Carbon dynamics of river corridors and the effects of human alterations

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Abstract. Research in stream metabolism, gas exchange, and sediment dynamics indicates that rivers are an active component of the global carbon cycle and that river form and process can influence partitioning of terrestrially derived carbon among the atmosphere, geosphere, and ocean. Here we develop a conceptual model of carbon dynamics (inputs, outputs, and storage of organic carbon) within a river corridor, which includes the active channel and the riparian zone. The exchange of carbon from the channel to the riparian zone represents potential for storage of transported carbon not included in the "active pipe" model of organic carbon (OC) dynamics in freshwater systems. The active pipe model recognizes that river processes influence carbon dynamics, but focuses on CO_2 emissions from the channel and eventual delivery to the ocean. We also review how human activities directly and indirectly alter carbon dynamics within river corridors. We propose that dams create the most significant alteration of carbon dynamics within a channel, but that alteration of riparian zones, including the reduction of lateral connectivity between the channel and riparian zone, constitutes the most substantial change of carbon dynamics in river corridors. We argue that the morphology and processes of a river corridor regulate the ability to store, transform, and transport OC, and that people are pervasive modifiers of river morphology and processes. The net effect of most human activities, with the notable exception of reservoir construction, appears to be that of reducing the ability of river corridors to store OC within biota and sediment, which effectively converts river corridors to OC sources rather than OC sinks. We conclude by summarizing knowledge gaps in OC dynamics and the implications of our findings for managing OC dynamics within river corridors.

Key words: active channel; carbon; dam; land use; riparian zone; river.

INTRODUCTION

We propose a conceptual framework that integrates the active channel and riparian zone and use this framework to examine organic carbon inputs, outputs, and storage within river networks. We believe that this framework most effectively illuminates the important influences that river process and form exert on organic carbon fluxes.

As recently as 2001, the Intergovernmental Panel on Climate Change (IPCC) report included a conceptual model of the global carbon cycle with only two biologically active boxes, the land and the oceans, connected through freshwater transport of carbon from the land to the oceans and through gas exchanges with the atmosphere (IPCC 2001). Research during the 1970s and 1980s demonstrated that rivers deliver significant amounts of terrestrial carbon derived from soils and vegetation to the ocean (e.g., Schlesinger and Melack 1981), but rivers were only included in early conceptual models of the carbon cycle as neutral or passive pipes for carbon transport (Cole et al. 2007). In the past decade, we have learned that much more terrestrial carbon enters rivers than is transferred to the ocean, indicating that rivers are active pipes for carbon transport (Cole et al. 2007). However, research into rivers as active pipes in the carbon cycle has focused primarily on losses of carbon via gas exchange in all freshwater habitats (Raymond et al. 2013) and via storage of carbon in lakes and reservoirs

(Tranvik et al. 2009). Carbon exchanges in riparian wetlands and floodplains are commonly treated separately. Much less known is the role of carbon storage in river channels and floodplains. Rivers can store organic-rich sediment within the active channel in bars, beneath the active channel, particularly in an aggrading river segment, and in floodplains, deltas, alluvial fans, and other portions of the river network typically dominated by deposition and storage over time spans of $10^1–10^3$ yr (Walter and Merritts 2008, Hoffmann et al. 2009, Cierjacks et al. 2010, Wohl et al. 2012b, Sutfin et al. 2016). Together, research in stream metabolism, gas exchange (e.g., Raymond et al. 2013, Hotchkiss et al. 2015), and sediment dynamics (Wohl et al. 2015) indicates that rivers are an active component of the global carbon cycle, rather than neutral pipes, and that river form and process can significantly influence partitioning of terrestrially derived carbon among the atmosphere, geosphere, and oceans (Aufdenkampe et al. 2011). However, most conceptual expressions of the global carbon cycle have not yet integrated such a view.

The lack of appropriate integration of rivers into global carbon models may partly reflect differing definitions of what constitutes a river. Here, discussion of carbon dynamics in river contexts includes inputs, outputs, and storage of diverse forms of organic carbon (OC) that occur within a river corridor. We define a river corridor as including the active channel, hyporheic zone, floodplain, riparian zone, and river depositional landforms such as deltas and alluvial fans (Ward 1989, Ward and Tockner 2001, Harvey and Gooseff 2015, Fig. 1). For simplicity, we distinguish the active channel and the riparian zone as primary components of the river corridor in discussing carbon dynamics. The active channel is a primarily erosional feature that routes water through the landscape at least episodically within defined banks: this is the portion of a river corridor that stream ecologists are most likely to think of as a river. The riparian zone includes the hyporheic zone, floodplain, delta, and alluvial fan, where these features are present. The riparian zone can be subject to erosion and deposition over differing time periods and spatial scales, but if an alluvial floodplain created and maintained by river processes is present, the riparian zone is a depositional environment over time spans of at least $10^1–10^2$ yr, and potentially 10^3 yr (Mertes et al. 1996, Wohl 2015).

The second State of the Carbon Cycle Report (SOCCR-2) will differentiate inland waters, types of wetlands, and types of terrestrial ecosystems into separate accounting units (USCCSP 2016). The conceptual model we present here takes a more holistic view of rivers and integrates into the river corridor the inland water, wetlands, and terrestrial components of the landscape that are directly influenced by river processes.

Although we are in the early stages of understanding the details of how river form and process influence carbon dynamics, human activities clearly have altered and continue to alter most aspects of sediment and carbon dynamics within river corridors (Hoffmann et al. 2010, Regnier et al. 2013, Wohl et al. 2015). People indirectly alter carbon dynamics within river corridors by changing inputs of OC to river networks from the uplands through changes in land cover and topography. People also directly alter the ability of river corridors to process, store, and release (downstream and to the atmosphere) OC. Direct anthropogenic modifications of river corridors that influence carbon dynamics include flow regulation, channelization, artificial levees, floodplain agriculture
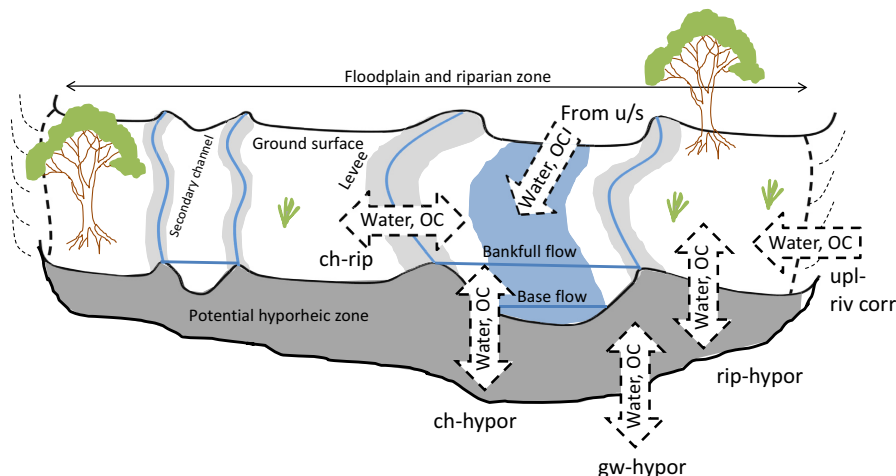

