Quantitatively Estimating Carbon Sequestration Potential in Soil and Large Wood in the Context of River Restoration

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Restoration aimed at rewetting the valley floor has the potential to increase organic carbon stock in the form of floodplain soil carbon, downed wood, and riparian vegetation. The primary goal of stream restoration is typically to restore habitat or maintain balance between natural ecosystem function and human land use. Although many benefits result from stream restoration, the carbon sequestration potential of different restoration approaches in diverse geographic settings has not yet been quantified. We investigate the carbon storage potential of restored stream segments (known as treatment segments) relative to otherwise analogous degraded and reference segments. We develop a conceptual framework to identify the conditions that maximize carbon storage in relation to characteristics of the river corridor and specific restoration practices and propose response surfaces for carbon storage. We illustrate application and quantification of the conceptual framework using data from a pilot study of treatment, degraded, and reference stream segments along two streams in Oregon, United States. The conceptual model is designed to help managers identify levels of hydrologic connectivity, channel and floodplain dynamics, floodplain vegetation, and other variables that may optimize carbon storage at a treatment site.

Keywords: floodplain, large wood, organic carbon, soil carbon, restoration, Oregon

INTRODUCTION

Increasing concern about warming climate is driving increased interest in diverse forms of carbon sequestration (e.g., Lal, 2008; Villa and Bernal, 2018; Gifford, 2020). The saturated, reducing environment of wetlands limits microbial decay of organic material, and wetland soils therefore typically have much higher concentrations of organic carbon than nearby soils with lower soil moisture (e.g., Nahlik and Fennessy, 2016; Carnell et al., 2018). Wetlands can also sequester carbon at rates 30 to 50 times higher than forests (e.g., Tangen and Bansal, 2020). River floodplains that are hydrologically connected to the active channel can include substantial areas with ponds, lakes, and diverse types of wetlands (e.g., marshes, swamps, carrs, wet meadows). River floodplains can also be seasonally or perennially wet because of regional groundwater inputs associated with geologic structures (e.g., Tooth and McCarthy, 2007; Assine et al., 2015; Koltzer et al., 2019). Consequently, river floodplains can contain disproportionately large soil carbon stocks relative to adjacent uplands (Wohl et al., 2012).

Organic carbon stock in river corridors (active channel(s), floodplain, hyporheic zone) occurs primarily in the form of floodplain soil (here, soil refers to all floodplain sediment and includes litter and duff created by particulate organic matter smaller than large wood), downed dead wood pieces >10 cm in diameter and 1 m in length (hereafter, large wood), and living riparian vegetation (Sutfin et al., 2016). Active channels can contain substantial quantities of large wood (Triska, 1984; Wohl, 2014; Boivin et al., 2015), but the majority of carbon in most river corridors is found in the floodplain (e.g., Sutfin et al., 2016; Scott and Wohl, 2020). River restoration has traditionally been focused on the active channel but is gradually broadening to include an explicit focus on hydrologic connectivity within the river corridor. River restoration has also expanded to include processes that create and maintain desirable floodplain characteristics (Wohl et al., 2015), such as lateral channel migration that results in secondary or abandoned channels and associated habitat diversity (Hall et al., 2007). In the context of this broadening focus, river restoration strategies that have the potential to increase diverse forms of floodplain carbon stock may be justified as a mechanism of carbon sequestration.

The cumulative historic, contemporary, and potential future carbon stocks in river corridors worldwide have not been quantitatively estimated, but evidence suggests that this cumulative stock could be significant for the global carbon budget. The most recent estimates of this budget suggest fluxes of 9.6 Gt C per year from fossil-fuel emissions, 1.6 Gt C/y from land-use emissions, 2.5 Gt C/y into the oceans, and 3.4 Gt C/y into terrestrial C sequestration (Friedlingstein et al., 2020). Floodplains constitute about 9% of total land area (Nardi et al., 2019). Although the range of soil organic carbon stock (Mg C/ha) reported for floodplains is too large to justify assuming a single median value (e.g., Sutfin et al., 2016), soils are the largest terrestrial carbon reservoir (e.g., Scharlemann et al., 2014) and wetland soils have higher carbon concentrations than other soils (e.g., Nahlik and Fennessy, 2016). Wetlands store 20–30% of the estimated 1,500 Gt of global soil carbon, for example, but occupy 5–8% of land area (Nahlik and Fennessy, 2016). These numbers suggest that wetland floodplain soils can cumulatively create globally significant carbon sequestration.

Here, we propose a framework for using multiple lines of evidence to assess the carbon sequestration potential for a river corridor at the reach scale. We define a reach as a continuous length of river corridor that is at least several times the average bankfull channel width and has consistent channel and valley morphology as delineated based on valley floor width and gradient, channel planform, and flow regime. We also provide two case studies using recently completed stage 0 restoration sites in Oregon, United States. Stage 0 restoration refers to returning a river corridor to the stage 0 condition in the Cluer and Thorne (2014) stream evolution model. Stage 0 in this model is an anastomosing wet woodland or grassed wetland with a high floodplain water table that intersects the ground surface at least seasonally. Stream restoration to stage 0 conditions involves some combination of introducing in-channel obstacles such as large wood or beaver dam analogues to promote local

aggradation and overbank flow (e.g., Pollock et al., 2014; Dixon et al., 2016) and regrading channel and floodplain topography to facilitate overbank flow and channel migration under an existing flow regime (Powers et al., 2019; Smith et al., 2020). Our case studies come from the South Fork McKenzie River and Deep Creek in Oregon, United States. At both sites large wood has been added and material from high surfaces was graded and used to fill previously straightened channel segments with the intent of facilitating more sinuous and multithread channel planforms. Our primary objective in this paper is to develop a methodology that can be used to identify river reaches with the greatest potential for carbon sequestration. We briefly illustrate the application of this methodology using preliminary data from the two case studies. We acknowledge that evaluating carbon stock at such recently restored sites does not effectively evaluate changes in carbon stock with time following river restoration. Rather, we use the case studies to illustrate the application and potential limitations of the method proposed here.

MATERIALS AND METHODS

Conceptual Model of Factors Controlling Floodplain Carbon Stock

We start with a conceptual model of the factors that influence carbon stock (mass per unit area; e.g., Mg C/ha) in floodplain soil and large wood. We do not address carbon stock in living floodplain vegetation, which typically forms a smaller proportion of total floodplain carbon stock than does soil carbon (Wohl et al., 2012; Hanberrry et al., 2015; Sutfin et al., 2016). We also do not explicitly address human activities in the drainage basin or the river corridor, which can either increase or decrease carbon stock by modifying the variables included in the conceptual model (Wohl et al., 2017).

Floodplain Soil Carbon Stock

Biogeochemists conceptualize wetland carbon sequestration as a balance between carbon inputs and outputs, which can also be done for floodplain soil carbon as a whole (e.g., Wohl et al., 2017). Inputs are primarily carbon within organic matter from plants (autochthonous) and carbon dissolved and suspended in inflowing waters or deposited from hillslope sources during slope wash or mass movements (allochthonous; this can originate from rock weathering as petrogenic carbon or from biotic processes as biospheric carbon; Blattmann et al., 2018). Outputs consist of dissolved and suspended organic carbon in outflowing waters, and CO_2 and CH_4 emissions from microbial decomposition of organic matter (Villa and Bernal, 2018). Organic matter decomposition by microbes can be slowed by 1) scarcity of nutrients that limit microbial growth, 2) physical protection of organic matter contained within soil aggregates, 3) a high proportion of organic compounds that are recalcitrant and difficult for microbes to degrade, 4) cold temperatures that limit plant productivity but also microbial activity, and 5) anaerobic conditions that force microbes to use less efficient metabolic pathways than exist under aerobic conditions (Villa and Bernal, 2018). Although carbon dioxide and methane emissions from

