (Chang et al. 2021). Productivity metrics (net ecosystem productivity, gross primary productivity and respiration) were also important to determining methane emissions across night/day, multiple days, and seasons (Figure 3 of Knox et al. 2021). Interestingly, methane emissions in freshwater lakes may also be affected by wetland fauna. In a high latitude Finnish lake, when European perch (*Perca fluviatilis*) were added to a previously fishless lake, methane emissions fell by 90% (Devlin et al. 2015). This occurred because the perch fed on the zooplankton that normally fed on methane-oxidizing bacteria (or bacteria that limited dissolved methane in the lake).

Combining carbon sequestration and methane emissions, Petrescu et al. (2015) found that methane emissions were more than 19 times the CO_2 sequestration rates in nearly 60% of the freshwater wetlands studied, where a 19:1 ratio of carbon sequestration to methane emissions would result in net cooling over 100 years. Similarly, native wetlands in the Prairie Pothole region had higher global warming potential over a 100-year time horizon than drained agricultural land, flooded croplands, and hydrologically restored wetlands (Tangen et al. 2015). However, most natural wetlands are typically much older than 100 years and have sequestered enough carbon to have a net cooling effect; nearly all of the natural wetlands examined in Taillardat et al. (2020) were likely formed with the retreat of glaciers in the last ice age (roughly 11,700 years ago).

Overall, we have **moderate-high confidence** in the freshwater carbon flux estimates. However, we caution that because of the variability in hydrology, geology and vegetation that define freshwater wetland classes differently, numerous flux measurements across wetland classes and conditions are required to generalize flux metrics accurately and with precision. For example, Rosentreter et al. (2021) describe how positive skew in methane emissions can lead to different global estimates if mean versus median methane emissions are used in calculations.

Riverine wetlands: high carbon sequestration and uncertain methane emissions

Riverine wetlands are incorporated into some of the estimates for freshwater wetlands described above, but we synthesized several papers specific to riparian wetlands separately. Interest in riparian conservation and restoration can be motivated by their potential for both soil and biomass carbon sequestration, along with their multiple other benefits (Table 3), and several studies have documented high potential for carbon sequestration in riparian wetlands and variation by fluvial setting, floodplain or forest age, and climate. However, as with other freshwater wetlands, methane emissions are important to consider. While we expect methane emissions in riparian wetlands to be similar to other freshwater systems, we did not find many studies specific to riparian wetlands.

Analyzing data from 42 stream reaches in Northern California, Matzek et al. (2020) found that soil carbon measured to 50 cm depth increased at a rate of 87 g C/m^2/yr on the floodplain and 112 g C/m^2/yr on upper banks. Mitsch et al. (2014) found higher rates of soil carbon sequestration ranging 219–267 g C/m^2/yr in created riparian wetlands along the Olentangy River in Ohio. Contrasting short- and long-term soil sequestration, Loder and Finkelstein (2020) found that short-term carbon accumulation rates in riverine systems of 105 and 207 g C/m^2/yr, in flow-through and coastal riverine systems respectively, compared to one long-term study reporting sequestration rates of 26 g C/m^2/yr.

In addition to soil carbon sequestration, accumulation of carbon in biomass is an important component of total carbon accumulation (Figure 7.3), though we found few estimates of the rates at which biomass carbon accumulates (and thus did not compile a separate figure). Matzek et al. (2015) estimated biomass carbon accumulated at a rate of approximately 325 g C/m^2/yr, and Dybala et al. (2019a) estimated a global average rate over the first 10 years of 347 or 392 g C/m^2/yr (for warm and dry climate and a cool and dry climate, respectively) if they were actively planted, more than double the rate of naturally regenerating forests (147–165 g C/m^2/yr). Similarly, Dybala et al. (2019b) estimated an average rate of biomass carbon accumulation over 30 years of 359 ± 73 g C/m^2/yr in a planted site and 229 ± 0.14 g C/m^2/yr in a nearby naturally regenerating site in California. Tan et al. (2020) report net ecosystem exchange of 787 g C/m^2/yr (standard error: 622–952 g C/m^2/yr).

In two global meta-analyses of methane emissions in riparian wetlands, Rosentreter et al. (2021) and Tan et al. (2020), reported estimates of 4.1 and 42 g C/m^2/yr, respectively. A study of methane emissions in riparian buffers in Indiana estimated an average of 0.12 ± 0.21 g C/m^2/yr (Jacinthe et al. 2015), but found high spatial variability, with one topographic depression representing < 8% of the total area contributing 78% of the annual total.

Overall, we have **low-moderate confidence** in the riverine carbon flux estimates because we have very few methane emission and carbon ecosystem flux rates.

Montane meadows: fluxes uncertain and more year-round studies needed

Carbon sequestration in montane meadows varied considerably, with some montane meadows being net sources and others net sinks of carbon. A primary driver of this variability is the level of inundation, where inundation depends on the amount and timing of snowpack melting. Methodology in montane ecosystems also differs from other wetland classes; because of the topographical heterogeneity within montane meadows, eddy flux measurements are relatively sparser in this ecosystem. As a result, we did not review any montane meadow ecosystem flux measurements. Finally, we found very few mentions of methane flux in montane meadows.

As a consequence of the seasonal and interannual variability in inundation, meadow soils can be either large net C sinks or sources of C to the atmosphere. Reed et al. (2021) estimated a range from 577.6 ± 250.5 to − 391.6 ± 154.2 g C/m^2/yr, although these measurements reflect carbon flux during the growing season, which is the period of highest carbon exchange (Tucker et al. 2014; Knowles et al. 2016). Using an eddy flux covariance tower in Loney Meadow (also included in Reed et al. 2021), Blackburn et al. (2021) found that the meadow progressed from a strong CO_2 sink due to high plant growth in the peak of the growing season under saturated to wet soil conditions (−18.51 g C/m^2/day), to a weak source (2.97 g C/m^2/day) following a rapid decline in soil moisture as runoff and plant growth decreased. The resulting annual net ecosystem exchange at Loney Meadow was estimated to be 285–450 g C/m^2/yr (Blackburn et al. 2021). Comparing growing season soil respiration and annual primary productivity in four transects in two Sierra Nevada meadows, Arnold et al. (2014) found high interannual variability in carbon fluxes. Specifically, there was a 100% increase in soil respiration in 2012 and 2013 compared to 2011, from 400–600 to 800–1250 g C/m^2 per growing season, with a moisture gradient dictating the range of values. Over the same time period there was an average 39% decline in aboveground net primary productivity, from 200–520 to 150–350 g C/m^2 per growing season, again with a moisture gradient dictating the range of values.

Variation in the direction and magnitude of net soil C flux appeared to be driven by belowground (e.g. root) C inputs (Tucker et al. 2014; Reed et al. 2021), where a surprising 35% of winter soil respiration was directly related to roots (Tucker et al. 2014). There was also a positive correlation between meadows that were carbon sinks and the fraction of their plant species associated with wetland-type functional groups (Reed et al. 2021), suggesting that more inundated meadows had higher carbon storage and sequestration potential, although Reed et al. (2021) attributed this finding to root traits.

Montane meadows may not be significant sources of methane. Reed et al. (2021) found that only 5/13 montane meadows emit methane (0.04 to 4.8 g C/m^2/yr; less than 0.6% of total soil carbon losses), and soils in the other eight meadows were small net sinks of atmospheric methane (-0.01 to -0.14 g C/m^2/yr). Reed et al. (2021) attribute the low methane emissions to the high mineral soil content in montane meadows; alternate electron acceptors may sustain high rates of heterotrophic respiration, suppressing methane production, during periods of low soil oxygen. Blankinship and Hart (2014) also found no measurable methane emissions, even when the soil was rather wet.

Overall, we have **low confidence** in the riverine carbon flux estimates because we have very few methane emission and ecosystem flux rates.

7.4 Disturbance and recovery

Partial or complete draining of wetlands, moving wetland soil, and converting wetlands to other land uses such as agriculture, disturb the fundamental functioning of wetlands and the benefits they provide. These disturbances expose soil carbon to oxygen, allowing microbial respiration to release soil carbon that has accumulated over decades, centuries, and millenia. Disturbances were defined differently in the papers we reviewed and include, for example, altering barriers to contain flooding, using wetlands for rice agriculture, or draining wetlands for dry-land agriculture. Disturbances to wetlands could occur because the consistency of inundation has been altered or because historical wetlands have become perennially drained. Restoration of disturbed wetlands are thus defined as an attempt to restore the original hydrology through active management or cessation of draining activities. Given this variety in the method and severity of disturbance, we expected variation in the effects of disturbance and the effectiveness of restoration on carbon storage and fluxes.