Fig. 1. Schematic illustration of the components of a river corridor referred to in this paper. The active channel includes the bankfull area of the main channel and any secondary channels present. The riparian zone includes the entire hyporheic zone, and the floodplain and riparian area. Gray shading indicates the potential extent of the hyporheic zone, only a portion of which would likely be active at any given time. Groundwater, which is not shown in this view, lies below and laterally beyond the hyporheic zone. Arrows with dashed outlines indicate primary flux directions: from upstream (from u/s), upland-river corridor (upl-riv corr), riparian-hyporheic (rip-hypor), groundwater-hyporheic (gw-hypor), channel-hyporheic (ch-hypor), and channel-riparian (ch-rip). Flux to downstream is not shown but also exists. OC, organic carbon.

Concepts & Synthesis

and urbanization, and many other activities. Although these human-induced alterations are ubiquitous throughout river networks in temperate latitudes, they have received very little attention in the context of carbon dynamics. However, human alterations of river networks create a context in which to understand how rivers transport and store C at varying timescales.

The effects of humans on OC dynamics in river corridors are numerous and varied, and the net effect of human activities differs between channel and riparian areas. We contend that most syntheses of river carbon dynamics fail to account for (1) bi-directional fluxes between the channel and riparian zone, (2) how the processes underlying these bi-directional fluxes influence OC dynamics, and (3) how human activities have altered these processes and the resulting carbon fluxes. Severing or disrupting the linkages between the channel and riparian zone may provide a key mechanism for altering carbon storage in rivers.

Our primary objectives in this paper are to review the existing knowledge of carbon dynamics in river corridors, to develop a conceptual model of how river form and process influence carbon dynamics, and to draw on this model to infer how human activities alter carbon dynamics within river corridors. An important aspect of our review is to integrate the disparate views of river corridors and of carbon dynamics held by diverse river scientists. For example, ecologists are more likely to view rivers as active channels and to investigate OC dynamics over relatively small temporal and spatial scales (e.g., annual efflux of CO_2 from a stream reach). River biogeochemists and hydrologists view rivers as continuous reactors with defined water and inorganic and organic carbon inputs and outputs, as well as biogeochemical transformations along surface and subsurface flow paths. Geomorphologists are likely to view rivers as coupled channel–floodplain systems and to investigate processes affecting OC dynamics (e.g., flux and storage of OC associated with sediment dynamics) over much larger temporal and spatial scales.

Our conceptual model of carbon dynamics within a river corridor takes the form of an OC budget that links OC pools and fluxes (Fisher and Likens 1973); shows how human activities have altered pools and fluxes; and estimates the relative magnitude of each pool, flux, and alteration. We start by briefly reviewing knowledge of the inputs, outputs, and storage terms in the OC budget. We then identify the pools, fluxes, and alterations that have been quantified and those for which relatively little is known. We show that the morphology and processes of a river corridor drive to a large extent the ability to store, transform, and transport OC, and that people are a pervasive changer of river morphology and processes that regulate these dynamics. We propose that dams create the most significant alteration of carbon dynamics within the active channel, but that alteration of riparian zones, including the lateral connectivity between the channel and riparian zone, constitutes the most significant and most highly altered aspect of carbon dynamics in river

corridors. Finally, we use these assessments to discuss implications for management of river corridors in the context of carbon dynamics.

Carbon Dynamics in Natural River Corridors

We conceptualize carbon dynamics in a river corridor as a carbon budget in the form of

$$\frac{dC_S}{dt} = C_I - (C_{Ogas} + C_{Oriver}) \tag{1}$$

in which C_S is carbon storage, t is time, C_I represents carbon inputs, C_{Ogas} represents gaseous carbon outputs, and C_{Oriver} represents fluvial outputs to downstream portions of the river corridor and the ocean (Fisher and Likens 1973).

The forms of inputs, outputs, and storage differ between channel and riparian components of the river corridor. In the active channel, inputs include OC in dissolved and particulate form (C_{IOC} in Eq. 2) from upstream, upland, groundwater, and riparian sources; CO_2; and net primary productivity within the channel. Outputs include emissions (C_{Ogas} in Eq. 2) and dissolved and particulate organic carbon fluxes (C_{Oriver}) downstream. Storage takes the form of downed, dead wood, particulate and dissolved organic carbon within the channel, and aquatic biomass (Fig. 2)

$$\frac{dC_S}{dt} = (C_{IOC} + C_{ICO_2} + C_{INPP}) - (C_{Ogas} + C_{Oriver}) \tag{2}$$

In the riparian zone, inputs include dissolved and particulate OC from upland, groundwater, and channel sources and net primary productivity within the riparian zone (Eq. 3). Outputs include plant respiration and soil respiration and OC decomposition, as well as OC and CO_2 entering the channel via surface and subsurface pathways (Eq. 3). Storage takes the form of large wood (pieces ≥ 10 cm diameter and 1 m length), smaller wood pieces, plant organic matter (litter), decomposing organic matter (duff), soil organic carbon, and aboveground biomass. Numerous exchanges occur between the channel and riparian zone (Fig. 2).

$$\frac{dC_S}{dt} = (C_{Iupl} + C_{Iriver} + C_{Igw} + C_{INPP}) - (C_{Ogas} + C_{Oriver}) \tag{3}$$

In the following sections, we summarize existing knowledge of inputs, outputs, and storage, and then summarize variations in OC dynamics across space and time. We focus on organic carbon.