FIGURE 1 | Conceptual model for influences on floodplain soil organic carbon stock. The numbers and letters superimposed on arrows represent the specific geomorphic ("g") and biotic ("b") drivers indicated by each arrow.
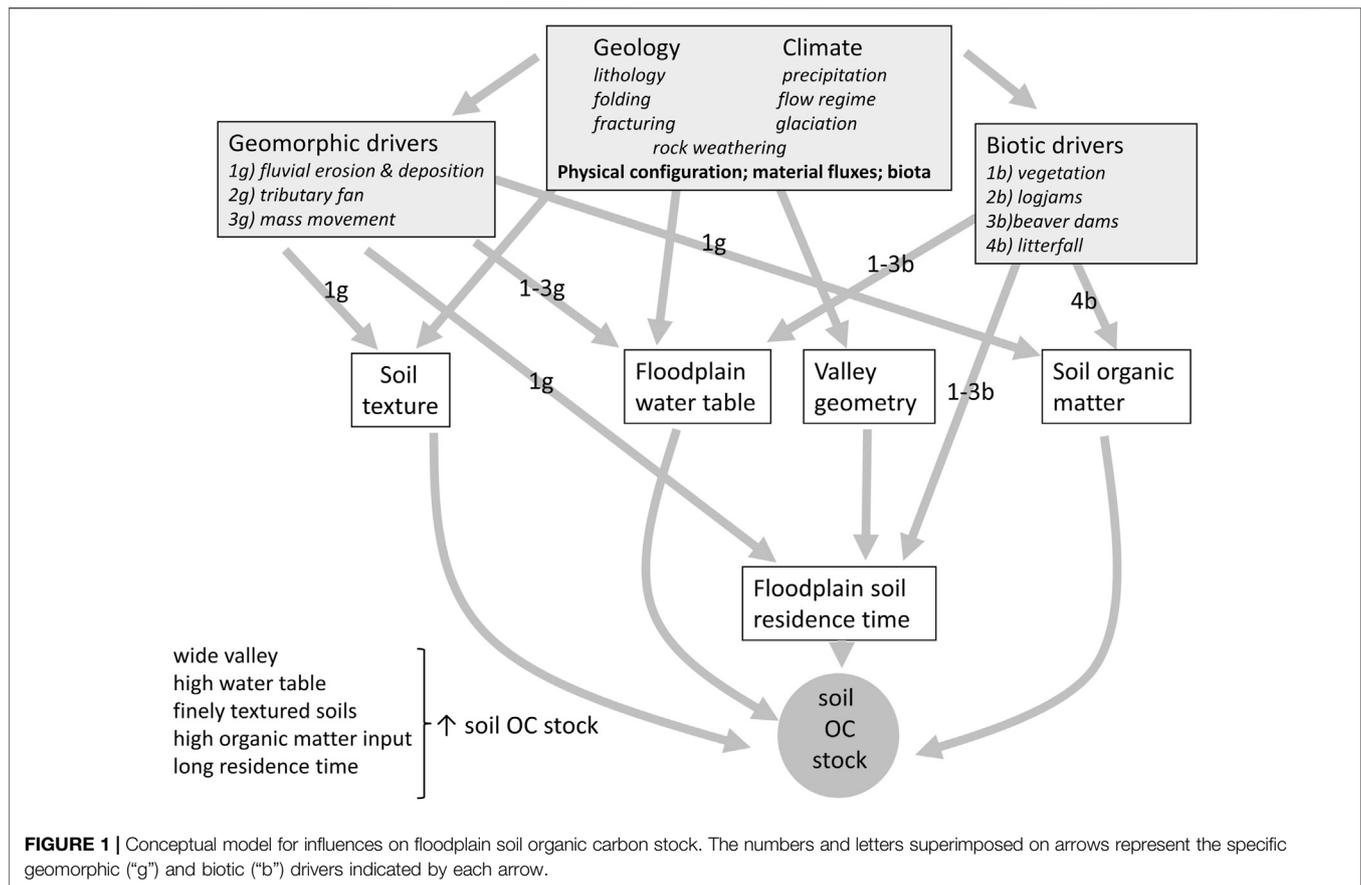
wetlands can form a significant source of wetland carbon loss, particularly as climate warms, the large carbon stocks in wetland soils suggest that naturally functioning (rather than degraded) wetlands in temperate and boreal latitudes typically have a positive carbon balance and sequester carbon over periods of decades to millennia (e.g., Whiting and Chanton, 2001). Consequently, the remainder of this paper focuses on influences on carbon sequestration. When referring to organic carbon present in soil, we are including carbon derived from autochthonous and allochthonous processes. We are using soil to refer to all floodplain sediments.

The carbon concentration of soil (% carbon) depends on multiple factors, including soil texture and the ability of carbon to sorb to soil particles; particulate and dissolved organic matter inputs; and temperature and moisture regimes, which influence microbial oxidation of soil carbon (Kaiser and Guggenberger, 2000; Rasmussen et al., 2018). In general, carbon concentration increases with clay content, organic matter inputs, and moisture, and correlates inversely with temperature (Jobbágy and Jackson, 2000; Falloon et al., 2011; Sutfin et al., 2016).

Starting with these immediate influences on soil carbon concentration, we can infer the reach-scale drivers of clay content, organic matter inputs, soil moisture, and temperature across various temporal and spatial scales.

At the largest scales of time and space, geology and climate interact to govern the physical configuration of the river corridor,

the weathering of bedrock, the fluxes of material into and through the river corridor, and the biotic communities present (upper, central box in **Figure 1**). Geology and climate govern valley geometry and the space available for floodplain storage (physical configuration); as well as soil texture, floodplain water table, and soil organic matter (material fluxes). The influences of geology and climate on these properties are direct but are also mediated by geomorphic and biotic drivers as listed in the second tier of boxes in **Figure 1**. All of the factors mentioned thus far interact to determine the residence time of floodplain sediment. Residence time, along with soil texture, moisture, and organic matter content, determine soil organic carbon stock.

Human activities are not explicitly illustrated in the conceptual model but can alter the forms and processes of stream corridors in manners that influence floodplain carbon stock (e.g., floodplain drainage that lowers the water table and creates oxidizing conditions in floodplain soils).

Geologic characteristics influence soil carbon *via* controls on valley geometry, floodplain water table, and soil texture. 1) Valley geometry. Bedrock structure, glacial history, and the spatial density of bedrock fracturing influence the width and downstream gradient of the valley floor and thus the space available for floodplain development. Structures such as grabens (Koltzer et al., 2019), glacially eroded troughs (Montgomery, 2002; Livers and Wohl, 2015), and dense fracturing that reduces bedrock resistance to weathering and

erosion (Ehlen and Wohl, 2002) promote the presence of a wide, low gradient valley floor at the reach scale and this valley geometry facilitates sediment storage in the floodplain. 2) Floodplain water table. Geologic characteristics can also promote the presence of a high water table, even in dry climates. Examples of such characteristics include structures that retain groundwater inputs from adjacent uplands (Koltzer et al., 2019); folded aquifer units (DesRoches et al., 2014); lithologic contacts between units with differing hydraulic conductivity (Muldoon et al., 2001); resistant lithologies that create a local base level and upstream alluvial depositional zone (Tooth and McCarthy, 2007); undulating bedrock topography that facilitates high water tables where bedrock is close to the surface (Hardie et al., 2012); and fractured bedrock that transmits groundwater readily to topographic lows (Briggs and Hare, 2018). 3) Soil texture. Lithology, in combination with climate and land cover, influences weathering regime and thus the soil texture of the floodplain.

Geomorphic processes influence soil carbon via controls on floodplain water table, soil texture, organic matter input, and sediment residence time in the floodplain. 1) Floodplain water table. Lateral and end moraines can influence groundwater fluxes and valley floor gradient in a manner that promotes a high water table (e.g., Christensen et al., 2020). Similarly, tributary alluvial fans and debris-flow fans, talus slopes, and landslide deposits can influence subsurface and surface water fluxes and create a high water table. Alluvial fans, for example, can serve as groundwater recharge zones, with a line of springs along the base of the fan (Miller et al., 2012). 2) Soil texture. The spatial pattern, rate, and frequency of fluvial erosion and deposition strongly influence the thickness, stratigraphy, and texture of floodplain alluvium (Bridge, 2003). 3) Organic matter input. Overbank deposition of particulate organic matter in transport within the channel reflects upstream sources of organic matter, but also the spatial details of deposition of upstream-sourced organic matter, as governed by hydrologic connectivity (Hupp et al., 2019). 4) Residence time. The balance between lateral and vertical accretion on the floodplain, as a function of active channel dynamics, exerts a fundamental control on the average residence time of sediment on the floodplain (Wittmann and von Blanckenburg, 2009; Wohl, 2015). A longer residence time can equate to higher soil carbon concentrations and stock if organic matter inputs continue through time (Lininger et al., 2018).