Understanding the effect of disturbance requires an understanding of both the change in carbon storage and the change in rates of carbon accumulation. Disturbance typically results in an initial loss of stored carbon, and while the magnitude of the initial loss is usually not known, it can be estimated by comparing carbon stocks in natural, undisturbed wetlands to disturbed or disturbed and subsequently restored wetlands. However, this comparison must also account for the new rate of carbon accumulation in the disturbed or restored wetlands and the amount of time since the disturbance. Thus, because of this interconnectedness, we synthesized the evidence for the impacts of disturbance and restoration on both carbon storage and flux together. Consistent with the papers we reviewed, we also distinguish between disturbed or restored wetlands and created wetlands, which are created by flooding a formerly dry region.

Based on our rapid evidence assessment, we summarized the general patterns found by comparing disturbed and restored wetlands to natural wetlands, to inform conservation and restoration priorities:

- **Because of the potential for loss of carbon upon disturbance, conservation of wetlands should be the highest priority.** Especially in systems, like peatlands, with high carbon storage, disturbance can lead to the rapid loss of stored soil carbon.

- **Upon restoration, carbon accumulation rates are slow.** Because of the decades, centuries, and millennia that it can take to recover the carbon that was lost upon wetland disturbance, wetland conservation is the highest priority.

- **Freshwater inundation produces methane, thus restoring wetland hydrology typically results in increased methane emissions .** From a short-term perspective, freshwater wetlands have a net warming influence on the atmosphere because of methane emissions from freshwater wetlands. However, ultimately, wetlands have a net cooling impact. Carbon storage over long time frames counteracts the warming effect of methane, a more short-lived gas.

- **Natural disturbances, such as fire, are also important determinants of carbon cycling in wetlands.** Roughly 4-12% of burned area in the U.S. occurs on wetlands, where wildfire can result in: (i) loss of soil and biomass carbon, (ii) altered plant-soil carbon exchange, and (iii) post-fire erosion. Considering periodic upheavals in carbon storage due to wildfire is important to total carbon accounting.

Wetland disturbances emit stored carbon and slow sequestration

The amount of carbon that is lost following disturbance depends on the amount of carbon in the soil prior to the disturbance and the degree of overall disturbance. For example, draining peat soils, which have very large soil carbon stocks, can cause huge losses of carbon along with emissions of other potent greenhouse gasses, such as N_2O in locations where fertilizer is applied (Teh et al. 2011; Hemes et al. 2019). Similarly, moving large amounts of soil, such as to create levees and support vegetation, can uncover previously buried carbon and lead to high rates of respiration (Oikawa et al. 2017). In an analysis of data from the 2011 National Wetland Condition Assessment, across a combined group of estuarine emergent, estuarine shrub, estuarine woody, palustrine, lacustrine, and riverine wetlands, the least disturbed wetlands stored over 400 Mg C/ha in soils, intermediately disturbed wetlands stored just under 300 Mg C/ha, and the most disturbed sites stored less than 250 Mg C/ha (Nahlik and Fennessy 2016). Similarly, comparing natural reference wetlands to areas that had been converted to other uses, such as for rice or dry-land agriculture, disturbed wetlands generally stored significantly less carbon in the soil than their undisturbed counterparts (Figure 9c-d; (Xu et al. 2019a).

Across studies, we found that undisturbed wetlands still had substantially larger soil carbon stocks than restored wetlands (Figure 7.9a), while restored wetlands had soil carbon stocks that were more similar to the disturbed wetlands (Figure 7.9b). Similarly, we found that undisturbed wetlands typically sequestered carbon at a higher rate than restored wetlands but also had increased rates of methane emissions (Figure 7.10; (Tan et al. 2020; Su et al. 2021) with positive skew in methane emissions data (Rosentreter et al. 2021). However, one notable exception to this pattern is in croplands, where crops such as rice can produce substantial greenhouse gas emissions. Globally, the potential carbon sequestration rate in restored wetlands has been estimated as between 0.3-1.3 Pg CO2(eq)/yr (Paustian et al. 2016).

Similarly, in a meta-analysis examining the potential impacts of wetland conversion to other land cover types, Tan et al. (2020) estimated that conversion from natural wetlands, riparian areas, and peatlands to any of the alternative land uses they considered would result in a net increase in global warming potential (GWP), with the exception of conversion of natural riparian areas to pasture. Specifically, the GWP of natural wetlands was only -0.86 with huge uncertainties (+/-3.92). Natural riparian areas are similarly uncertain, with a GWP = -8.74 +/-13.94. In peatlands, the global warming potential is -2.85 +/- 2.06. Across flooded regions, GWP were largely dictated by methane fluxes,where methane fluxes were highly dependent on salinity, inundation, nitrate concentration, and bulk density.