Inputs

Fluvial carbon inputs to river corridors reflect terrestrial sources from adjacent uplands and upstream
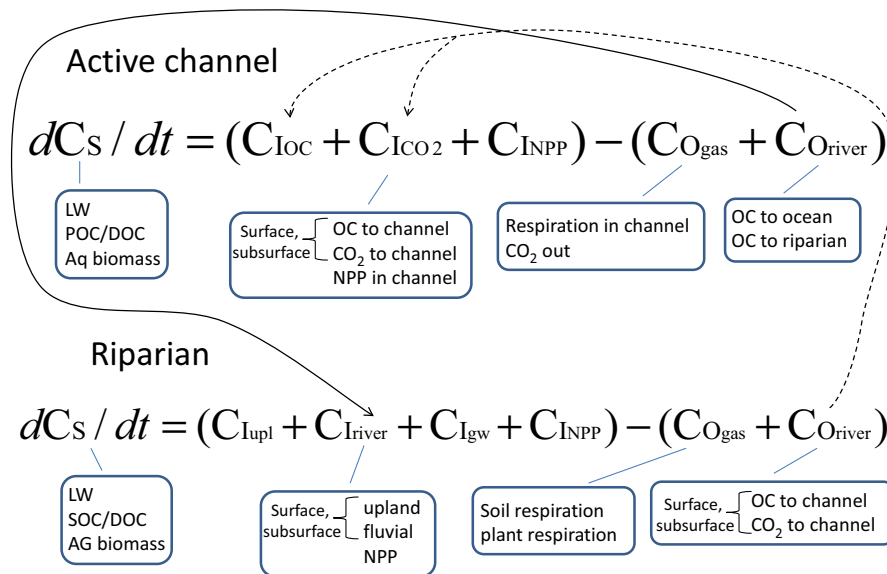
Fig. 2. Schematic illustration of the forms of carbon storage (C_S), inputs (C_I), and outputs (C_O) as gas or river fluxes within the active channel and riparian areas (fluxes between the two compartments indicated as dashed or solid arrows). LW, large wood; POC, particulate organic carbon; OC, organic carbon; NPP, net primary productivity; SOC, soil organic carbon; Aq, aquatic; AG, aboveground vegetation biomass; upl, upland; gw, groundwater.

portions of the river corridor, as well as longitudinal and lateral sources within the river corridor (Fig. 2). Upland carbon inputs to river corridors originate from bedrock, soils, and vegetation (Regnier et al. 2013). Fossil carbon eroded from bedrock via processes such as deep-seated landslides can be a significant source of dissolved inorganic carbon (DIC) and OC inputs to some river networks (Alin et al. 2008, Galy et al. 2008a). Soils, however, appear to be the major source of terrestrially derived OC in most river networks (Lal 2004, Fischlin et al. 2007, Hilton et al. 2008a, Bouillon et al. 2009, Gomez et al. 2010). The importance of soils as a terrestrial carbon source to river corridors is reflected in correlations across diverse river networks between fluxes of POC and suspended sediment eroded from upland soils (Galy et al. 2015), as well as progressively older POC in ecosystems carrying higher sediment loads (Marwick et al. 2015). Progressively older POC with rising discharge in Arctic rivers reflects a shift from aquatic (in situ productivity) to terrestrial sources (McClelland et al. 2016), although terrestrial in this context includes erosion of river banks, which we define as being within the river corridor. Much of the literature on POC and DOC sources to flowing water considers active channels as aquatic and all other sources as terrestrial.

Soil-derived OC can reach rivers as dissolved organic carbon (DOC), particulate organic carbon (POC), and free dissolved CO_2 from soil respiration (Sarmiento and Sundquist 1992). Episodic inputs of soil and trees associated with extreme storms that induce widespread shallow landslides can be important (Hilton et al. 2008b, 2011b, West et al. 2011, Ramos Scharrón et al. 2012,

Wohl and Ogden 2013), but remain a poorly quantified component of terrestrial carbon inputs. Significant amounts of soil-derived OC can also enter rivers each time rain falls, even in the absence of landsliding (Smith et al. 2013). Carbon inputs via groundwater can be particularly important in headwater streams (Johnson et al. 2008).

Carbon inputs also originate within river corridors as a result of in situ net primary productivity and photosynthetic carbon fixation by aquatic and riparian biota. The river continuum concept (RCC) posits that the ratio of gross primary productivity (P) to respiration (R) in shaded headwater streams is <1, where OC inputs from riparian biota are particularly important. The P:R ratio increases as autochthonous primary production increases downstream, but then again declines farther downstream as turbidity and depth limit primary production (Vannote et al. 1980). Data for this pattern show much higher variability than suggested by this simple model, and the continuum can be disrupted by tributary confluences (Ward and Stanford 1983). Low-order streams, however, can have much lower P:R than midsized rivers (Hall et al. 2016). In very large rivers, lateral OC inputs from riparian forests can exceed those from upstream sources (Sedell et al. 1989), particularly during the waning stages of the seasonal flood pulse (e.g., the flood-pulse concept; Junk et al. 1989) but also during other portions of the year (e.g., the river productivity model; Thorp and Delong 1994). Recent investigation of OC dynamics within large tropical rivers indicates that inputs of carbon from floodplain wetlands are major contributors to channels (Borges et al. 2015).

Isotopic signatures of DOC and POC show strong seasonal variations and indicate that the origins of these forms of OC can vary throughout a year and are largely uncoupled (Bouillon et al. 2012). On a large tropical river tributary to Africa's Congo River, for example, POC comes primarily from in situ phytoplankton production during the low-flow season, whereas topsoil and forest litter are primary POC contributors during high flows, and DOC isotopic signatures suggest at least three distinct sources that vary in importance seasonally (Bouillon et al. 2012). DOC composition in large Arctic rivers suggests substantial contributions of aged DOC from mosses and peat bogs during low flow and greater inputs of more recently produced DOC during high flow (Amon et al. 2012).