Biotic communities influence soil carbon via controls on floodplain water table, organic matter input via litterfall, and sediment residence time. 1) Floodplain water table. Living vegetation, beaver dams, and large wood can increase frictional resistance along channels and create obstructions to flow (Larsen and Harvey, 2010; Collins et al., 2012; Aberle and Järvelä, 2013). This can enhance hyporheic exchanges (Lautz et al., 2006; Sawyer and Cardenas, 2012; Doughty et al., 2020) and overbank flows (Westbrook et al., 2006; Oswald and Wohl, 2008) that result in a higher water table (Larsen et al., 2016). 2) Organic matter input. The primary productivity of floodplain vegetation and the rate of litterfall largely govern the organic matter added to floodplain soils along portions of a floodplain with dense

vegetation and limited surface hydrologic connectivity with the active channel (Lininger et al., 2018; Hupp et al., 2019), although overbank deposition of fluvially transported organic matter can be important on some floodplains. 3) Sediment residence time. Sediment residence time is fundamentally a function of bank erosion and floodplain surface erosion, but these fluvial processes can be strongly mediated by floodplain vegetation, beaver dams, and large wood that increase streambank and floodplain erosional resistance and reduce the hydraulic forces of flow within the channel and across the floodplain (e.g., Micheli and Kirchner, 2002; Perignon et al., 2013).

Floodplain Large Wood Carbon Stock

The carbon concentration of large wood varies by tree species (Martin et al., 2018) but is typically on the order of 50%. The amount of large wood stored on the floodplain reflects a wood budget governed by inputs and outputs. Inputs occur as tree fall on the floodplain, mass movements that introduce large wood to the floodplain, and fluvial transport of large wood onto the floodplain (Wohl, 2020). Outputs occur as decay *in situ* and fluvial transport of large wood out of the floodplain. As for soil carbon, we can start with these immediate influences on floodplain large wood load (m³ wood/ha) and infer the reach-scale drivers of large wood inputs and outputs at larger temporal and spatial scales.

Geology and climate are the primary influences on valley configuration, material fluxes, and biotic communities, as explained in reference to **Figure 1**. Geology and climate also influence the geomorphic and biotic drivers that are particularly relevant to floodplain large wood load (**Figure 2**). In particular, geomorphic and biotic drivers influence the supply of wood to the floodplain and the retention of wood on the floodplains. Again, the conceptual model does not explicitly include human activities, although many human activities (e.g., timber harvest, channelization, beaver trapping) strongly influence floodplain large wood load.

Geology and climate influence wood load *via* controls (weathering and erosion) on valley geometry, which then influences wood supply and wood trapping. The controls on valley geometry are as described for floodplain soil carbon. The width of the floodplain influences wood supply *via* the spatial extent of the floodplain forest and the potential for hillslope mass movements such as avalanches, landslides, or debris flows to directly introduce large wood to the floodplain (Wohl, 2020). The width of the floodplain and the channel planform also influence the retention of large wood moving down the river corridor. Wider reaches with anastomosing channel planform, bars and islands, and sinuous channels can be more effective at trapping and retaining large wood in transport than relatively straight, narrow channels (Gurnell et al., 2000; Wyzga and Zawiejska, 2005; Lassettre et al., 2008; Wohl and Cadol, 2011). Lateral channel migration and avulsion across broad floodplains can leave large wood that is buried and accreted to the floodplain (O'Connor et al., 2003; Guyette et al., 2008; Collins et al., 2012) and then sometimes exhumed and returned to the channel (Benda and Sias, 2003).

FIGURE 2 | Conceptual model for influences on floodplain organic carbon stock in large wood. As in **Figure 1**, the numbers and letters superimposed on arrows represent the specific geomorphic ("g") and biotic ("b") drivers indicated by each arrow.
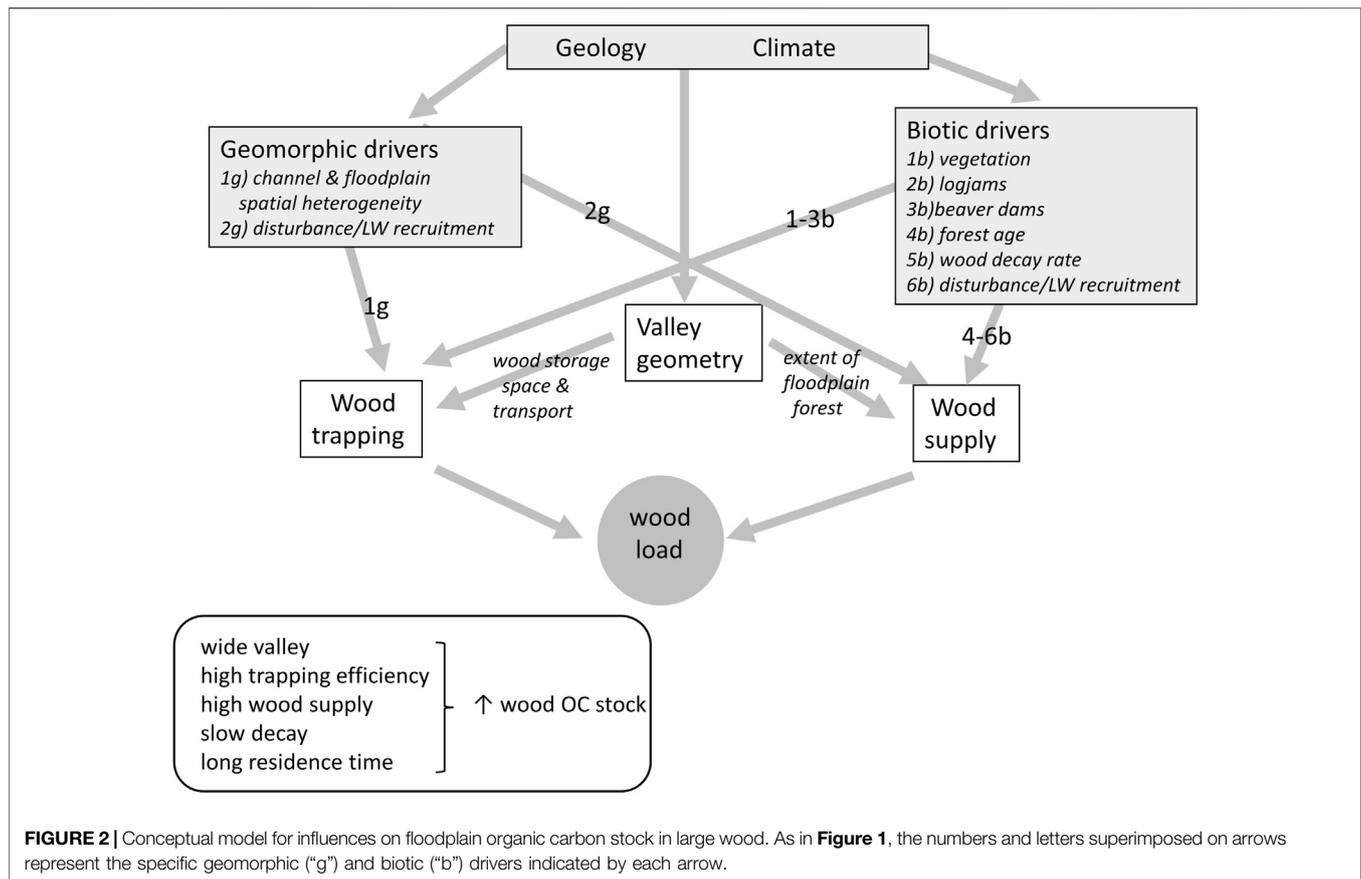
Geomorphic processes influence large wood load *via* controls on wood supply and wood trapping. 1) Wood supply. The supply of large wood to the floodplain partly reflects recruitment of living trees through bank erosion that topples the trees. Although many of these trees fall into the active channel and are carried downstream, others remain on the floodplain. Channel dynamics also indirectly influence wood supply by creating disturbances (erosion, deposition) that can limit the age of floodplain forests and create new germination sites for trees (e.g., Everitt, 1968). 2) Wood trapping. As noted in the description of interactions between valley geometry and wood trapping, the details of channel and floodplain spatial heterogeneity strongly influence how effectively wood being transported downstream in the active channel is trapped and retained on the floodplain (Scott and Wohl, 2018b). Numerous studies indicate that wood preferentially accumulates on geomorphic features such as bars, islands, meander bends, and secondary channels, as noted above, and these wood accumulations can be incorporated into the floodplain *via* lateral accretion.