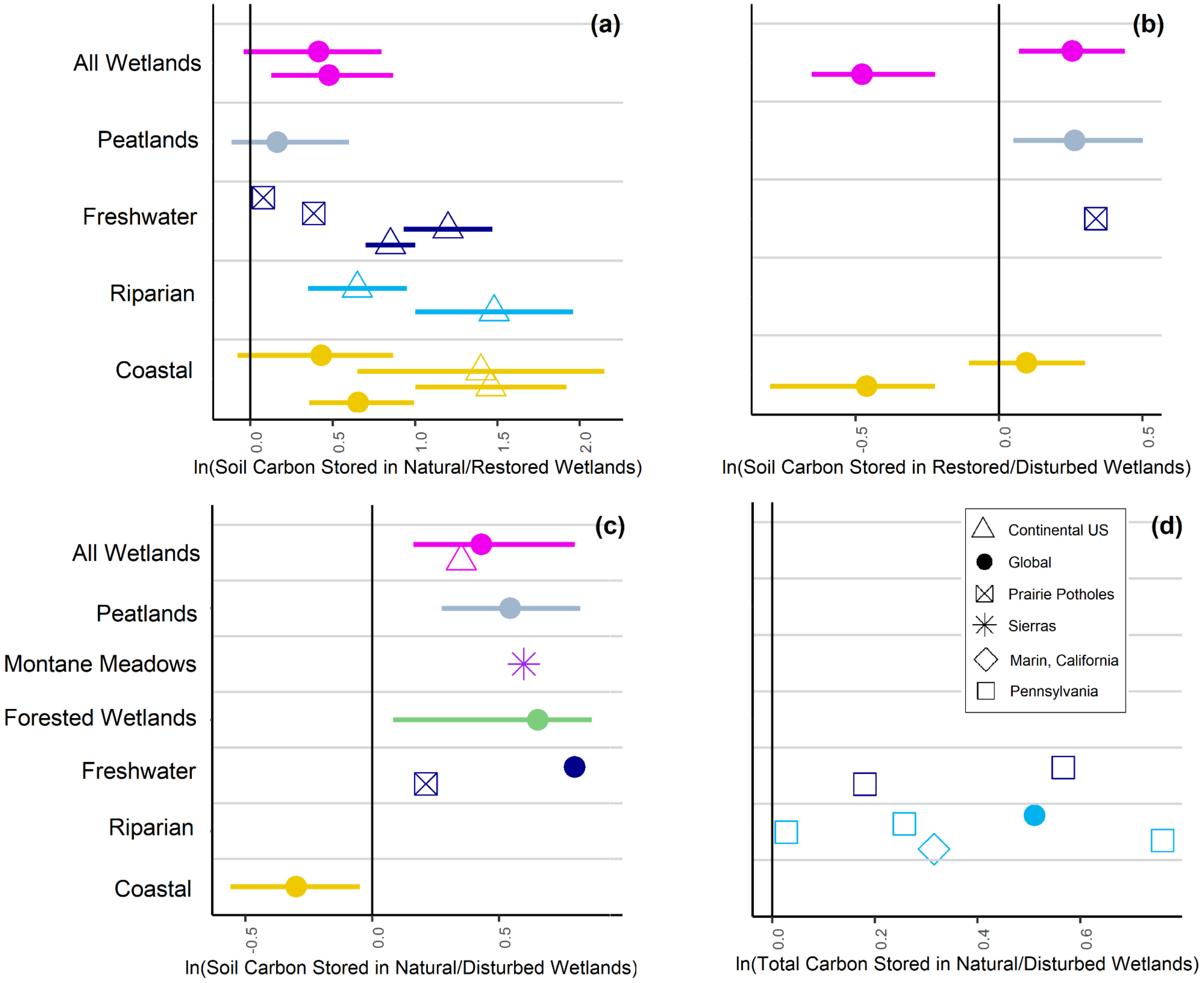

Figure 7.9. Carbon storage in natural, restored, and disturbed wetlands, shown as the ratio of Comparison of (a) soil carbon in natural versus restored wetlands, (b) soil carbon in restored versus disturbed wetlands (primarily by agricultural cultivation) wetlands, (c) soil carbon in natural versus disturbed wetlands (varied types of disturbance), and (d) total carbon (soil, debris, and above and belowground biomass carbon) in natural versus disturbed wetlands (varied types of disturbance), as measured by response ratios. By taking the natural log (ln) of the ratio of soil carbon in two different disturbance regimes, the reader can quickly interpret the impact of disturbance: values greater than zero suggest that there is more carbon in natural (for a, c, and d) or restored (b) wetlands, with larger values suggesting a larger difference in soil carbon between natural (for a, c, and d) or restored (b) wetlands. Points represent the mean value and bars are the confidence intervals, where some studies did not report confidence in their estimate. Where the bars do not cross zero, the estimated value is significantly different from zero. As shown in the legend, point shape indicated the geographic scope of the study. (Values from Moreno-Mateos et al. 2012; Norton et al. 2014; Nahlik and Fennessy 2016; O'Connell et al. 2016; Yu et al. 2017; Mazurczyk and Brooks 2018; Xu et al. 2019a, 2019b; Matzek et al. 2020; Tangen and Bansal 2020; see Conlisk et al. 2022 for additional information for individual studies)

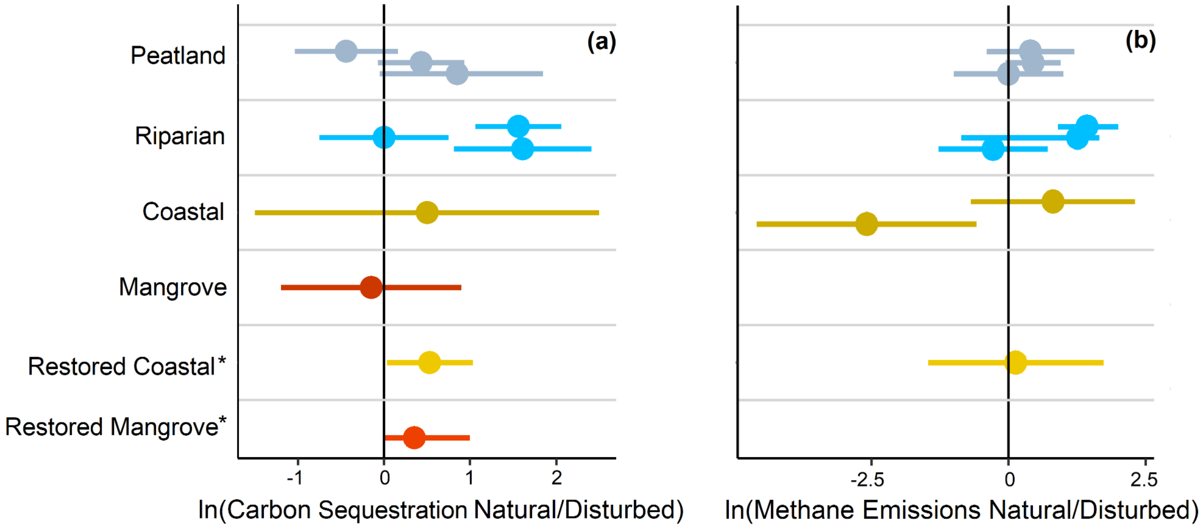

Figure 7.10. Carbon fluxes in natural, restored, and disturbed wetlands, shown as ratios of (a) carbon sequestration rates in natural versus restored or disturbed wetlands (primarily by agricultural cultivation) landscapes, and (b) methane emissions rates in natural versus restored or disturbed (primarily by agricultural cultivation) wetlands. Disturbances were primarily due to agricultural cultivation. The * indicates that a natural landscape was being compared to a restored landscape (instead of a disturbed landscape). Values greater than zero indicate that more carbon is sequestered and more methane is emitted in natural landscapes, for (a) and (b) respectively. Points represent the mean value and bars are the confidence intervals. (Values from Tan et al. 2020; Su et al. 2021; see Appendix C for additional information for individual studies)

Full recovery after disturbance may take centuries

Wetland recovery is likely to occur very slowly and wetlands may never fully recover the original magnitude of carbon stocks in undisturbed wetlands (Moreno-Mateos et al. 2012; Xu et al. 2019b; Tangen and Bansal 2020). Thus, although restored wetlands do increase in soil carbon stocks over time, in one study soil carbon stocks that were 70% lower than undisturbed wetlands after 5 years, were still 55% lower after 20 years and a projected 1000 years would be required to recover overall biogeochemical function (Moreno-Mateos et al. 2012). Similarly, in a separate study of restored and created wetlands in the continental U.S., soil carbon stocks were 72% lower than reference sites after 5 years, and still 51.7% lower after 11-20 years (Yu et al. 2017). Given the large quantities of carbon stored in peatlands, and the relatively slow rates of accumulation (Buffam et al. 2011), disturbances to these carbon stocks will also likely take a very long time to recover. In contrast, a study in the Prairie Pothole region estimated it would take only 20-64 years for soil organic carbon levels of a restored wetland to return to natural reference conditions (Tangen and Bansal 2020), but these were among the fastest rates of accumulation we found. These results suggest that while wetlands can recover from disturbance and conversion, the protection of existing wetlands from any further disturbance is essential to maintaining their carbon stocks and ability to continue sequestering carbon.

Recovery rates are influenced by inundation, hydrological connectivity, and plant growth rates

The simplest form of wetland restoration is to inundate a drained or partially drained site, limiting microbial respiration and slowing the rate of CO_2 emissions. A meta-analysis of 123 studies of CO_2 exchange from freshwater ecosystems found dry soils tended to emit carbon and inundated soils sequestered it (Wilson et al. 2016). Specifically, rewetting wetlands would increase the carbon sequestered in the soil by 0.04 – 0.05 Mg C/ha/yr for every centimeter increase in the mean water table level, with more pronounced impacts in warmer or more nutrient-rich sites and smaller impacts in colder or more nutrient-poor sites (Wilson et al. 2016). In alignment with these results, Tangen and Bansal (2020) found the fastest recovery rates in wetlands with consistent flooding.

Recovery rates can also be influenced by the composition and growth rates of the vegetation community, such as in the case of non-native and invasive plant species that can contribute to the rapid accumulation of organic matter (Davidson et al. 2018). However, we note that the introduction of non-native vegetation may have implications for other wetland benefits, such as biodiversity support (Chapter 5). Plant growth and biomass carbon accumulation may also be accelerated by actively planting wetland vegetation over allowing natural recruitment (Dybala et al. 2019a).

Wildfires release biomass carbon, store black carbon, and can alter hydrology

Despite being areas of high moisture, many wetlands across the continental U.S. experience periodic wildfire, including: pine barrens, pocosins, northern spruce peatlands, and Alaskan lowlands (Uhran et al. 2021). As a fraction of total burned area in the continental U.S., wetlands made up from 4.33-12.24% of total burned area, as measured in select years across 1984-2015 (Uhran et al. 2021). Carbon storage after a fire in floodplains can be affected by: (i) loss of pre-existing soil organic matter and burned biomass, (ii) altered plant-soil carbon exchange, and (iii) trapping floodplain sediments after increased post-fire erosion (Wohl et al. 2020). While soils can lose a large amount of carbon during a wildfire, the incompletely burned carbon (black carbon) remains in a highly stable state. The amount of black carbon, the depth of the burned layer, and the original carbon content in the soil are important determinants of the net effect of wildfire on total soil carbon content. In deep fires and peatlands, the impacts of fire can be substantial. Reddy et al. (2015) found that 440 Mg C/ha of carbon were lost in the peatland Great Dismal Swamp of Virginia and North Carolina when the 2011 Lateral West Fire burned, on average 47 cm deep, through the 25 km^2 burned area. The presence of beavers has been shown to mitigate the instantaneous effects of wildfire, providing temporary refugia in a fire event (Fairfax and Whittle 2020). Changes in plant-soil interactions following fire can also lead to altered carbon storage through downed wood, altered erosion, and changes in plant composition. Finally, wildfire can lead to higher inundation through floodplain erosion and changes in tributary flow (Wohl et al. 2020). These effects can be particularly pronounced in steep hill slopes where erosion can occur across large areal extents.