Whatever the source of the external inputs to a river, microbial processing within the river can alter the balance of externally and internally sourced OC within the river. Thawing of permafrost along Arctic rivers mobilizes ancient DOC into headwater streams, for example, but the DOC exported from the mouth of large Arctic rivers is predominantly modern because this ancient DOC is rapidly mineralized by microbes (Mann et al. 2015, Spencer et al. 2015).

Outputs

Organic carbon can leave a river corridor via gaseous emissions, losses to groundwater, or downstream transport as DOC or POC (Fig. 2). Most rivers are heterotrophic, indicating that they respire more OC than they produce. This flux can be globally significant; in the conterminous United States, about 30% of the total CO_2 emissions from rivers result from respiration of OC (Hotchkiss et al. 2015). Recent estimates of global annual fluxes of carbon suggest that gaseous emissions (1.8 Pg C/yr) from active river channels are approximately double those of downstream transport to the ocean (0.9 Pg C/yr; Battin et al. 2009, Aufdenkampe et al. 2011, Raymond et al. 2013). Estimated gaseous emissions are from active channels and lakes (Battin et al. 2008, Tranvik et al. 2009, Butman and Raymond 2011, Hotchkiss et al. 2015), however, and do not account for emissions from riverine wetlands (Abril et al. 2014) or from terrestrial components in riparian zones and floodplains. Although a substantial portion of OC produced within river corridors appears to be returned to the atmosphere via decomposition within inland waters (Cole et al. 1994), recent quantification of carbon sinks within Amazonian floodplain lakes suggests that accumulation rates may exceed rates of evasion and degassing from channels and floodplain wetlands (Sanders et al. 2017).

Remobilization of stored sediment within the river corridor can also result in loss of mineral-associated OC from mineralization during transport (Bouillon et al. 2009). Mineralization of OC during river transport elevates concentrations of dissolved CO_2, facilitating outgassing to the atmosphere (Handique 2015). Increased

supply of OC from dissolved or particulate constituents may increase the rate of mineralization through priming of previously carbon-limited sediments (Fontaine et al. 2007).

Nutrients, particularly N and P, also influence OC dynamics within rivers. Excess nutrients stimulate the production of algal biomass and associated OC sequestration, thus allowing lakes to be sinks and not sources for CO_2 (Pacheco et al. 2013). Conversely, excess nutrients can also stimulate OC loss through increased microbial processing of POC that releases CO_2 to the atmosphere and to downstream transport of OC (Rosemond et al. 2015).

Estimates of global rates of CO_2 evasion from inland waters are a function of three variables: the relative concentrations of CO_2 in the water and the atmosphere; the global surface area of inland waters; and the gas transfer velocity, which varies as a function of turbulence in the surface water and water temperature (Aufdenkampe et al. 2011). The source of inland water CO_2 is still not known with certainty (Raymond et al. 2013), but appears to be mostly CO_2 from groundwater and a smaller fraction of CO_2 derived from instream metabolism (Hotchkiss et al. 2015). The concentration of CO_2 in water does not correlate strongly with climate or landscape variables, but Raymond et al. (2013) note that estimates include substantial uncertainty because of lack of understanding of factors such as how gas transfer velocity changes with increasing discharge, in steep catchments, and in higher latitudes.

Organic carbon outputs to oceans are partitioned between DOC and POC. Global and regional syntheses differ on the relative magnitude of DOC and POC fluxes, for which limited measurements are available. Meybeck (2003) estimated a global river DOC flux of 0.2 Pg/yr, roughly double the estimated POC flux of 0.1 Pg/yr, whereas Gordeev and Kravchishina (2009) estimate that annual fluxes of DOC (0.02 Pg) are five times higher than POC (0.004 Pg) in Eurasian Arctic rivers. In contrast, Zhang et al. (2009) estimate similar global proportions at 55% DOC and 45% POC. Quantitative estimates of POC export focus almost entirely on finer POC and neglect large wood, which likely constitutes an important contemporary component of OC outputs from some drainages such as the Mackenzie River (Kindle 1921; Kramer et al., *in press*), and was probably a greater output from most forested river drainages prior to widespread deforestation. Both DOC and POC fluxes reflect the magnitude of upland and riparian inputs vs. gaseous emissions, biotic uptake, and OC storage along river corridors. All of these factors can vary substantially through time, making it difficult to accurately quantify OC outputs.

Storage

Organic carbon is stored within river corridors in six forms: downed, dead wood in the channel and riparian

zone (LW); POC within the channel; biomass of aquatic biota; soil organic carbon (SOC), including litter and duff layers in the riparian zone; above- and belowground biomass of riparian vegetation; and DOC in water temporarily stored in the channel and riparian zone. Remarkably little is known of the sources (upland vs. riverine) or quantities of OC stored in these diverse forms or of how these quantities vary within a river network or among networks (Raymond and Bauer 2001, Downing et al. 2008). Instream obstructions such as logjams (Beckman and Wohl 2014) and beaver dams (Naiman et al. 1986, Johnston 2014) can promote storage of locally significant quantities of POC within the channel. Carbon storage within the river corridor, however, appears to occur predominantly in riparian SOC and LW in smaller rivers (Naiman et al. 1987, Wohl et al. 2012b), and the importance of SOC storage is likely to be true for larger rivers with extensive floodplains as well (e.g., Robertson et al. 1999, Hoffmann et al. 2009, Hanberry et al. 2015). Quantities of OC per unit area stored in riparian soils are disproportionately large relative to upland soils in temperate latitudes (Ricker et al. 2013, 2014). Although the details of where most OC is stored (small vs. large rivers, floodplains vs. deltas, high latitudes vs. temperate or low latitudes) remain poorly constrained (Sutfin et al. 2016), recent work suggests that floodplain lakes and wetlands are likely to store disproportionately large stocks of OC (Sanders et al. 2017).

The fate of terrestrially derived carbon in river corridor sediment depends in part on exchanges between sediment and the atmosphere via chemical weathering of inorganic substances, OC composition, riverine fluxes of sediment and OC, and decomposition by biota. Seemingly labile forms of SOC can be protected from decomposition by speciation and adsorption to mineral facies or protection within soil aggregates (Doetterl et al. 2015, 2016). An analogous process occurs in freshwater ecosystems where organo-mineral complexes can lower mineralization rates of carbon (Hunter et al. 2016). Sediment inputs and burial in floodplains remove carbon from oxygen and from greater microbial activity at the surface. However, mixing and introduction of fresh OC can facilitate the metabolism of more recalcitrant carbon at depth via the priming effect (Fontaine et al. 2007, Doetterl et al. 2015, 2016), in which labile DOC can stimulate decomposition of more refractory OC (Guenet et al. 2010).