Biotic communities influence floodplain large wood load through controls on wood supply and wood trapping. 1) Wood supply reflects the age of the forest and the processes of mortality and disturbance that change living trees to downed, dead wood. Individual tree mortality and mass mortality associated with wildfires, drought, insect infestations, and blowdowns can all recruit wood to the floodplain (Wohl, 2020). In regions with very rapid wood decay, however, such as the tropics (e.g., Clark et al., 2002), wood load may remain low despite substantial recruitment. 2) Wood trapping can reflect characteristics of floodplain vegetation, as well as geomorphically induced heterogeneity. Dense herbaceous or shrubby vegetation and closely spaced trees can limit and direct overbank transport of large wood (Wohl et al., 2018a,b) and significantly increase trapping and retention of large wood on floodplains.

Prediction of Floodplain Carbon Stock

Understanding the variables and interactions described in the preceding section can be used to identify reaches within a river network that have the potential for greater floodplain carbon stock in the form of soil carbon and/or large wood. Several sources of information can also be used to quantitatively constrain predictions of floodplain carbon stock. Among these are published values of carbon stock (**Table 1**) and rates (**Table 2**) of carbon inputs in diverse floodplains. This database is limited but steadily growing and at a minimum provides likely upper and lower bounds for floodplain carbon stock in a variety of field settings.

We used the data in **Table 1** to create preliminary illustrations of the magnitude of floodplain carbon stocks in the form of large wood and soil in relation to mean annual precipitation and temperature (**Figure 3**). These are preliminary

TABLE 1 | Review of published values of floodplain large wood and soil organic carbon stocks.

Location	Organic carbon stock (Mg C/ha)	Reference
Large wood		
North St. Vrain Creek, Colorado, United States	166–2,743	Wohl et al. (2012)
Central Yukon River, Alaska, United States	2–11	Lininger et al. (2017)
Congaree River, South Carolina, United States	26–44	Wohl (2011)
Quebec, Canada	57	Naiman et al. (1987)
Central Chile	23–158	Comiti et al. (2008)
SF Calawah River, Washington, United States	67–230	Scott and Wohl (2020)
Tierra del Fuego, Argentina	30	Comiti et al. (2008)
Danube River, Austria	5–40	Cierjacks et al. (2010)
Jalisco, Mexico	13–23	Jaramillo et al. (2003)
Soil		
North St. Vrain Creek, Colorado, United States	60–1,013	Wohl et al. (2012)
Lower Mississippi alluvial valley, United States	167	Hanberry et al. (2015)
Danube River, Austria	154–212	Cierjacks et al. (2010)
Jalisco, Mexico	114	Jaramillo et al. (2003)
Central Yukon River, Alaska, United States	152–402	Lininger et al. (2019)
Congaree River, South Carolina, United States	248–1,118	Ricker and Lockaby (2015)
MF Snoqualmie River, Washington, United States	123–263	Scott and Wohl (2020)
Big Sandy River, Wyoming, United States	57–131	Scott and Wohl (2020)
Headwaters in s New England, United States	117–400	Ricker et al. (2012)
Rhine River, Germany	538–671	Hoffmann et al. (2007), Hoffmann et al. (2009)
Mid-Atlantic Piedmont streams, United States	250–1,350	Walter and Merritts (2008)
MF Flathead River, Montana, United States	7,735	Appling (2012)
Cosumnes River, California, United States	83–182	D'Elia et al. (2017)
Midwestern US and Czech Republic	0.5	Craft et al. (2018)
Tallgrass prairie streams, United States	166–610	Wohl and Pfeiffer (2018)
Shortgrass prairie streams, United States	4–326	Wohl and Pfeiffer (2018)
Queensland, Australia	57–430	Adame et al. (2020)
Dee River, Scotland	323 (34–1,469)	Swinnen et al. (2020)
Mean of published values as of 2017	1.2	Craft et al. (2018)

TABLE 2 | Published values for soil organic carbon accumulation rates on floodplains (after Sutfin et al., 2016; **Table 3**).

Location	Rate (Mg C ha^{-1} yr^{-1})	Reference
Georgia, United States	0.18–1.07	Craft and Casey (2000)
Headwaters in s New England, United States	0.03–0.007	Ricker et al. (2012)
Rhine River, Germany	0.034–0.254	Hoffmann et al. (2007), Hoffmann et al. (2009)
Virginia, United States	0.3–1.4	Noe and Hupp (2009)
Atchafalaya River, Louisiana, United States	8	Hupp et al. (2008)
Tar River, North Carolina, United States	2.8	Brinson et al. (1980)
Southwestern England	0.7–1.1	Walling, (2006)
Danube River, Austria	2.9	Tockner et al. (1999)
Ebro River, Spain	1.4–3	Cabezas et al. (2009)
Kankakee River, Illinois, United States	0.6	Mitsch et al. (1979)
Appalachicola River, Florida, United States	0.2	Mulholland, (1981)
Amazon River, Brazil	1–2.5	Moreira-Turcq et al. (2004)
Queensland, Australia	1.7–12.2	Adame et al. (2020)
Amazon floodplain lake	2.7	Sanders et al. (2017)

versions for several reasons. First, precipitation and temperature are not necessarily the most important controls on either form of floodplain carbon stock, but they do represent variables that are commonly reported or relatively easily accessible for published field site locations. Both floodplain wood load and soil organic carbon concentration can increase with the age of the floodplain surface, for example, as a maturing floodplain forest provides a recruitment source for larger and more abundant downed wood and as litterfall from floodplain plants accumulates in reducing soils (Scott and Wohl, 2018a; Lininger et al., 2018). Similarly, within a spatially extensive floodplain on a large river, local differences in soil moisture can
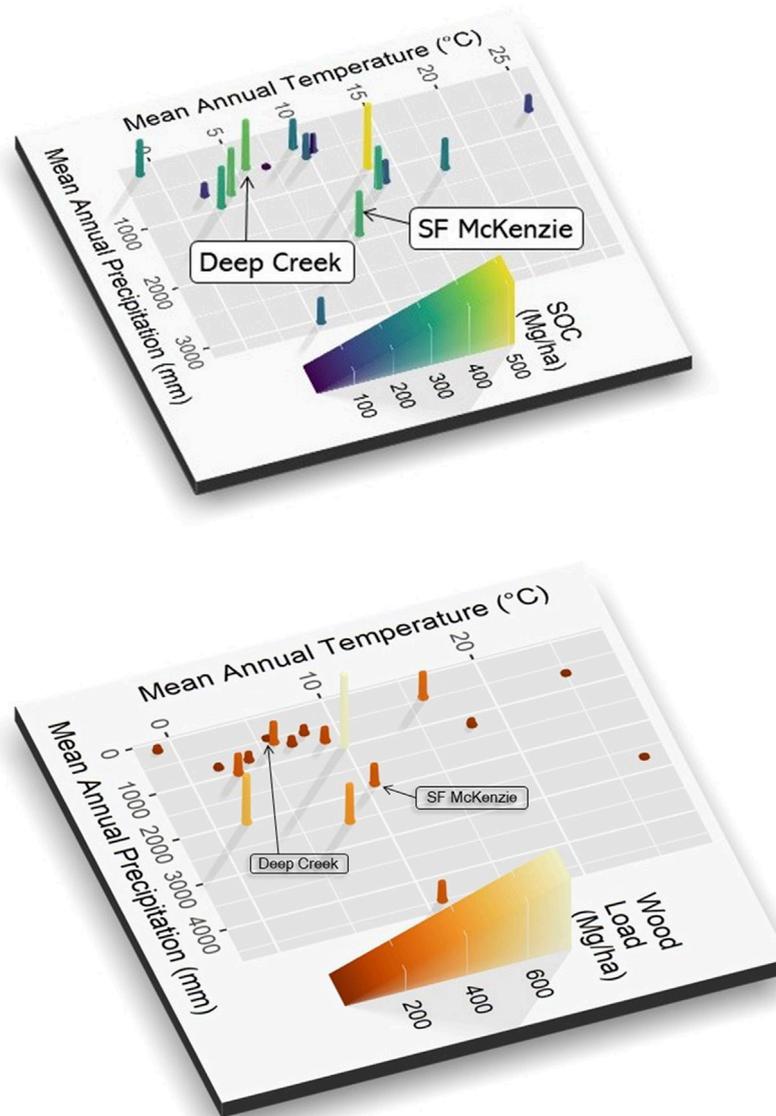
FIGURE 3 | Magnitude of organic carbon stock in floodplain soil (top) and floodplain wood load (bottom). These distributions are based on limited data and thus likely to change as more field sites are characterized. Labels indicate the location of the reference-site values for the case studies in this paper. Data used to generate these plots are in supplementary tables.

equate to significant differences in soil carbon stock, despite a consistent climate across the floodplain (e.g., Lininger et al., 2018). Consequently, differences in the ages or site-specific soil moisture of floodplain surfaces included in the dataset could produce trends in **Figure 3** that might prove to be misleading as more quantifications of floodplain carbon stocks are published. Second, the number of published values is limited and these values do not fully represent the range of floodplain soil carbon or large wood carbon stocks likely to be present along rivers. Finally, we used median values to create a data point for each published case study. Most of the case studies included a range of values that reflect the local variation in floodplain carbon stock. Despite these caveats, we believe that in the absence of

published floodplain carbon stocks for a particular geographic region, the data in **Figure 3** provide a starting point for estimating potential floodplain carbon stocks (**Supplementary Data Tables**).