8. Special topic: The role of beaver as a wetland ecosystem engineer

By creating dams that slow river flow, beaver (*Castor fiber, C. canadensis*) are well-recognized ecosystem engineers that have profound impacts on riverine wetlands and montane meadows, and their capacity to provide multiple benefits (Jordan and Fairfax 2022). Beaver population management can be a low-cost, effective approach to restoring wetland function and resilience (Silverman et al. 2019), and while the impact of an individual beaver dam and associated ponds may be small, the cumulative impact of multiple dams within a network ("a beaver cascade") can significantly increase their impact (Larsen et al. 2021). In the absence of beaver, the construction of beaver dam analogs (BDAs), a low-tech, process-based restoration technique (Wheaton et al. 2019), can also serve to restore many of these same benefits (Case study: Bridge Creek). Thus, beaver emerged as an important topic in our rapid evidence assessment, and we summarized their impacts on multiple benefits in this separate section.

Water Supply Regulation

Beaver impoundments slow the flow of water, spread water on adjacent floodplains, and expand the wetted area (Weber et al. 2017; Karran et al. 2018; Larsen et al. 2021), increasing water storage capacity and groundwater-surface water connectivity, and contributing to drought resilience (Weber et al. 2017; Wegener et al. 2017; Karran et al. 2018). For example, prior to an extreme flood that breached beaver dams, median water tables within a 150m radius of beaver dams were on average 12.8 cm higher and more than twice as stable as compared to after the dam breaches (Karran et al. 2018). The installation of BDAs can also be effective in increasing water storage capacity and raising water tables, although it may take more than a year (Scamardo and Wohl 2020). As described above (Chapter 1), water supply regulation can have substantial impacts on other wetland benefits.

Flood Risk Mitigation

Closely related to the impacts of beaver impoundments on water supply regulation, they may also be able to attenuate smaller floods by increasing floodplain connectivity and storage capacity upstream of the dam and spreading flood waters over a larger area, but their impact on larger flood events is likely to be minor to negligible (Larsen et al. 2021).

Water Quality

Because of their influence on water flow, the presence of beaver increased nutrient removal capacity and thus water quality in montane meadows. More ammonium nitrate, dissolved organic nitrogen, total dissolved nitrogen, and dissolved organic carbon was retained in wide valley floodplains where beaver were present relative to narrower riverine floodplains without beaver (Wegener et al. 2017). For BDAs, sediment storage was positively correlated to BDA height and pond surface area (Scamardo and Wohl 2020).

Biodiversity Support

Through their impacts on the water table and wetted area, beaver dams and BDAs can have multiple impacts on biodiversity support benefits of wetlands. For example, by moderating summer water temperature extremes in their associated ponds, beaver dams and BDAs can benefit aquatic species that are sensitive to temperature, such as steelhead (Weber et al. 2017). Beaver dams and BDAs also promote the growth of vegetation (Silverman et al. 2019), in turn benefitting other wildlife and even improving resilience to wildfire. Comparing riparian wetlands where beaver damming was present and absent, Fairfax and Whittle (2020) found that after a wildfire, sections of creek with beaver were on average >3 times greener (based on Normalized Difference Vegetation Index) than sections without beaver. Further, sections with beaver may serve as wildfire refuges, providing protection from species that cannot escape fire, including fish, amphibians, reptiles, small mammals, wild and domestic ungulates, and birds (Fairfax and Whittle 2020). The installation of BDAs can also benefit beaver themselves, creating the conditions that promote the recruitment and establishment of local beaver populations. In one study, the number of natural beaver dams increased from 0 to 9 within 7 years after the installation of BDAs (Weber et al. 2017), and in another, natural beaver dams increased 4-fold within just three years (Silverman et al. 2019).

Carbon Storage and Flux

By increasing the extent of flooding in surrounding floodplains and meadows, beaver impoundments can lead to reduced microbial respiration and increased ability to accumulate organic matter in the soil. The accumulation of organic matter over years can lead to a more carbon-rich surface layer of soil that persists even after the original beaver dam is breached. In addition to affecting how much carbon is stored, beaver dams change the flow of organic matter and sedimentation moving through these systems (Wohl 2013), affecting where in the watershed carbon is stored. Carbon in sediments upstream of the beaver dam can accumulate behind beaver impoundments since they are prevented from flowing downstream. The amount of carbon accumulation in the soil depends on many factors, such as soil before the dam was constructed, how long the dam was in place, and the degree to which the dam changed the vegetation and inundation at the site after the dam was breached.

Areas with former beaver ponds have litter-dominated soil top layers (called the O-horizon) that were found to be at least twice as thick and store twice as much carbon as unimpounded areas (Johnston 2014). The specific values varied by soil type, with 57 Mg C/ha versus 123.6 Mg C/ha (to 60 cm depth) in unimpounded and impounded sandy glacio-fluvial soils (coarse-particle soil, transported by glaciers), respectively. Clay glacio-lacustrine soils (fine-particle soils from lake bottoms) had 63 Mg C/ha versus 162 Mg C/ha in unimpounded and impounded areas, respectively. Parent soil material mattered more if it had been inundated prior to the beaver dam, where one region that had been a cedar swamp prior to the beaver dam had 366.7 Mg C/ha. Regardless of soil type, the overall average beaver unimpounded and impounded soil storage was 82 versus 151 Mg C/ha, respectively. Looking across carbon pools, Mazurczyk and Brooks (2018) report 27.5, 62.5, and 14.2 Mg C/ha in aboveground, soil, and coarse woody debris carbon in beaver wetlands.

Beaver dams change the flow of organic matter and sedimentation moving through riparian systems (Wohl 2013). Upstream carbon in sediments are more likely to be deposited at a site adjacent to a beaver dam than in a location with free-moving water. Wohl (2013) estimated that 1150-1400 Mg C/ha are stored in sediments at active beaver sites and 300-400 Mg C/ha in relict beaver sites. The former estimate provides areal carbon storage estimates roughly an order of magnitude larger than studies looking specifically at soil carbon in the upper soil layers (Johnston 2014; Laurel and Wohl 2019).

Time-since beaver activity has a complicated relationship with carbon storage. The authors of Laurel and Wohl (2019) predicted that soil carbon would initially decline after beaver activity and then stabilize. While they found that long-since abandoned areas had lower soil carbon than recently abandoned areas (106 and 258 Mg C/ha, respectively), they also found that partially active areas had higher soil carbon than active areas (143 and 72 Mg C/ha, respectively; Figure 6 in Laurel and Wohl 2019). In regression analyses, the depth of the core (some cores could not be extended to their full 114 cm because of rocks), clay content, and surface geomorphic heterogeneity (of the diversity of morphology, elevation, and vegetation within the area) were all significant determinants of soil carbon stocks.

While beaver dams typically increase soil carbon and sedimentation, they also contribute to considerable CO_2 and methane emissions. During the first three years after impoundment, dissolved organic carbon in impounded waters is high (Nummi et al. 2018), leading to 'hotspots' in carbon emissions. Across 13 studies, records of daily carbon dioxide emissions scaled up to a total ranging 3.8–311 g C/m^2/yr across meadows and an average of 134 g C/m^2/yr, assuming a growing season of 100 days (Nummi et al. 2018). In addition, methane emissions ranged 2–69 g C/m^2/yr with an average of 17 g C/m^2/yr. These emissions are quite high, but are counteracted by the higher carbon storage and sedimentation rates in beaver inundated soils. We found no estimates of specific switchover times that directly address when a beaver pond would have a net warming or cooling potential.