Relatively little is also known of the influence of gross primary production (GPP) on large-scale OC cycling and transport. The implicit assumption is that riverine GPP is mineralized in place, and relatively quickly, so that there is no net effect on the OC cycle. That assumption may be mostly correct, but existing syntheses have not considered, for example, that some of the OC storage may be from algal-derived OC in rivers and lakes, and that there can be substantial export of algal-derived OC to oceans. In addition, GPP represents a labile OC source to rivers, and this labile C source may stimulate the mineralization of OC via the priming effect (Sampere et al. 2011; Hotchkiss et al. 2014), although this finding is not universal (Catalán et al. 2015).

Battin et al. (2009) estimated that 0.6 Pg C/yr are buried in sediments associated with inland waters, but this estimate is based primarily on quantification of reservoir storage because channel-riparian linkages and the potential for riparian OC storage were not considered in this calculation. Other estimates of burial rate in freshwater sediments (lakes, reservoirs, rivers) vary between 0.2 and 1.6 Pg C/yr, with the large range in estimated values reflecting the limited field data available to constrain this process at the global scale (Regnier et al. 2013). In particular, knowledge of OC storage in the form of riparian SOC is limited, although regionally focused studies (e.g., Wohl et al. 2012b, Ricker et al. 2013, Hanberry et al. 2015, Ricker and Lockaby 2015, Omengo et al. 2016) suggest that cumulative storage of SOC in river corridors could be a significant component of global carbon dynamics. Working in an 83 km^2 forested watershed in Rocky Mountain National Park, Colorado, for example, Wohl et al. (2012b) estimated that the riparian portion of river corridors stored ~25% of the total OC within the watershed, despite occupying less than 1% of watershed area.

Variation in OC dynamics across space and time

The absolute and relative magnitudes of the diverse forms of inputs, outputs, and storage of OC appear to vary substantially across space and through time. Although relatively little is known of these variations, we summarize that understanding here as a means of informing our conceptualization of variations in OC dynamics within and among river networks.

Variations in OC dynamics can reflect differences in geology. Geology includes bedrock lithology and tectonic regime, as these influence topography and hillslope stability, thickness, and OC content of soils, and rates of soil erosion and delivery to river corridors (Galy et al. 2008a, b). Small, mountainous river networks, for example, typically deliver fossil POC derived from bedrock and POC from soils relatively efficiently via downstream transport to the ocean (Leithold et al. 2006, 2015, Hilton et al. 2011a, Hovius et al. 2011), and the rate of POC transport increases rapidly with discharge (Hilton et al. 2008b, 2012, Hatten et al. 2012, Jeung et al. 2012, Goñi et al. 2013, Lloret et al. 2013, Jung et al. 2014). Steep catchments have fewer storage zones for sediment and associated SOC in both uplands and river corridors (Schumm 1977, Montgomery and Buffington 1997). Organic carbon produced within river corridors is more likely to dominate carbon exports to the ocean in large, lowland rivers with extensive floodplains (Leithold et al. 2006, Galy et al. 2008b). Large, lowland floodplains both promote deposition of POC derived from upland terrestrial sources and enhance net primary productivity within the river corridor relative to steep, narrow rivers in mountainous terrain. In addition to originating predominantly

from different sources between river catchments, POC exports vary over several orders of magnitude between river catchments. High-standing oceanic islands in the southwest Pacific, for example, contribute disproportionately more to river exports relative to their proportion of global land mass (Lyons et al. 2002).

Analogous to POC, the primary controls on variation in DOC fluxes are runoff, slope, land cover, and SOC content (Lauerwald et al. 2012). DOC annual yields, however, vary over less than three orders of magnitude among diverse river catchments, whereas POC exhibits greater variability.

Location within a river network can also strongly influence several aspects of OC dynamics, such as primary productivity or residence time of water, which influences DOC supply and removal (Casas-Ruiz et al. 2017). For example, the percentage of CO_2 emissions from aquatic metabolism increases with river size; terrestrially derived CO_2 is most important in small streams (e.g., Hotchkiss et al. 2015).

Variations within a river network can also reflect reach-scale channel and river corridor geometry as this variation in morphology influences storage and residence time of sediment and organic matter. Portions of mountainous river networks with relatively wide, low-gradient river corridor geometry store much greater volumes of SOC per unit length of river than steep, narrow river corridor segments, for example, and the wide river corridor segments may account for the majority of total OC storage within the network (e.g., Wohl et al. 2012b, Sutfin 2015). Floodplains also have a greater overall community respiration than confined river segments (Bellmore and Baxter 2014). These downstream variations in relation to river corridor geometry are better approximated by conceptual models such as geomorphic process domains (Montgomery 1999) or the riverine ecosystem synthesis (Thorp et al. 2006), which emphasize patch mosaics rather than the continuous longitudinal gradients of the RCC.

There are likely also differences in the magnitudes of OC fluxes among climates. Climate influences precipitation and temperature regimes that govern primary productivity of uplands and river corridors. Precipitation influences river flow and groundwater that influence fluxes of OC to and within river corridors, including water temperature and CO_2 emissions from rivers (Raymond et al. 2013). Climate also influences precipitation-induced landslides and associated delivery of fossil OC, SOC, and upland biomass to river corridors.

Regional studies indicate strong correlations between CO_2 evasion and annual precipitation. Measured concentrations of pCO_2 in rivers typically range from 1000 to >12,000 ppm and are typically higher in tropical waters than in temperate waters (Aufdenkampe et al. 2011). The greatest CO_2 efflux occurs in Southeast Asia, Amazonia, and the eastern edge of East Asia (Rasera et al. 2013, Raymond et al. 2013). Mechanistic understanding of why river networks in these tropical regions might contribute

disproportionately to global fluxes is limited. The high values, however, likely reflect high terrestrial productivity, rapid decomposition rates linked to high moisture content and temperatures, and subsequent respiration in floodplains and wetlands and substantial hydrologic fluxes from uplands to river corridors in tropical regions (Butman and Raymond 2011).