Existing quantifications of floodplain carbon stocks indicate enormous variation based on site-specific geology, climate, biome, flow regime, and channel and valley morphology, even within a single river network (Sutfin et al., 2016; Scott and Wohl, 2018a; Sutfin and Wohl, 2019). In the context of river restoration, regional reference sites based on the least human-altered portions of a river can help to constrain estimates of the potential for enhanced carbon sequestration at a restoration site. This is the approach that we use in the case studies presented in this paper.

The quantification of carbon stock at reference sites can be based on field sampling or existing databases.

Field sampling for soil organic carbon requires 1) collecting sediment samples that represent the lateral heterogeneity of the site and variation with depth, and analyzing the percent of organic carbon and 2) measuring or estimating soil bulk density (Structx, 2021). These measurements can then be used to calculate Mg C/ha of floodplain, commonly expressed as a volume based on 1 m depth of soil. The 1 m depth is based on the well-documented pattern of declines in organic carbon concentration with increasing depth in soils (e.g., Malone et al., 2009). Previous studies indicate that this depth decline is also common in floodplains (e.g., Lininger et al., 2018), in which most organic carbon comes from autochthonous sources such as litterfall from floodplain vegetation in portions of the floodplain that are farther from the active channel and have limited hydrologic connectivity (Hupp et al., 2019). In the floodplains of smaller rivers or on floodplains that have or are undergoing rapid deposition as a result of human activities such as altering land cover in the drainage basin, carbon-rich soil layers may be buried more deeply (Ricker and Lockaby, 2015; D'Elia et al., 2017). The potential for buried carbon-rich layers to exist in a floodplain should be assessed based on knowledge of site geomorphic and human history, and sampling depth for soil carbon should be modified if needed.

Previous work suggests that stratifying the floodplain into geomorphic units and then obtaining a minimum number of samples to adequately characterize the carbon concentration with depth in each unit (e.g., Lininger et al., 2018) can provide a basis for extrapolating carbon stock across an entire reach based on the spatial extent of each geomorphic unit (Sutfin and Wohl, 2017; Lininger et al., 2019; Scott and Wohl, 2020).

Field sampling for carbon in the form of large wood requires measuring the volume of individual wood pieces or logjams. This can be done for all pieces in the entire reach if the surface area is relatively small or wood loads are low. For larger areas or sites with substantial wood loads, stratified random sampling using floodplain transects (Wohl et al., 2018b) stratified on the basis of valley geometry, floodplain forest age or extent, or other relevant variables can provide sufficient data for extrapolation to the entire floodplain.

Existing databases can also be used to estimate floodplain carbon stock semi-quantitatively. Soil maps, including those available for much of the United States through the SSURGO online database (Soil Survey Staff, 2021), provide representative values for soil texture with depth, total organic carbon, bulk density, and soil thickness for a soil series, and these parameters can be used to calculate soil organic carbon stock (e.g., Wohl and Pfeiffer, 2018). Existing databases for large wood are largely nonexistent, although regional case studies that quantify expected wood loads based on position in the river network or channel width are available for a few locations, such as Washington State in the United States (Fox and Bolton, 2007). Remote imagery obtained *via* a drone, lidar data (Atha and Dietrich, 2016), Google Earth (Atha, 2014), aerial photographs, or other satellite imagery can be used to obtain at least a minimum count of large wood visible from the air. Ideally,

ground measurements of piece diameters can be used to calibrate estimates of wood volume based on piece length, which is most easily measured in remote imagery. Where floodplain wood load is strongly influenced by fluvial transport, rather than just tree fall from the floodplain forest, the spatial distribution of large wood can be highly non-uniform (e.g., Piégay, 1993; Piégay and Gurnell, 1997; Wohl et al., 2018b), suggesting the need to obtain sufficiently large samples to include spatial heterogeneity of wood distribution.

Case Studies
Deep Creek
Deep Creek is located in Ochoco National Forest in the Ochoco Mountain range in central Oregon, United States (elevation 1,325 m) (**Figure 4**). The creek drains 224 km^2 and flows southwest into the North Fork Crooked River immediately downstream of the study area. The catchment is dominated by old-growth ponderosa pine (*Pinus ponderosa*) forest. Average temperature and precipitation in Deep Creek are 5.7°C and 468 mm, respectively. Deep Creek represents a case study for potential carbon sequestration in a river corridor with mixed forest and beaver-modified willow carrs. Past grazing activities and artificial berms caused Deep Creek to incise up to 1 m below the floodplain and led to water table lowering and channel-floodplain disconnectivity. In an effort to restore habitat for Columbia River redband trout (*Oncorhynchus mykiss gairdneri*), Columbia spotted frog (*Rana luteiventris*), and other aquatic species, the U.S. Forest Service regraded the floodplain, added large wood, and reconnected relict secondary channels in 2018 and 2019 (Paul Powers, USDA Forest Service, June 2020, pers. comm.). The regional reference site used as a measure of carbon sequestration potential for Deep Creek is Gray's Creek, located approximately 15 km from Deep Creek and also in the larger North Fork Crooked River watershed. The Gray's Creek floodplain houses an active beaver complex that has been established for approximately 20 years (Jason Gritzner and Paul Powers, USDA Forest Service, June 2020, pers. comm.). We chose a degraded analog for Deep Creek just upstream of the restoration site, where the channel remains relatively incised with rows of alder (*Alnus* spp.) topping elevated surfaces surrounding the single active channel (**Figure 5**). Additional descriptive data for Deep Creek are in **Supplementary Table S1**.