Overall, we have **low confidence** in the storage and flux estimates of beaver areas because we only reviewed 6 studies over relatively small geographic extents

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Case Studies

Photo credit: Kristen Dybala/Point Blue

Sonoma Baylands: Building Cultural Investment in Shoreline Habitats

At a glance

- **Location:** Sonoma Baylands, San Francisco Estuary, California, USA; Adjacent to major transportation corridor State Route 37
- **Restoration Type:** Conversion of agricultural/hay fields to tidal slough, tidal marsh, tidal wetland, and upland-transition zone restoration
- **Area Restored:** ~305 acres
- **Project Leads:** Sonoma Land Trust, followed by the US Fish and Wildlife Service & California Department of Fish and Game
- **Benefits:** wildlife habitat, including for endangered and migratory species; carbon sequestration; flooding and sea level rise attenuation; supported local livelihoods, community involvement/education/access to nature; policy support for more restoration
- **Current Status:** restoration complete; monitoring and community access/use are ongoing

Overview

The Sonoma Baylands Project is a highly collaborative, multi-million dollar restoration effort initiated in 1996 covering the majority of a ~350-acre tidal marsh. In combining a large-scale tidal wetland restoration project with clean dredge from an industrial port dredging project to initiate re-building of tidal wetland habitat, followed later with a community-based upland transition-zone restoration model, this project is a model of a creative, collaborative solution to both ecological and human community problems.

Project success contributed to highlighting the importance and value of shoreline restoration in the San Francisco Estuary. For example, restoration projects such as Sonoma Baylands contributed to raising the visibility among Bay Area residents of the value of shoreline restoration efforts, paving the way for the passing of Measure AA in 2016, a 20-year, $12 parcel tax that will raise ~$500 million over twenty years to fund restoration projects in the Estuary (Wong 2020). The passing of Measure AA with 69% of the vote represents clear evidence that Bay Area residents have internalized the importance of tidal marsh habitats (Gutierrez 2016). Securing additional funding for the estuary has also been an ongoing goal. U.S. Rep. Jackie Speier, a Democrat in San Mateo County, introduced several bills and in 2022, Congress and President Joe Biden approved more than $50 million in funding to the U.S. Environmental Protection Agency for projects to restore wetlands throughout the San Francisco Bay to support wildlife habitat, natural defense against sea level rise impacts, and community access (Houston 2022; Spier 2022).

Challenge

The San Francisco Bay Estuary is the largest estuary on the Pacific Coast of the United States and accounts for 77% of California's remaining perennial estuarine wetlands (Ramsar 2013), despite the loss of over 90% of historic tidal marsh habitat (Cordell 2020) over the past 200 years to diking, draining, and bayfill (Goals Project 2015). As climate change impacts intensify, significant portions of the remnant shoreline habitat are at risk without restoration intervention (Stralberg et al. 2011, Veloz et al. 2013). Prior to the application of dredge material, the Sonoma Baylands were subsided up to 2.1 meters below their natural elevations, and up to 1.2 meters below mean sea level, and enclosed from bay water (Marcus 2000). In addition, flood events regularly impact local transportation infrastructure, including State Routes 37 and 121 (Webster et al. 2020); floods caused the closing of SR 37 for 28 days in 2017 and 8 days in 2019 (Landers 2022).

Solutions

This project illustrated the potential to overcome multiple challenges related to loss of shoreline wetland habitat with one creative, collaborative large-scale tidal wetland restoration project. A comprehensive strategy included consideration of the habitat needs of several endangered, threatened, and species of special concern, the potential for flood risk mitigation for neighboring landowners and transportation infrastructure, and public access benefits (Webster et al. 2020). The initiating design incorporated a significant portion of the roughly ~2.0 million cubic meters of dredge from the Port of Oakland to build elevation to allow restoration, and also helped provide a resolution to a previously-beleaguered channel-deepening project needed to support the area’s economically important shipping industry that had been previously hampered by lack of a solution for what to do with dredged material (Marcus and Grimm 1995, Marcus 2000).

Key Outcomes & Projected Benefits

- Carbon sequestered: We estimated 175 Mg C sequestered aboveground in the 280.6 vegetated acres of project site, and 3751 Mg C recovered in the soil (see below for a description of how these were estimated)
- Wildlife showed a positive response to restoration, including the federally listed California Ridgway’s rail. Other target species for the project included the endangered salt marsh harvest mouse and multiple species of migratory shorebirds (Marcus 2000)
- Protected local livelihoods at the Port of Oakland, as Oakland's dredging was used to produce the material that was applied to initiate project site restoration (Marcus 2000)
- At least 300-500 community members engaged in restoration activities and immersive wetland stewardship-building experiences through Point Blue’s community-based STRAW restoration program [Students and Teachers Restoring A Watershed] (L Giambastiani & J Parodi, pers. comm.)

Lessons Learned

- Tidal marsh restoration provides multiple benefits, including increasing: wildlife habitat value, carbon sequestration, and opportunity for engaging people with the value of natural habitats.
- Project success demonstrated the importance of deep stakeholder engagement and coalition building in implementing and sustaining large-scale restoration projects.
- Committed community and stakeholder engagement can help build cultural investment in wetland habitats. This project engaged local community members in discussions around, and in some cases active participation in, restoration of tidal marsh and adjacent transition zone habitats through Point Blue’s STRAW program. [Students and Teachers Restoring A Watershed] (L Giambastiani & J Parodi, pers. comm.)

Carbon Sequestration Estimates

To determine a coarse‑scale estimate of the aboveground carbon sequestration benefits in biomass, we fit a boosted regression tree model to predict observed tidal marsh vegetation biomass (Byrd et al. 2017) as a function of several spatially explicit covariates: distance to bay, distance to channel, channel density, tidal range, summer salinity, and elevation. We used the model to predict the total biomass in each cell representing all 280.6 vegetated acres of project site, assuming 2020 elevations, and assumed the carbon fraction of the total biomass was 0.441 (Byrd et al. 2018). Summing over all vegetated area, we estimated a total of 175 Mg C stored in the biomass.

To estimate the recovery of belowground carbon stored in the soil, we first assumed a conservative estimate of 200 Mg C/ha in conterminous US estuarine marshes (Bridgham et al. 2006). We estimated 40% of the initial soil carbon storage, or 80 Mg C/ha, was lost due to disturbance, based on an estimated 40% decrease in carbon storage between “least disturbed” and “most disturbed” wetlands in the conterminous U.S. (Nahlik and Fennessey 2016). We then estimated the total amount of soil carbon recovered since restoration by assuming 1.33% of the initial 80 Mg C/ha lost is sequestered per year, based on estimates that a restored wetland plot can improve over time from a 70% decline in carbon storage relative to an undisturbed wetland to a 50% decline over 15 years (Moreno-Mateo et al. 2012; Yu et al. 2017; [(0.7-0.5)/0.7]/15 = 0.0133). With 25 years since the restoration of 141 ha, we therefore estimated a total of 3,751 Mg C sequestered in the soil as a result of the restoration project.

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Hamilton City: A Small California Community's Journey to Solve a Flooding Problem

At a glance

- **Location**: Hamilton City, Glenn County, California
- **Population**: 2,446 (2020); 1,759 (2010)
- **Restoration Type**: Reconnected floodplain, set-back levee, and restored riparian forest habitat
- **Area restored**: 6.8 miles of levee, 1,400 acres habitat
- **Project Leads**: Army Corps of Engineers, Hamilton City Reclamation District 2140, The Nature Conservancy, California Department of Water Resources
- **Project Websites:** https://www.spk.usace.army.mil/Missions/Civil-Works/Hamilton-City/ ; https://riverpartners.org/project/hamilton-city/
- **Benefits**: Flood risk mitigation, wildlife habitat, inclusion and empowerment of an underserved community
- **Current status**: As of summer 2022, levee is nearly complete, with some culverts, drainage outlets, and pumps still need to be installed; Restoration is in the first phase: end goal is 1400 acres and 900 acres done so far

Overview

The residents of Hamilton City, a small unincorporated Northern California community, had been tirelessly working since the 1970s to protect their community from flooding that often evacuated them from their homes. With time and perseverance of the residents, multiple stakeholders from the local to federal levels came together to solve the problem by working not only in partnership with each other, but also with nature. This restoration project was the first in the nation to be designed using new Army Corps of Engineers guidelines to develop multipurpose projects that include both flood risk reduction and ecosystem restoration (U.S. Army Corps of Engineers 2003; Plain 2011). It is also an excellent example of a more horizontal structure in planning and decision-making, where residents were consulted and included in the creation of the final outcome (Miyake 2012). The project entailed constructing 6.8 miles of setback levee, an earthen mound that is set back a certain distance from the main river channel to allow space for flood water to distribute across the floodplain in high water events, instead of flooding the adjacent community. The setback levee would improve flood protection for the community while also restoring about 1,400 acres of native habitat, including riparian forest, scrub, oak savannah, and grassland, with multiple benefits for fish and wildlife. According to those project staff consulted for this case study, budget approval hinged on the inclusion of the value of ecosystem restoration alongside flood protection. The cost-benefit analysis considering only flood protection did not meet the federal government's requirement that for every dollar spent in construction was matched or exceeded by the value of reduced future damages caused by flooding, but including the value of the restored ecosystem increased the benefit-to-cost ratio to 1.8 (Cook 2006).