Conversely, river corridors of the high latitudes and high altitudes are likely to dominate OC storage (Sutfin et al. 2016) and to have very long residence times of POC (Hilton et al. 2015, Marwick et al. 2015), not least because the largest of these rivers have extensive floodplain lowlands with enormous quantities of SOC stored in permafrost (Schuur et al. 2015). Although these rivers contribute substantial POC fluxes to the Arctic Ocean (McClelland et al. 2016), tropical rivers dominate global OC fluxes to oceans. Meybeck (1993) estimated that rivers in the humid tropics account for 66% of the total mass of riverine DOC and 49% of POC transport to oceans. The specific estimates have changed with time, but subsequent studies support the dominant role of rivers in the humid tropics. Tropical rivers commonly have high rates of CO_2 efflux (Aufdenkampe et al. 2011, Rasera et al. 2013, Raymond et al. 2013), as well as DOC and POC transport (Meybeck 1993, Jahnke 1996). Although floodplains along tropical rivers can store substantial quantities of OC in soil and wetland or lake sediment (Sanders et al. 2017), the residence time of this OC appears to be shorter than in floodplains of the temperate latitudes (Omengo et al. 2016).

Variations in OC storage and flux in river networks through time can be driven by processes operating at vastly different time scales. Short-term disturbances such as severe storms (Hilton et al. 2008b, West et al. 2011, Wohl and Ogden 2013), floods (Sutfin 2015), or wildfires influence inputs of OC to the river corridor, storage of OC within the river corridor, or transport downstream over periods of days to a few years. Storms and floods, in particular, typically result in greater DOC, POC, and LW inputs to rivers, which can be stored in depositional areas within the river corridor, or exported downstream to a reservoir or the ocean (Rathbun et al., *in press*). More sustained changes such as droughts or land use can influence OC dynamics over periods of 10^1–10^2 yr (e.g., Worrall and Burt 2004). Anthropogenically enhanced fluxes of OC from the Mississippi River basin, for example, which is heavily agricultural and urbanized, now dominate the OC balance within the river basin (Raymond et al. 2008). Longer-term changes in OC dynamics are associated with processes operating over 10^2–10^4 yr, such as continuing tectonic uplift (Galy et al. 2008a, 2015) or changing climate (Trumbore 1993, Smittenberg et al. 2006).

Finally, drainage area can influence outputs of OC, but does not appear to have consistent effects on storage. Measured fluxes of DOC, POC, and TOC (total OC, represented by the sum of DOC and POC for a river) strongly relate to drainage area simply because larger

drainage areas have higher water flux (Fig. 3A–C). More interesting is that the standardized major axis regression (Warton et al. 2006) slopes of the relationships for DOC and total export are ~0.95, showing allometric scaling in which export increases slightly more slowly than drainage area (Fig. 3A, C). This finding is evidence for lower

fluxes of DOC in big rivers, possibly due to removal along the river network, although with a sample size of approximately 100, this finding is preliminary. Sedimentation flux loosely relates to drainage area, with the slope insignificantly different than 1 (Fig. 3D). The smaller number of published values of riparian OC stocks in soil
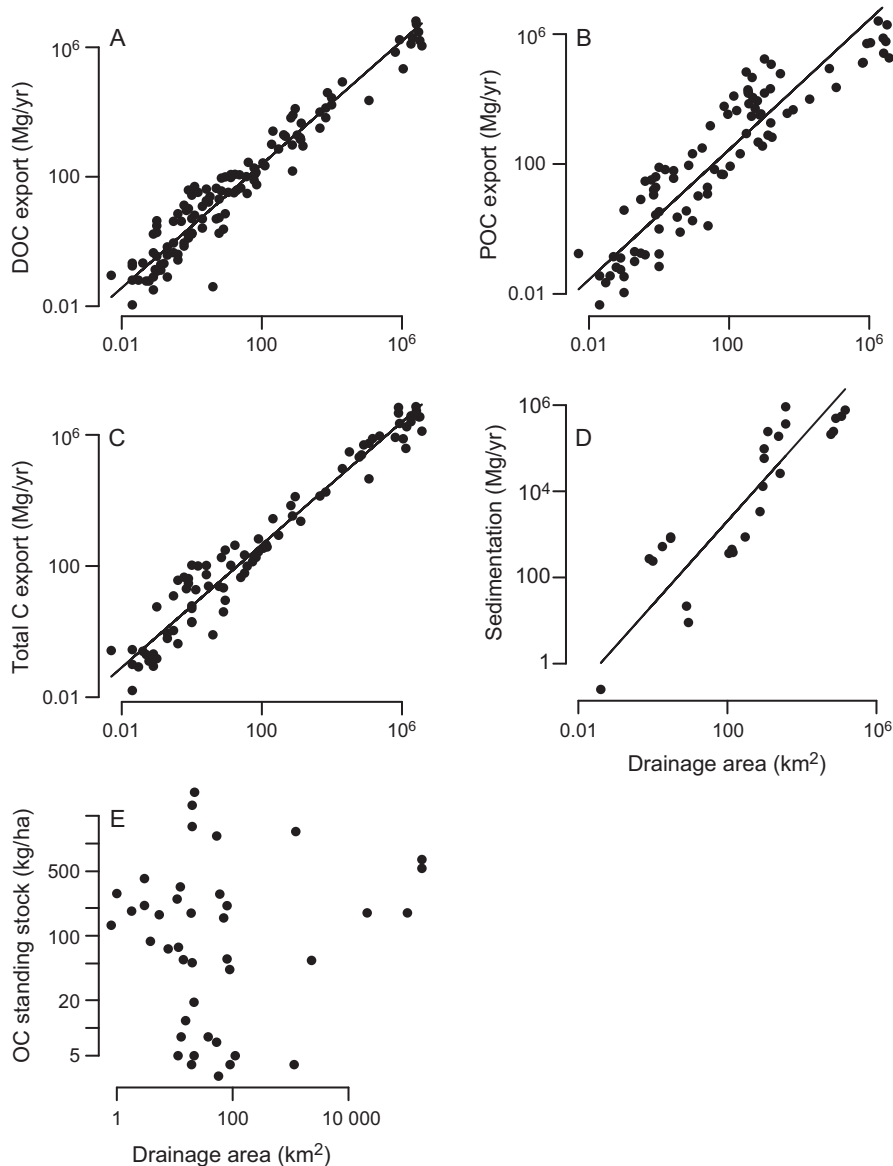