South Fork McKenzie River
The South Fork McKenzie River (drainage area 560 km^2) is located near Rainbow, Oregon in the Willamette National Forest in the western Cascade Mountains (elevation 340 m) (**Figure 4**). With average annual temperature and precipitation of 10.6°C and 1882 mm, respectively, this site represents a case study for carbon sequestration potential in a densely forested river corridor and a Mediterranean climate with significant hydrologic influence from an upstream dam. Common forest species here include western hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and western redcedar (*Thuja plicata*), with red alder (*Alnus rubra*) and cottonwood (*Populus trichocarpa*) adjacent to streams. Following construction of Cougar Dam in 1963, the channel incised to a
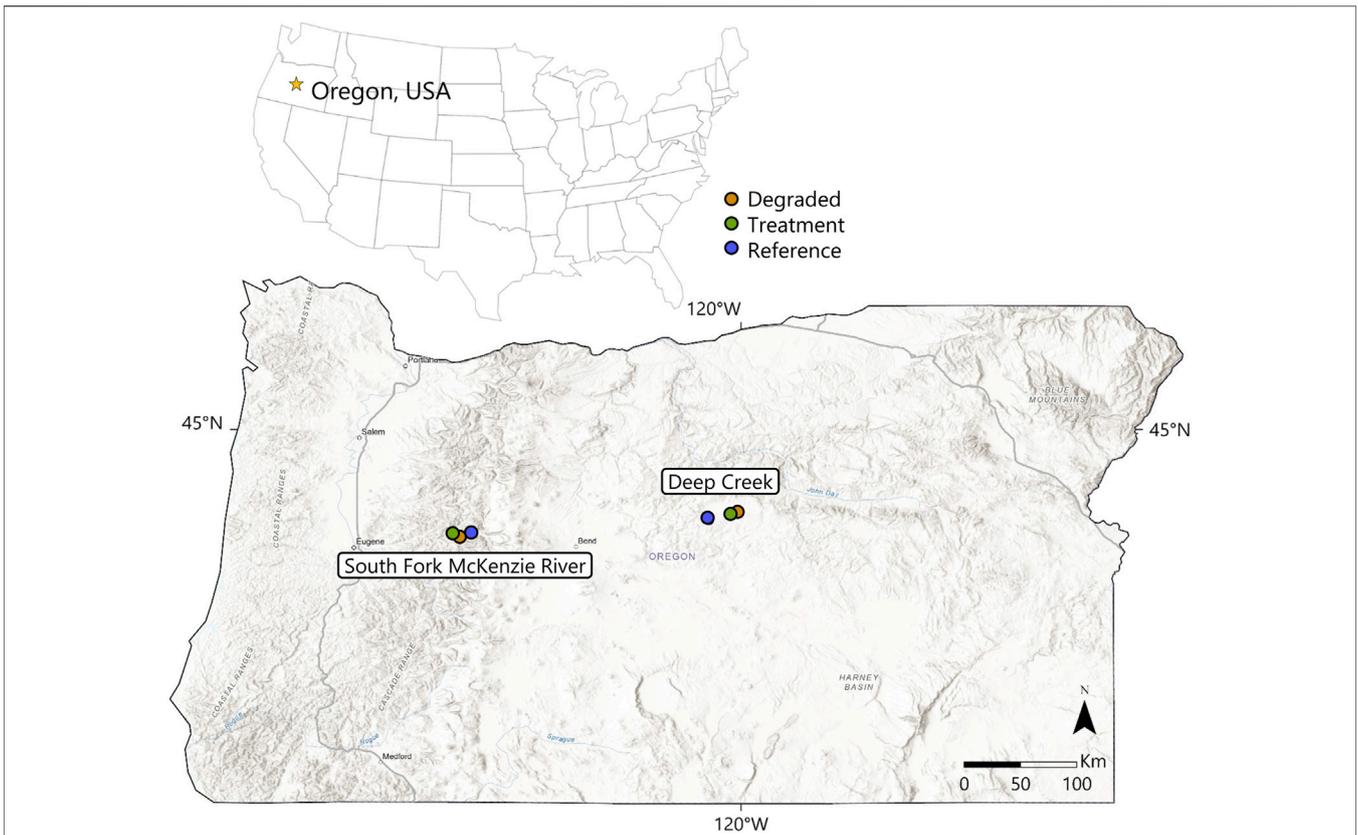
FIGURE 4 | Location map of the sites in Oregon, United States used as case studies in this paper.

FIGURE 5 | Field photos of the South Fork McKenzie River in Willamette National Forest (left panel) and Deep Creek in Ochoco National Forest (right panel), Oregon, United States. Each photo is labeled with its assigned class as degraded, treatment, and design reference ("Reference"), and arrows in the bottom right of each photo indicate flow direction. The person in the left panel is 1.86 m tall and the person in the right panel is 1.58 m tall.

single thread with primarily boulder substrate. Restoration was initiated in 2018 and involved dense placement of large wood across the entire floodplain, lowering of high alluvial surfaces, and filling of the incised channel to reconnect the channel and floodplain and improve habitat conditions for Chinook salmon (*Oncorhynchus tshawytscha*), bull trout (*Salvelinus confluentus*), Pacific lamprey (*Entosphenus tridentatus*), and other species. Horse Creek, approximately 10 km away, provides the reference condition for this site. Horse Creek has an anastomosing planform with one primary and several secondary channels, each with high wood load, frequent channel avulsions, and abundant forested wetlands in the river corridor. We selected a reach of the South Fork McKenzie upstream of the restoration activities for a degraded site. This degraded site is planned to be included in future restoration actions. The degraded reach is single threaded with a pine-dominated forested floodplain and history of channel manipulation and levee construction (**Figure 5**). Additional descriptive data for the South Fork McKenzie are in **Supplementary Table S1**.

Methods

Our sampling design to estimate carbon sequestration potential includes three classes per study area. The three classes, characterized by floodplain condition and inferred history, are referred to as degraded, treatment, and design reference. We assume that degraded sites represent typical pre-restoration conditions where human activities have moved the site beyond the natural range of variability present prior to European settlement of the region (Rathburn et al., 2011; Wohl, 2011). Treatment sites represent conditions of recently implemented stream restoration treatments. Design reference sites represent relatively natural environments with minimal human alteration and carbon stocks that reflect the natural range of variability for a particular type of river corridor. Choice of degraded and design reference sites is subjective. The Deep Creek and South Fork McKenzie River sites are both on U.S. national forest lands and we worked with local Forest Service hydrologists and fish biologists to identify degraded and design reference sites. We also used our knowledge of analogous sites from other portions of the western US.

A beaver-modified willow carr in which beaver are still present is characterized by numerous dams, ponds, and beaver canals in various stages of activity (i.e., active, or abandoned for progressively longer times) (Laurel and Wohl, 2019). The channel and floodplain are hydrologically connected. Abundant surface water storage in ponds (Hood and Bayley, 2008) and a high floodplain water table (Westbrook et al., 2006) create reducing conditions throughout much of the floodplain, even in regions with a dry climate. Spatially dense woody vegetation attenuates downstream fluxes of water and sediment, and beaver dams promote hyporheic exchange flows (Lautz et al., 2006). This configuration, known as a beaver-meadow complex (Polvi and Wohl, 2012), represents design reference conditions for Deep Creek treatment site. In contrast, when beaver abandon a site and their dams fall into disrepair, river flow is more likely to concentrate into a single

channel. Consequently, the channel incises and helps to lower the floodplain water table, hydrologically disconnecting the channel and floodplain and creating an alternative state known as an elk grassland (Wolf et al., 2007). This configuration represents degraded conditions for the Deep Creek treatment site.

A densely forested river corridor, such as that in the South Fork McKenzie River, has abundant recruitment sources for large wood. As described for other rivers in the U.S. Pacific Northwest, the continuing recruitment and storage of large wood facilitates formation of logjams that influence the distribution of hydraulics and sediment transport, commonly leading to a multithread channel planform and spatially heterogeneous floodplain geomorphology and floodplain forest (Fetherston et al., 1995; Collins et al., 2012). This represents design reference conditions for the South Fork McKenzie treatment site. When large wood is actively removed from the river corridor and/or deforestation removes wood recruitment sources, the loss of logjams in the channel and floodplain can lead to formation of a single channel flowing through a less diverse floodplain (Collins et al., 2002). This configuration represents degraded conditions for the South Fork McKenzie treatment site.

Soil Carbon

We anticipated within-reach heterogeneity in soil carbon content based on differences in hydrologic connectivity and soil moisture. Consequently, we differentiated wet and dry floodplain patches based on field conditions when sampling soil. We assigned these moisture conditions based on ground moisture at the time of field work and the type of vegetation present. For example, wetland vegetation such as sedges, rushes, or cattails characterize wet areas and more xeric vegetation such as reed canary grass (*Phalaris arundinacea*), cheatgrass (*Bromus tectorum*), or ponderosa pine (*Pinus ponderosa*) characterize dry areas. We randomized the exact sample location within a wet or dry area when choosing specific sample points.

We sampled soil carbon with a tubular soil corer of 3 cm diameter that collects 30-cm lengths of sample in each increment. At each soil sampling point, we sampled up to 1 m depth or refusal by cobbles in shallower soil. Previous research suggests that sampling bias and variance of organic carbon content cease to decrease after 11 samples within a floodplain along smaller rivers (Sutfin and Wohl, 2017) and within individual floodplain geomorphic units on larger rivers (Lininger et al., 2018). The case studies presented here involve smaller rivers, so we targeted 11 samples per moisture category in each floodplain class. This equates to 11 samples each in wet and dry soils per class per study area, for a total of 22 samples per class and 66 samples per study area. However, floodplains with degraded conditions were commonly hydrologically disconnected and dry. Consequently, fewer wet soil samples were collected from degraded floodplains. Additionally, some samples were lost during lab analysis. In total, we conducted statistical analyses on 59 soil samples from Deep Creek (12 dry and two wet from the degraded class, 11 dry and 13 wet from the treatment class, and 10 dry and 11 wet from the design reference class), and 53 samples from the South Fork McKenzie (11 dry from degraded, 11 dry and 10 wet from treatment, and 10 dry and 11 wet from design reference).