Challenge

The small agricultural town of Hamilton City experienced flooding that caused residential evacuation on a regular basis between 1974 and the early 2000s. The community is small, with about 2,400 residents with a median household income of $50,552 (U.S. Census Bureau 2020). The project cost to deconstruct the ineffective existing “J levee” and construct a new setback levee was much higher than the total calculated protected assets. From an ecosystem perspective, continuous healthy river habitat had been reduced, disrupted, and divided by agricultural pursuits over the decades resulting in species decline, harmful erosion, and damaging flooding.

Solutions

Adding in the monetary value of restoring the floodplain and associated riparian habitats increased the benefit-to-cost ratio of the project more than enough to justify budget approval for the project. Hamilton City community members' experiences and knowledge were also incorporated into the planning and design through several town meetings over the many years of the project’s planning phase. Government administration at the outset of the project was in support of endeavors that bolster ecological, economic, and community health. Initial funding was authorized by Congress in 2007 through the Water Resources Development Act. President Barack Obama's Civil Works 2012 budget proposal allocated another $8 million towards the project, one of only two federally approved construction new-starts in the nation. An Army Corps of Engineers policy (U.S. Army Corps of Engineers 2003) was in place to support Civil Works projects that were consistent with environmental sustainability. Leadership from Congressman Garamendi, Congressman LaMalfa, and Senator Feinstein helped support the last $22 million of needed funding to complete the project. There was also a strong ongoing partnership between the leading entities–US Army Corps of Engineers, CA Department of Water Resources, The Nature Conservancy, and County Reclamation District 2140–to push the project forward through the decades it took to complete.

Key Outcomes & Projected Benefits

The restoration effort is projected to provide Hamilton City with protection against a 75-year and other intermittent flood events that affect their safety and livelihoods. The expected ecosystem benefits include wildlife habitat and habitat connectivity for species like the federally threatened valley longhorn beetle, Western Yellow-billed Cuckoo (a species of concern), ringtails, and bats (Snow 2021). The types of habitats restored are riparian forest, scrub, oak savanna, and grassland. Reconnection of the floodplain with the river will support the recovery of impacted wildlife like endangered salmon. The inclusive approach to planning and implementation that thoroughly considers people and wildlife needs and benefits will also set an example for future projects.

Plant survival was monitored over three years after planting occurred in restoration areas and was documented as very high in year three (an overall average of 89.4% survival rate). Along with good plant survival and vigor, birds such as House Finch and Lesser Nighthawk were observed nesting in the project area (River Partners 2020).

Lessons Learned

- Putting a value on ecosystem components enables multi-benefit projects to secure funding while traditional single-purpose projects may be more difficult to fund, particularly in smaller, more rural communities

- Combining human community needs and wildlife ecosystem needs creates a powerful case for support from multiple stakeholders
- Including the local community's existing experience and efforts is invaluable to long-term success

Partners & Their Roles in the Restoration Project

The U.S. Army Corps of Engineers is the federal sponsor for the project and provided planning, engineering, project management, and contracting services for the ecosystem restoration and construction services for the Project. The California Department of Water Resources funded a majority of the local share of the Project's cost through a Flood Corridor Program Grant to Reclamation District 2140. Reclamation District 2140, a California Reclamation District formed and existing pursuant to California Water Code Sections 50000 et seq, is the non-federal sponsor of the Project, and now owns, operates, and maintains the new setback levee after it was completed in 2021. The Nature Conservancy, a non-profit organization, completed all of the necessary real estate work for the project between 2001 and 2021, designed the habitat restoration, provided technical support for the levee design, and led lobbying efforts for the community to secure all of the state, federal, and private funding required for the project. Home and business owners in the community and local agricultural landowners within the project boundary supported assessment fees to be paid to RD2140 to operate and maintain the new set-back levee since 2006 and have been organizing to construct a better levee to protect the community since the mid-1990s. They ran "levee festivals" from 1998 to 2004 to raise money for the effort and raised over $40K over that total time, falling short of the need to fix the levee, but initiated the effort to gain the funding and support needed.

Timeline of the Restoration Project

- 1974 and the early 2000s: residents experience a history of flooding and evacuation due to flooding, lead actions to fix the levee including running annual Levee Festival fundraising events.
- 2007: Project authorized by Congress through the Water Resources Development Act
- 2012: Obama's Civil Works budget proposal allocates $8 million towards the project, one of only two federally approved construction new-starts in the nation
- 2015: Phase 1 of the new set-back levee construction began
- 2017: The Phase 1 levee construction was completed
- 2017-2021: Phase 1 habitat restoration completed
- 2018: The Phase 2A levee construction was completed
- 2020: The Phase 2B levee construction completed which brought the total to 5.5 miles of the 6.8-mile levee.
- 2021: $22 million in federal appropriations was awarded to the Project to complete the remainder of the levee construction: 1.3 miles of levee construction
- 2023-2025: Phase 2 habitat restoration will be implemented.

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Bridge Creek: Using Beaver Dam Analogues to Restore Floodplain Processes

At a glance

- **Location**: Lower 32 km of the Bridge Creek Watershed, within the John Day River Basin, Oregon, USA
- **Restoration Type**: Low-tech, process-based restoration of a riverscape using beaver dam analogs (BDAs)
- **Project Leads**: NOAA Northwest Fisheries Science Center, Eco Logical Research, Inc., and South Fork Research, Inc.
- **Project Funding:** Bonneville Power Administration, NOAA Fisheries
- **Benefits**: Threatened steelhead (*Oncorhynchus mykiss*) recovery, floodplain reconnection, improved riverscape health, wildlife habitat
- **Current status**: The lessons learned from this study were used to establish standards of practice for low-tech process-based restoration, which has led to hundreds of practitioners implementing projects across thousands of stream kilometers around the United States.

Overview

The Bridge Creek Intensively Monitored Watershed Project was a long-term study aimed at increasing the population of wild Middle Columbia Steelhead (*Oncorhynchus mykiss*), which is listed as threatened under the Endangered Species Act, through restoration of the lower 32 km of Bridge Creek. The project was designed as a large-scale experiment to evaluate the effectiveness of beaver dam analogs (BDAs) in restoring river and floodplain function, with the ultimate goal of measurable improvements to steelhead trout population processes. Beaver and salmonids were once abundant in the Pacific Northwest, but beaver were nearly extirpated by 1900, river and stream channels were incised and degraded due to land and water use practices, and as a direct result, salmonid populations declined steeply. By restoring natural riverine processes in Bridge Creek, this experiment demonstrated multiple benefits to the ecosystem, including habitat quantity and quality for both beaver and fish. Within four years, the abundance and survival of juvenile salmon had significantly increased compared to an untreated watershed nearby. The success of this project provides evidence for the use of BDAs as an effective technique to facilitate the restoration of river and floodplain function. Once re-established, these naturalized systems can support resilient beaver colonies and therefore the extent and persistence of beaver-modified floodplains, in turn providing clear benefits to aquatic species, including salmonids.

Challenge

The challenge was to transform Bridge Creek from a simplified, incised channel with high flow energy to a structurally complex riverscape dominated by low flow energy. River systems with incised channels are a widespread problem across arid and mesic environments. Incised channels can result from changes in land use or alterations to stream geometry that contribute to steeper channel slopes or higher rates of water flow (Pollock et al. 2014), promoting erosion of the streambed faster than sediment is replaced. In deeply incised channels, the stream becomes disconnected from its associated floodplain wetlands, and will often have warmer water temperatures, lower base flows, increased potential for intermittent flows, and reduced quantity, quality, and diversity of in-stream habitat for fish and other aquatic organisms (Pollock et al. 2014). In addition, incised stream channels can only support a narrow corridor of riparian vegetation, further impacting the quality and quantity of habitat for terrestrial wildlife.