FIG. 3. Scaling of organic carbon fluxes and stock with river drainage area (see Appendix S1: Table S2 for relevant data). Lines are shown for standardized major axis regression (SMA) where the confidence interval did not exceed 1 (Warton et al. 2006). If slope is significantly less than 1, flux or stock decreases more slowly than increasing drainage area. If slope is significantly greater than 1, flux or stock decrease at a faster rate than drainage area. A slope of 0 indicates no relationship between flux or stock and drainage area. (A) The regression between dissolved organic carbon (DOC) flux and drainage area exhibits a slope < 1 (slope = 0.95, 0.91 < CI < 1.00, sample size = 123, $P = 0.053$, $r^2 = 0.93$). (B) The regression between particulate organic carbon (POC) flux and drainage area exhibits a slope not significantly different than 1 (slope = 1.00, 0.93 < CI < 1.10, sample size = 91, $P = 0.83$, $r^2 = 0.83$). (C) The regression between total organic carbon (TOC = DOC + POC) and drainage area exhibits a slope < 1 (slope = 0.94, 0.89 < CI < 0.98, sample size = 85, $P = 0.009$, $r^2 = 0.95$). (D) The regression between sedimentation rate of OC within the riparian zone and drainage area exhibits a slope not significantly different than 1 (slope = 0.98, 0.78 < CI < 1.18, $P = 0.75$, $r^2 = 0.78$). (E) Stock of OC in the form of SOC and large wood within the riparian zone for diverse rivers has no significant relationship to drainage area and is highly variable (OLS regression, $P = 0.54$ ($r^2 = 0.01$)).

and downed dead wood yield highly variable values and do not relate to drainage area (Fig. 3E). The lack of relationships between riparian storage and drainage area is not surprising, given the site-specific controls on river corridor geometry and sedimentation rate. These controls can vary substantially and non-systematically downstream within a river network and among river networks.

Variations in OC Dynamics Among River Segments

We posit strong differences in the relative importance of different forms of inputs, outputs, and storage for natural (or relatively undisturbed) rivers in the temperate, tropical, and high latitudes (Table 1). Because Table 1 is organized largely around expected, rather than demonstrated, patterns of OC dynamics in river corridors, we evaluated only very broad categories of river size, river corridor geometry, and climate. Headwater rivers in Table 1 are first- to third-order rivers (Strahler 1952), whereas large rivers are fourth order or higher. We categorize river corridor geometry with respect to lateral confinement. Confined river corridors have a floodplain and riparian corridor less than twice as wide as the active channel (Livers and Wohl 2015). Unconfined river corridors are

TABLE 1. Most important components of OC storage, inputs, and outputs (A) within a river corridor type or at the reach scale and (B) among river corridor types or at the network scale.

River corridor geometry	$d\mathrm{C_S}/dt$ temperate	$d\mathrm{C_S}/dt$ tropical	$d\mathrm{C_S}/dt$ high latitude	$\mathrm{C_I}$	$\mathrm{C_{Ogas}}$	$\mathrm{C_{Oriver}}$
(A)						
Active channel						
Headwater$_{con}$†	POC	POC	POC	OC to channel CO$_2$ to channel		POC/DOC‡ downstream
Headwater$_{uncon}$	LW POC	POC	LW	OC to channel CO$_2$ to channel		POC/DOC to floodplain POC/DOC downstream
Large river$_{con}$	POC	POC	POC	OC to channel CO$_2$ to channel		POC/DOC downstream
Large river$_{uncon}$	LW POC	POC	LW	OC to channel		POC/DOC to floodplain POC/DOC downstream
Riparian						
Headwater$_{con}$	LW	LW	LW	Upland	Soil respiration Plant respiration	OC to channel CO$_2$ to channel
Headwater$_{uncon}$	SOC	SOC AGB§	SOC	NPP Fluvial	Soil respiration Plant respiration	OC to channel CO$_2$ to channel
Large river$_{con}$	SOC LW	SOC LW	SOC	Upland	Soil respiration Plant respiration	OC to channel CO$_2$ to channel
Large river$_{uncon}$	SOC	SOC AGB	SOC	NPP Fluvial	Soil respiration Plant respiration	OC to channel CO$_2$ to channel
(B)						
Active channel						
Headwater$_{con}$				OC to channel	CO$_2$ from channel	
Headwater$_{unco}$	LW POC	POC		OC to channel	CO$_2$ from channel	POC/DOC to floodplain
Large river$_{con}$						
Large river$_{uncon}$	POC Aq bio	POC Aq bio	LW	OC to channel		POC/DOC to floodplain
Riparian						
Headwater$_{con}$						
Headwater$_{uncon}$	SOC	SOC AGB	SOC	NPP Fluvial		OC to channel
Large river$_{con}$						
Large river$_{uncon}$	SOC	SOC AGB	SOC	NPP Fluvial	Soil respiration Plant respiration	OC to channel

Notes: For the active channel, variables considered are storage (LW, POC, aquatic biota), inputs (OC to channel, CO$_2$ to channel, NPP in channel), outputs (CO$_2$ from channel, POC to floodplain, POC downstream). For the riparian zone, variables considered are storage (SOC, LW, AGB), inputs (NPP, upland, fluvial), and outputs (soil respiration, plant respiration, OC to channel, CO$_2$ to channel).

† Subscript con refers to confined, subscript uncon refers to unconfined.
‡ The relative magnitude of POC to DOC is uncertain based on existing studies
§ AGB is aboveground biomass

more than twice as wide as the active channel. Channel gradient typically correlates with lateral confinement, such that confined river corridors are steeper than unconfined river corridors (Livers and Wohl 2015). These differences in river form and associated processes can profoundly affect primary productivity, CO_2 emissions, lateral fluxes from uplands and riparian zones, and storage of OC in river corridors (e.g., Thorp et al. 2006, Wohl et al. 2012b, Bellmore and Baxter 2014, Sutfin 2015).