For final floodplain-scale estimation of soil carbon stocks, we weighted the estimated soil OC stocks for each moisture category by the approximate area of wet or dry soil in each floodplain. We determined wet versus dry conditions using National Agricultural Imagery Program (NAIP) aerial imagery from July 2020, the same month field samples were collected. Horse Creek, the design reference site for the South Fork McKenzie River, has too much forest vegetation to confidently estimate wet versus dry soil, so we used topography to estimate areas of wet and dry soil instead. Using a digital elevation model made from LiDAR data collected in 2016 and downloaded from Oregon Department of Geology and Mineral Resources, we assumed that soil below the average relative elevation above a channel surface (detrended by valley gradient) in the sampling area could be categorized as potentially wet. We validated this assumption with our sample locations of wet and dry soil. Through these analyses, we found the existing sample sizes to appropriately reflect area proportions of wet and dry soil, with roughly equal proportions in treatment and design reference floodplains, and little to no wet soil in degraded floodplains.

Soil samples were stored in a freezer and shipped to a commercial laboratory for analysis of organic carbon, total carbon, and texture with a hydrometer. Grain size percentages of silt, sand, and clay were used to categorize soil samples into soil types. Bulk density estimates were assigned based on soil type after evaluation of values according to soil type in comparison to two pedotransfer functions (Leonaviciute, 2000; Ruehlmann and Korschens, 2009), three regression analyses of data from Chaudhari et al. (2013), and a constant density value. Estimates of bulk density were the median result of the seven methods evaluated and offered the most reasonable estimates of carbon stocks. The following formula was used to calculate carbon stocks from percent organic carbon:

$$soil\ organic\ carbon\ stock\ \left(\frac{Mg}{ha}\right)$$
$$= \%OC \times bulk\ density\ \left(\frac{Mg}{m^3}\right) \times 1\ m\ depth \times 10,000\ \frac{m}{ha} \quad (1)$$

Large Wood Carbon

Wood load was estimated via transects following the protocol of Van Wagner, 1968) in the degraded and design reference reaches. Transects were evenly spaced where possible and extended either across the entire floodplain or across one half of the floodplain, depending on accessibility. We measured the diameter of each piece of large wood encountered along transects, and input diameters and lengths into the following equation to estimate volume per area of wood (Van Wagner, 1968):

$$V = \frac{\pi^2 \Sigma d^2}{8L} \quad (2)$$

Where V = volume of wood per area, d = diameter of a piece of large wood >10 cm diameter and >1 m length, L = transect length, and all metrics are in the same units of measurement. Wood volume in treatment floodplains was estimated with timber sale data from managing parties who completed the restoration projects, and from monitoring data in the South Fork McKenzie River (Hinshaw et al., in review). Monitoring data are from 2020 and include large wood

volume measured in 23 plots randomly located in the South Fork McKenzie River treated floodplain. We calculated carbon stocks of large wood using average wood densities of the dominant tree species in each study area and the assumption that wood contains 50% organic carbon:

$$large\ wood\ organic\ carbon\ stock\ \left(\frac{Mg}{ha}\right)$$
$$= wood\ load\ \left(\frac{m^3}{ha}\right) \times 50\%\ OC \times wood\ density\ \left(\frac{Mg}{m^3}\right) \quad (3)$$

For the South Fork McKenzie River, we averaged densities for the five dominant tree species mentioned above. Ponderosa pine density was used for Deep Creek degraded and treatment sites, and willow density was used for the Deep Creek design reference floodplain.

Statistical Analyses

We used a type 3 ANOVA to determine whether there are differences between classes for each study site using a linear model of soil organic carbon stock predicted by class (degraded, treatment, and design reference). R version 4.0.3 was used to conduct the analysis with the car package (Fox and Weisberg, 2019; R Core Team, 2020). Separate models were fitted for the two study sites and two models were fit for Deep Creek: one model with class as the sole predictor, and one that included moisture. We used the emmeans package to conduct Tukey-adjusted pairwise comparisons between classes groups at the 95% confidence level (Lenth, 2020). Moisture was not included in the model for the South Fork McKenzie River because there were no wet samples in the degraded reach. Depth and texture, other potential predictors of carbon stocks, were not included because we assume they are adequately integrated into the calculation of soil organic carbon stocks. Moisture was included in a model for Deep Creek, but the sample size of $n = 2$ wet samples in the degraded reach precludes reliable interpretation of comparisons of wet samples from the degraded class to other classes.

RESULTS

We first tested whether there were significant differences in percent organic carbon in relation to moisture category (wet vs dry) at each of the six sites. We found significant differences at only two of the sites: percent organic carbon is significantly higher in 1) wet soils of the design reference site for South Fork McKenzie (p value 0.0275) and 2) dry soils for the treatment site at Deep Creek (p value 0.036).

We then compared the six sites based on organic carbon stock. There were no significant differences between classes for Deep Creek when class was the sole predictor of soil organic carbon stock (**Figure 6**). When moisture and class were used to predict soil OC stocks, we found evidence of an interaction between class and moisture ($p = 0.04697$). Among dry samples, there was a notable, but not statistically significant, difference between degraded and treated floodplain carbon stocks ($p = 0.0782$), and a significant difference between treated and design
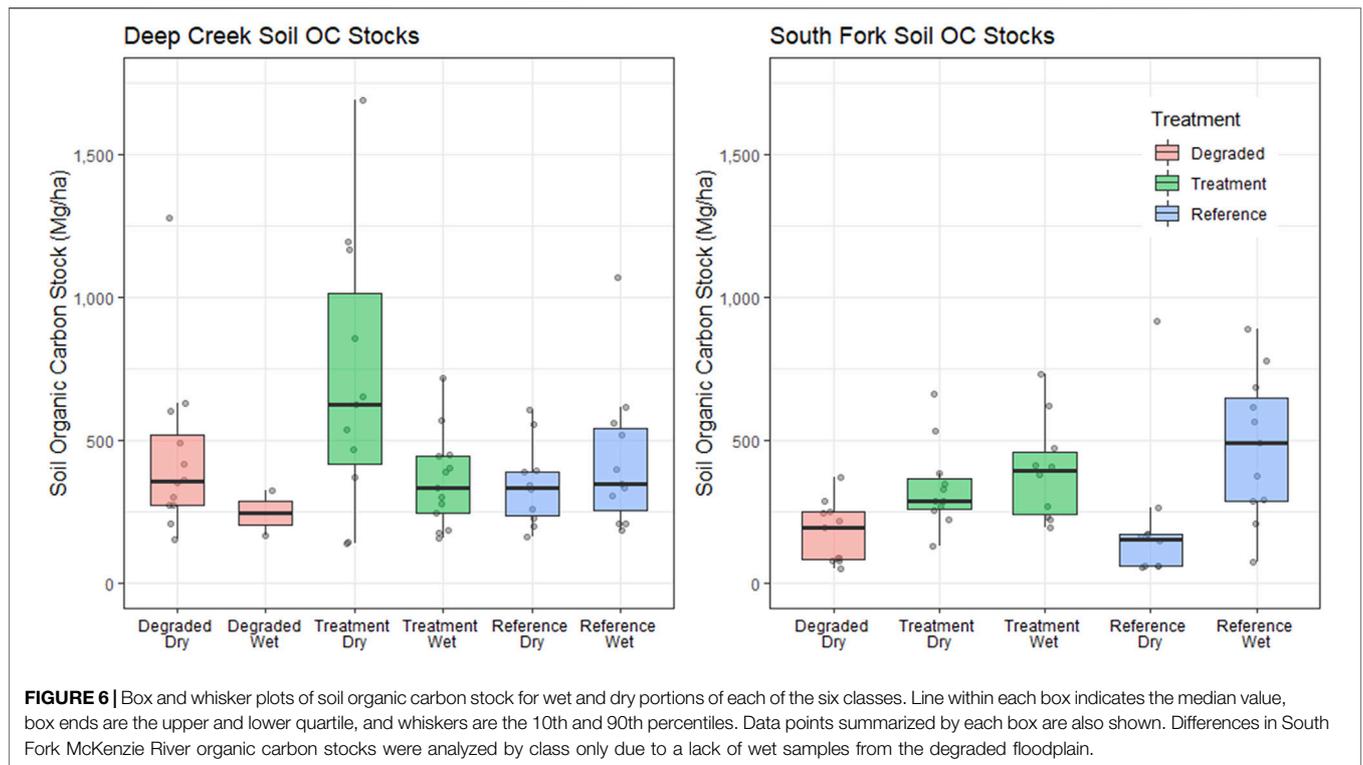
FIGURE 6 | Box and whisker plots of soil organic carbon stock for wet and dry portions of each of the six classes. Line within each box indicates the median value, box ends are the upper and lower quartile, and whiskers are the 10th and 90th percentiles. Data points summarized by each box are also shown. Differences in South Fork McKenzie River organic carbon stocks were analyzed by class only due to a lack of wet samples from the degraded floodplain.