Recovery from channel incision and disconnected floodplains requires restoring the natural processes that support structural complexity in the stream channel and on the valley floor, such as large downed trees, erosion-resistant vegetation (e.g., sedges), and features constructed by beaver, such as dams, canals, and food caches. In healthy riverscapes, this structural complexity slows the flow of energy, water, and materials downstream and encourages their spread onto the adjacent floodplains. However, in an arid setting such as Bridge Creek, large woody plants are rare, and because of the channel simplification, erosion resistant vegetation was no longer present. Although the watershed did have a small beaver population, their dams regularly failed during seasonal high water events due to a lack of sufficient large woody vegetation for their construction, particularly in an incised channel with artificially elevated stream power (Demmer and Beschta 2008). Consequently, beaver dam presence was highly variable, spatially and annually, and beaver alone would be unable to contribute to the transformation of Bridge Creek until the unit stream power was reduced and the riparian vegetation community was more robust (Macfarlane et al. 2017).

Solutions

The Bridge Creek IMW project experimented with hand-built structures meant to mimic the hydraulic roughness of beaver dams, but withstand the artificially elevated stream power due the simplified channel structure (see Pollock et al. 2012 for a detailed description of the study design). Hand-built hydraulic roughness elements, such as BDAs or post assisted log structures (PALS), are a low-tech process-based approach to initiating and promoting natural processes for restoring rivers and their associated floodplain wetlands. BDAs can partially replicate the function of natural beaver dams, and where beaver are present but woody riparian vegetation for their dams is limited, they can also provide stable platforms for the beaver to build on.

As a basic design principle, BDAs (and PALS) are constructed from locally-sourced, primarily organic material that can be harvested and transported by hand. When appropriate, additional anchoring strength can be achieved through the use of small diameter (<4”) wood posts driven into the stream bed or bank. Requiring only 1–4 hours for a team of three to install each BDA (Bouwes et al. 2016), the Bridge Creek implementation team was able to install a high density of BDAs (>120 over the treatment reaches). The physical and biological response to the addition of structural complexity was rapid in Bridge Creek due to the high sediment wash- and bed-load moving throughout the year, riparian vegetation species and growing conditions that supported vigorous spread and sprouting once wetted, and an existing beaver population that responded by actively maintaining and adding to the BDAs. As a result, the total number of dams, BDA, beaver, and hybrid, increased substantially from pre- to post-treatment, and continued to increase after BDA construction had ceased.

Establishing this project as a watershed-scale restoration experiment, with extensive physical and biological monitoring pre- and post-treatment, was also an effective way to document the benefits of the restoration to salmonid populations (Pollock et al. 2012; Bennett et al. 2016). By comparing treated reaches with BDAs, control reaches without BDAs, reference reaches with minimal beaver activity, and a nearby watershed, the impacts on stream temperatures, habitat quality, and fish populations were all assessed on the scale of individual reaches to entire watersheds.

Key Outcomes & Projected Benefits

- Natural beaver dam presence in the Bridge Creek watershed was previously highly variable (ranging 9–103 dams). Within 4 years of BDA installation, 236 total dams were functioning on the initial 4 km of treated stream reach, of which 115 were initiated by beaver, and 56 more were built on BDAs (Bouwes et al. 2016).
- Due to the combined impact of BDAs and natural beaver dams, both treatment and control reaches had more and deeper pools than reference reaches without minimal beaver activity, and maximum stream temperatures averaged 1.47°C cooler (Bouwes et al. 2016).
- The groundwater level in treatment reaches was 0.25m higher than control reaches (Bouwes et al. 2016)
- Wetted channel area increased by as much as 334% in stream sections with a high amount of beaver activity (Weber et al. 2017).
- Compared to a nearby watershed selected as a control, the entire lower 32 km of Bridge Creek had 81 more juvenile fish per 100 m, juvenile survival increased by 52%, and overall juvenile production (density x growth x survival) was 175% higher (Bouwes et al. 2016).
- There was no change in the percent of tagged adult steelhead found upstream of specific detection points before and after the BDA installation and the increase in natural beaver dam numbers, and juvenile salmonids were tracked moving freely across natural and constructed dams at all seasons and flow stages, indicating that neither were a barrier to fish movement (Bouwes et al. 2016).

Lessons Learned

- The large-scale restoration experiment and extensive, rigorous data collection were effective in demonstrating the multiple benefits of this low-tech, process-based restoration approach, and provided confidence in using this approach more broadly and incorporating process-based restoration planning into other restoration projects.
- The low-tech nature of hand-built BDAs makes them more accessible to those interested in restoration, and the technique is being shared through workshops, classes, and the Beaver Restoration Guidebook, emphasizing the importance of appropriate design and placement (Pollock et al. 2015). In addition, the Low-Tech Process-Based Restoration (LTPBR) of Riverscapes Design Manual emphasizes the creation of a standard of practice informed by multiple restoration projects across the western US, including Bridge Creek (Wheaton et al. 2019).
- Some ranchers viewed elevating the water table and subirrigation of adjacent pasture as a potential benefit and sought to install BDAs on land they owned; other ranchers expressed concern about the potential for unwanted flooding in adjacent pastures and beaver activities including felling large trees and blocking irrigation structures (Davee et al. 2019). Because BDAs are likely more effective when beaver are present to maintain and expand upon them, promoting beaver tolerance and finding ways to prevent conflict, such as through pond levelers and culvert fencing devices, will be helpful.

- The Oregon Department of State Lands is developing a fast-track permitting process for future BDA applications. Oregon Department of Fish and Wildlife has developed a fish-passage assessment criteria for BDA based projects based on the standard of practice in the LTPBR Manual. Multiple programmatic ESA consultations (BPA HIPBO 3, USACE WA Slopes, USDA/DOI ARBO, NRCS Conservation Practices) include BDA based work as an approved restoration practice in accordance with the Endangered Species Act.

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Appendix A. Rapid evidence assessment methods

To synthesize the vast scientific literature on wetlands, carbon sequestration, and other co-benefits, we adapted a protocol for rapid evidence assessment (Varker et al. 2015). For simplicity, we conducted an assessment of the scientific literature on a broad range of wetland benefits separate from a targeted assessment of the literature on wetland carbon. However, for each of these, we followed a consistent process of defining the search strategy, screening the literature using predetermined criteria for inclusion or exclusion, and data extraction. For both assessments, we defined the scope of our study to focus on recent scientific publications containing data from field studies conducted in the contiguous United States.

General assessment

Search Strategy

We conducted a search of the wetland benefits literature using Web of Knowledge on July 20, 2021, using Advanced Search with the following terms: TS=(riparian OR floodplain OR meadow OR wetland OR "vernal pool" OR estuar* OR marsh OR fen) AND TS=(soil OR *forest* OR vegetation OR buffer* OR stand* OR peat* OR montane OR mountain OR freshwater OR tidal) AND TS=(habitat OR biodivers* OR "water quality" OR flood* OR agricultur* OR *economic OR flow* OR groundwater OR recharge OR runoff OR wildlife) AND TS=(service* OR *benefit* OR improv* OR enhance*). The search produced 15,889 records, which we limited to those published within the past 10.5 years (2011-2021), resulting in 10,943 records that were exported to Excel. After removing duplicates, including those identified by the carbon assessment described below, there were 10,939 records remaining.

Screening

For all 10,939 records we screened the titles and abstracts, using similar predefined criteria for inclusion and exclusion in the synthesis including:

- *Scope:* Study focused on one or more co-benefits provided by wetlands. Co-benefits were defined as those aspects of wetland function that benefit local communities of people as well as provide high quality habitat for fish and wildlife (after Gardali et al. 2021). Examples of co-benefits include but are not limited to habitat and biodiversity, flood risk mitigation, groundwater recharge, erosion control, aquatic refugia, and water supply.
- *Study Type:* Record represented a peer-reviewed article describing meta-analyses, reviews, and empirical studies with field observations. We included modeling studies when the phenomenon under investigation is otherwise difficult to empirically assess, such as flood mitigation. We excluded empirical studies that were highly experimental, manipulative, and/or lab-based, and we excluded conference proceedings, white papers, and book chapters.
- *Geography:* Study occurred in North America, including the U.S., Canada, and Mexico. We also included studies for which observations occurred in multiple geographies as long as at least one observation occurred within North America.
- *Habitat Type:* Study occurred in a terrestrial wetland system, such as freshwater wetlands, fens, or wet montane meadows. We excluded studies conducted in other landcover types, such as seagrass meadows, estuaries, or non-riparian/non-wetland

forests. We also excluded studies of constructed or created wetlands unless they were compared to a natural wetland.
- *Time Period:* Observations were based on modern conditions. We excluded studies focused on paleo carbon.
- *Language:* Study written in English, to allow the project team to interpret accurately.
- *Availability:* Full text of the study was available through the UC Davis library system.

After this initial review, we identified 439 relevant papers for further review. We then reviewed titles, abstracts, and full text as needed to further evaluate which papers were suitable for inclusion. This resulted in 263 papers that were excluded and 176 papers that were included.We added 7 studies identified from reviewing the Literature Cited in these studies, 14 studies that were identified through our carbon literature search as being relevant for co-benefits, and 1 additional study of montane meadows that was deemed important to include based on expert opinion (Hunt et al. 2018). This resulted in a total of 198 records prioritized for deeper review.

Metadata Extraction, Prioritization, & Gap Analysis

For the remaining 198 records, we extracted metadata from the article full text, including:
- *Study Area:* The location where the study occurred.
- *Country(ies):* The country or countries where the study occurred.
- *Habitat(s):* The specific kind(s) of wetland(s) studied (e.g., freshwater wetland, tidal marsh)
- *Habitat Condition:* The condition of the wetland as described by the author(s) (e.g., natural, restored, disturbed, constructed, etc.)
- *Aims:* The purpose/objective(s) of the study.
- Study Type: The general study methodology, such as meta-analysis, review paper, or empirical study.
- *Co-Benefits:* The specific co-benefits studied by the authors (e.g., flood risk, agriculture, water supply, etc.)

Based on review of these metadata, we identified 58 records as high priority for inclusion in the synthesis because together they provide a representative sample of the co-benefits and wetlands we were exploring, and represent a mix of meta-analyses, literature syntheses, modeling, and experimental studies. We then conducted a gap analysis to identify co-benefits for which we did not sufficient representation, and identified a need for more information about flood protection benefits provided by riparian and floodplain wetlands. Therefore, we conducted a supplemental search using Web of Knowledge on October 7, 2021, using Advanced Search with the following search terms: TS=(riparian OR floodplain) AND TS=(flood*) AND TS=(protect*). After refining the results to those published within the past 10.5 years (2011-2021), and screening titles and abstracts, we exported 6 additional high priority articles to Excel, for a combined total of 64 high priority records. As we reviewed the 64 high priority records in-depth, we further added an additional 16 papers that were frequently cited or suggested per expert advice for inclusion, for a grand total of 81 high priority papers included in this synthesis.

Data Extraction & Synthesis

We focused on the 81 high priority studies identified in (Appendix B). These publications include empirical and modeling studies, but also a large suite of papers that are meta-analyses, reviews, or syntheses, containing data from many other studies, sites, and geographic regions; thus our evidence synthesis represents far more than 80 individual studies. From each publication, one reviewer extracted relevant data and qualitative information about the reported wetland benefits, including paper aims and overall conclusions, geographic coordinates, wetland class, benefit category, data type (e.g. nitrate removal efficiency, dollar amount per acre) and summary statistics if provided. We also extracted overall findings and covariates when relevant (e.g. restored/constructed wetland, assimilation wetland, canopy cover, beaver dam presence). A second reviewer then reviewed the full text of each of the 79 papers, confirming the accuracy of data extracted. Due to the breadth and diversity of data extracted, benefits were then synthesized primarily in a qualitative fashion, with a focus on reporting key takeaway points from the evidence base of publications reviewed, while reporting quantitative values where possible and appropriate.

Targeted carbon assessment

Search Strategy

We conducted an initial search of the wetland carbon literature using Web of Knowledge on July 6, 2021 using Advanced Search with the following search terms, intended to capture the range of terminology used to describe different wetland classes and the range of possible carbon-related metrics: TS=(riparian OR floodplain OR meadow OR wetland OR “vernal pool” OR estuar* OR marsh OR fen) AND TS=(soil OR *forest* OR vegetation OR buffer* OR stand* OR peat* OR montane OR mountain OR freshwater OR tidal) AND TS=(carbon) AND TS=(pool* OR stock* OR stor* OR restor* OR sequest* OR accum* OR devel* OR product* OR emit* OR emission*). The initial search produced 12,767 records, which we further limited to those published within the past 10.5 years (2011-2021), resulting in 8,872 records that were exported to Excel. After removing duplicates, there were 8,868 records remaining.

After further discussion among the project team, we conducted a supplementary search of Web of Knowledge on July 19, 2021 to identify papers focused on net ecosystem exchange of CO2 in wetlands, which may have been missed by the previous search, using Advanced Search with the following search terms: TS=(riparian OR floodplain OR meadow OR wetland OR “vernal pool” OR estuar* OR marsh OR fen) AND TS=(soil OR *forest* OR vegetation OR buffer* OR stand* OR peat* OR montane OR mountain OR freshwater OR tidal) AND TS=(“net ecosystem exchange”). This supplemental search initially produced 329 records, which we again further limited to those published within the past 10.5 years (2011-2021), resulting in 235 records exported to Excel. We compared the records from this supplemental search to those already identified in the previous search, and found and removed 211 duplicate records, leaving 25 additional records to include in the rapid evidence assessment, for a grand total of 8,893 records.

Screening

We screened the titles, abstracts, and full text as needed, for all 8,893 records using predefined criteria for inclusion and exclusion in the synthesis, resulting in 457 records remaining:

- *Scope:* Study focused on one or more aspects of carbon cycling in wetland systems, such as carbon storage, aboveground biomass carbon, and/or greenhouse gas emissions.
- *Study Type:* Study results were based on field measurements, excluding pure modeling studies and manipulative experimental studies. We occasionally retained modeling studies when they incorporated field-based observations.
- *Geography:* Study occurred in North America, including the U.S., Canada, and Mexico. We also included studies for which observations occurred in multiple geographies as long as at least one observation occurred within North America.
- *Habitat Type:* Study occurred in a terrestrial wetland system, such as freshwater wetlands, fens, or wet montane meadows. We excluded studies conducted in other landcover types, such as seagrass meadows, estuaries, or non-riparian/non-wetland forests. We also excluded studies of constructed or created wetlands unless they were compared to a natural wetland.
- *Time Period:* Observations were based on modern conditions. We excluded studies focused on paleo carbon.
- *Language:* Study written in English, to allow the project team to interpret accurately.
- *Availability:* Full text of the study was available through the UC Davis library system.

Metadata Extraction & Prioritization

For the remaining 457 records, we extracted metadata from the article full text, including:

- *Study Area:* The location where the study occurred.
- *Country(ies):* The country or countries where the study occurred.
- *Habitat(s):* The specific kind(s) of wetland(s) studied (e.g., freshwater wetland, tidal marsh)
- *Habitat Condition:* The condition of the wetland as described by the author(s) (e.g., natural, restored, disturbed, constructed, etc.)
- *Focus:* The specific aspect(s) of the carbon cycle that were measured by the author(s) (e.g., soil organic carbon, CO2 net ecosystem exchange, aboveground biomass C)
- *Aims:* The purpose/objective(s) of the study.
- *Co-Benefits:* Any additional co-benefits associated with the wetland(s) under study that were measured by the author(s) (e.g., water quality, flood protection)

Based on these metadata, we then identified 69 high priority records to include in the synthesis because they incorporated data from North America, included estimates of CO_2 emissions (i.e., did not only report N_2O or CH_4), and represented either (1) a meta-analysis of multiple studies, (2) a single study with a large sample size, or (3) a wetland class that was otherwise underrepresented (e.g., beaver wetland, montane meadow). To these, we added 9 more that were frequently cited by these publications, for a grand total of 79 high priority publications.

Data Extraction & Synthesis

We focused our synthesis on the 78 high priority publications identified above (Appendix B). However, most of these publications represented meta-analyses, reviews, or syntheses, containing data from many other studies, sites, and geographic regions, and thus our data synthesis represented far more than 78 individual studies. From each publication, we extracted relevant data quantifying carbon flux or storage, as well as important covariates identified as having an influence on these metrics, including wetland class, frequency of inundation, salinity, climate, and vegetation type. For meta-analyses, reviews, or syntheses that provided summary statistics across multiple studies (e.g., mean and variance), we generally extracted these summary statistics for use in comparing estimates across wetland classes, rather than attempting to extract data for each of the underlying studies included, which were not always reported. For the few publications that represented an individual study and provided multiple estimates of individual data points, we calculated the median values of target metrics, as well as extracted the absolute minimum and maximum values reported.

Appendix B. Literature Synthesized

General assessment

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