TABLE 3 | Estimated carbon stocks in large wood and soil at the case study sites.

Site and treatment	LW OC stock (Mg C/ha)	SOC stock (Mg C/ha)	Total OC stock (Mg C/ha)
Deep Creek			
degraded	21	416	437
treatment	7	521	528
reference	1	391	392
SF McKenzie			
degraded	39	177	225
treatment	136	364	500
reference	46	348	394

reference floodplain soil OC stocks ($p = 0.0150$). The estimated difference between treated and degraded dry floodplain OC stocks is 268 Mg/ha, with higher estimated carbon stocks in treated floodplains. Our data in Deep Creek showed higher estimated OC stocks in dry treated than dry design reference conditions, with an estimated difference of 367 Mg/ha.

At the South Fork McKenzie site, there were no significant differences between class at the 95% confidence level, but there was a nonsignificant but notable estimated difference of 187 Mg/ha in mean values for OC stocks between the treated and degraded floodplains, with higher predicted soil OC stock in the treated floodplain ($p = 0.0572$). There was also a notable but nonsignificant difference between the degraded and design reference floodplains for wet and dry combined ($p = 0.0886$), with estimated mean OC stocks in the design reference site predicted to be higher than the degraded site by 171 Mg/ha.

Estimates of large wood load and large wood carbon stocks varied depending on the method used and are presented in **Table 3**. Total estimated OC stocks, estimated by the sums of the simple mean soil OC stock of each class and large wood OC stocks, were greatest for the treated floodplain reaches in both study areas, and are also presented in **Table 3**. Basic organic carbon data are in **Supplementary Table S3**.

DISCUSSION AND CONCLUSION

Our conceptual framework for quantifying carbon sequestration potential in river restoration involves the use of regional design reference and degraded floodplains for comparison to treated floodplains. The difference in measured carbon stocks between a degraded and design reference floodplain conceptually represents

potential minimum differences in carbon storage that could be achieved through restoration actions, and measured carbon stocks in treatment areas indicate further potential for carbon storage. We illustrated application of the conceptual framework using two study areas, for both of which our results suggest higher organic carbon stocks in the treated areas than in their respective degraded and design reference sites. We also estimated higher carbon stock in the degraded reach of the Deep Creek study area than in the design reference site.

The importance of large wood as a contributor to total carbon stock varies in relation to valley morphology and the presence of wood-trapping sites (**Figure 1B**), and concentration of large wood in limited areas (e.g., beaver dams or very large logjams) can make it difficult to accurately estimate total wood load. We estimated the smallest carbon stocks in the form of large wood at the Gray's Creek site, for example, which is an active beaver meadow complex and design reference floodplain for Deep Creek. We expected a low wood load in the anastomosing grassed wetland of Gray's Creek, but the transect method of sampling large wood did not cross any beaver dams at this site, which may have caused us to underestimate wood load in the reach. Disproportionately high concentrations of wood are also located in large logjams at Horse Creek, the design reference floodplain for South Fork McKenzie. Because only one of five transects at Horse Creek crossed a logjam, the transect sampling method may have caused us to underestimate wood load at Horse Creek. In addition, high surface soil moisture and associated wood decay in Horse Creek likely kept us from measuring some buried, partially buried, decayed, and moss-covered pieces of wood from our transect surveys. Finally, the use of timber sale and project design data to estimate wood load in the treated Deep Creek floodplain likely led to an underestimate. Project design data provide an easy way to estimate wood loads in restoration reaches, but do not include pieces recruited since project construction.

Floodplain manipulation and other land use histories also influence large wood loads. The highest wood carbon stocks of our case studies occurred in the treated area of the South Fork McKenzie River. As part of the project implementation, large wood was placed in a lattice across the floodplain to create hydraulic roughness and encourage deposition. The use of large wood in restoration practices, such as the Stage 0 project at the South Fork McKenzie, increases the large wood carbon stocks in treated floodplains. At the degraded reach of the South Fork McKenzie, cut logs also increased measured wood loads in the study area. We observed that many of the logs measured along transects in the degraded site were cut and accompanied by burn piles, presumably due to forest management practices such as hazardous fuel control. These cut logs increased our measured wood loads, but have separate, more static, geomorphic roles than large wood that more regularly interacts with water and sediment in dynamic floodplains such as Horse Creek.

We interpret the range of carbon stocks measured at all three sites in each case study as reflecting the potential range of floodplain carbon storage for the geographic and geomorphic setting of each case study. As described in the conceptual model, numerous factors interact to govern floodplain carbon accumulation rates through time and carbon stock at any point in time. This makes it both difficult to precisely predict floodplain carbon storage and difficult to use degraded and design reference reaches to demonstrate the effects of river restoration on carbon storage over short timespans of a few years. However, it is encouraging that the values of wood load and soil organic carbon at each of the design reference sites seem reasonable relative to preliminary carbon storage plots and summary tables developed from published values of floodplain large wood loads and soil carbon (**Figure 3** and **Supplementary Material**). This suggests that the type of data reported in **Table 1** and **Figure 3** may eventually be useful in assessing potential floodplain carbon stocks following restoration, when the available data are expanded to include a larger number of sample sites across a wider range of climate and geomorphic conditions.

The optimal timeframes for monitoring organic carbon stock after restoration are not presently known but would presumably be at least several years to decades after restoration activities cease (e.g., Schiefer et al., 2018). This would allow newly (re)saturated floodplain soils time to accumulate organic matter. It would also allow time for large wood introduced to the site during restoration or trapped by the enhanced spatial heterogeneity of the restored site to accumulate in natural trapping sites such as bars, islands, or forested floodplain margins. Depending on the size and spatial heterogeneity of the restored floodplain, we suggest that floodplain soil-carbon sampling be based on a stratified random design that, at a minimum, differentiates wet and dry floodplain surfaces and for larger rivers differentiates geomorphic patches (e.g., Lininger et al., 2018). Floodplain large-wood-carbon sampling using survey transects can also be based on a stratified random design where wood is known to be concentrated in large accumulations, including beaver dams.

Candidate sites for restoration are commonly chosen based on known or inferred historic conditions prior to excessive human disturbance and degradation. More specifically, in process-based and Stage 0 stream restoration, sites chosen for restoration activities are commonly wide, depositional valleys. This type of valley can be inferred, based on historic aerial imagery, local knowledge, relict hydric soils or floodplain landforms, to have historically possessed highly complex planforms with shallow groundwater tables, high degrees of lateral hydrologic connectivity, high wood loads, and floodplain wetlands. Abandoned beaver meadows are capable of retaining high carbon stocks for decades after abandonment (Laurel and Wohl, 2019). High measured soil carbon stocks of treatment floodplains in our case studies may primarily reflect past conditions, rather than the effects of recent restoration. In particular, the highest sampled soil carbon stocks from the Deep Creek treated floodplain (**Figure 5**) are from topographically high floodplain patches (referred to as "leave islands" during restoration) with old-growth ponderosa pines,

suggesting the potential for prolonged organic matter input and accumulation.

There are many benefits associated with quantitatively estimating carbon stocks in degraded, treated, and design reference floodplains. We propose this conceptual model and study design as an example for practitioners, managers, scientists, investors, and other stakeholders to track changes of floodplains over time, collect baseline data in pre-treatment degraded areas, and build databases of design reference conditions that can be available to future stream restoration designers. At a broader scale, rivers play a role in the global carbon cycle, but the definition and quantification of carbon sequestration potential in rivers remains largely unknown. We suggest that the type of quantitative estimations summarized here can be used to enhance the use of river restoration as a tool for carbon sequestration.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/**Supplementary Material**, further inquiries can be directed to the corresponding author.

ETHICS STATEMENT

Written informed consent was obtained from the individual(s) for the publication of any potentially identifiable images or data included in this article.

AUTHOR CONTRIBUTIONS

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/feart.2021.708895/full#supplementary-material

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