We categorize climate as temperate, tropical, and high latitude (60° N or S and higher). Although we recognize the substantial diversity within each of these broad categories, we assume that tropical climates have warm temperatures, large average annual rainfall, high rates of biological productivity and organic matter decay, high rates of river transport for OC and river emissions of CO_2, and minimal OC storage in soils outside of wetlands (Donato et al. 2011, Wohl et al. 2012a, Raymond et al. 2013, Dommain et al. 2014). We assume that climate in temperate latitudes is typically associated with cooler temperatures, moderate average annual precipitation, lower rates of biological productivity and organic matter decay, moderate rates of river OC transport and emissions, and greater OC storage in soils (Meybeck 1993, Raymond et al. 2013, Hanberry et al. 2015). (Our assumptions about tropical and temperate latitudes ignore the existence of drylands and the seasonal tropics, which we acknowledge can function quite differently than rivers in wetter regions, but we exclude drylands and the seasonal tropics here for the sake of brevity and simplicity.) We assume that high-latitude climates correspond to cold temperatures, low average annual precipitation, low rates of biological productivity and very slow organic matter decay, moderate rates of river transport and emissions, and substantial OC storage in soils (Raymond et al. 2013, Hugelius et al. 2014).

The relative importance assigned to different forms of OC inputs, outputs, and storage in Table 1 represents our interpretation of the existing literature and our personal observations of river corridors in diverse environments. These rankings reflect relative magnitude of the OC pool or flux, rather than magnitude per area. Table 1A lists the most important OC pool or flux within a river corridor type and represents a reach-scale emphasis. Within confined headwater river segments, for example, we assume that POC dominates OC storage in the active channel compared to LW or aquatic biomass. Table 1B lists the most important OC pools or fluxes among river corridor types and represents a network-scale emphasis showing which types of river segments or reaches contribute most to different components of the OC budget for the entire river network. In temperate latitudes, for example, we assume that OC pools will be greatest in LW and POC in unconfined headwaters and POM and aquatic biomass in unconfined large river segments. We cannot identify a single dominant OC pool or flux within or among river corridor types for many of the cells in Table 1 because of either variation among

individual river networks that limits generalization or because of lack of place-based data from which to generalize. Thus, these blank cells emphasize some of the more glaring data gaps that limit our ability to predict riverine OC dynamics at the global scale.

Among the trends apparent in Table 1 is that laterally confined river corridor segments, whether in the upper or lower part of a drainage network, have minimal potential for OC storage. Confined river corridor segments commonly have a steep channel with high transport capacity, as well as minimal development of an alluvial floodplain and associated storage of SOC and riparian biomass (Wohl et al. 2012b, Bellmore and Baxter 2014, Sutfin 2015, Sutfin et al. 2016).

Storage across regions, in terms of relative storage within different river segments within a network, among river networks within a region, and among river networks in different regions, is very poorly constrained by existing research. However, we expect differences in the relative importance of various forms of OC storage among regions, as reflected in the three columns for OC storage in Table 1. We expect LW to be more important in temperate and high latitudes than in the tropics, for example, because of the extremely high decay rates for downed wood in the tropics (Harmon et al. 1986; Lininger et al., in press). Conversely, we expect aboveground biomass to be a particularly important OC pool in tropical riparian zones as a result of high terrestrial net primary productivity (Ruesch and Gibbs 2008).

Carbon inputs to the active channel come primarily from uplands in headwater river segments and from upstream river segments and the riparian zone in large rivers. Relative contributions between upstream and riparian sources depend on the position of the river segment in the network and on the length of the river. In confined large river segments, for example, most OC enters the active channel from upstream, whereas at the cumulative, network scale, lateral or riparian OC sources are likely the most important input in large river segments (e.g., Meyer et al. 1997).

Carbon outputs from the active channel occur as CO_2 emissions and as DOC and POC fluxes downstream and to the riparian zone. We expect CO_2 emissions to be particularly important in headwater portions of a river network relative to emissions from downstream portions (Butman and Raymond 2011, Crawford et al. 2015). Downstream DOC and POC fluxes dominate river outputs in confined river segments, whereas we expect lateral fluxes to the riparian zone to be more important where floodplains are present. Carbon outputs from the riparian zone include soil and plant respiration and lateral fluxes of CO_2 and OC to the active channel. We expect soil and plant respiration to be equally important in most riparian zones, with the greatest output from the floodplain of large rivers. We expect lateral fluxes of OC to be particularly important in river segments with floodplains.

In populating Table 1, we concluded that there is no evidence to suggest differences in the relative importance

of various forms of OC inputs and outputs among and within river corridor types among tropical, temperate, and high-latitude rivers. Consequently, the three right-hand columns of Table 1 are the same for all regions. The values of fluxes likely vary in magnitudes among regions, but we have no basis for assuming that the relative magnitudes among river corridor types differ between regions. Table 1 does not address where in a river network each term is most important (e.g., OC to channel in unconfined headwaters vs. unconfined large rivers), because information from river-segment studies is not yet sufficient to support scaling to entire river networks.

We conceptualize our tabular model graphically in Fig. 4A and B, which correspond to Table 1A and B, respectively. Fig. 4 illustrates, in terms of the OC budget, those components that we consider to be most important at the reach scale within different types of river corridor segments (Fig. 4A) and those components and river corridor segments that we consider to be most important at the network scale (Fig. 4B). In particular, Fig. 4B emphasizes the relative importance of low gradient, laterally unconfined river segments within river networks.

Source–Sink Dynamics of OC in River Corridors

Within a watershed's carbon budget, river corridors can be either net sources of OC to the atmosphere or ocean, or sinks of OC that enters sediments in the active channel or riparian zone. For the entire river corridor to be a sink, two criteria need to be met: net ecosystem productivity (NEP) plus import must be positive and exceed export such that dC_S/dt is positive. This scenario is unlikely for the active channel component of a river corridor, but is more likely in the riparian component given positive NEP in the terrestrial portion of the riparian zone and storage of upland or river-transported sediment that is likely to be at or close to saturation with respect to soil moisture.

The balance between source and sink terms varies through space and time in a manner that reflects the terms in Eq. 1. Fluxes of OC become particularly important in this context as does the manner in which a river is defined (i.e., active channel vs. river corridor). Fluxes are controlled by numerous factors, including decomposition, rates of sediment deposition vs. erosion, and conversion of POC and DOC to CO_2 (Fig. 5).
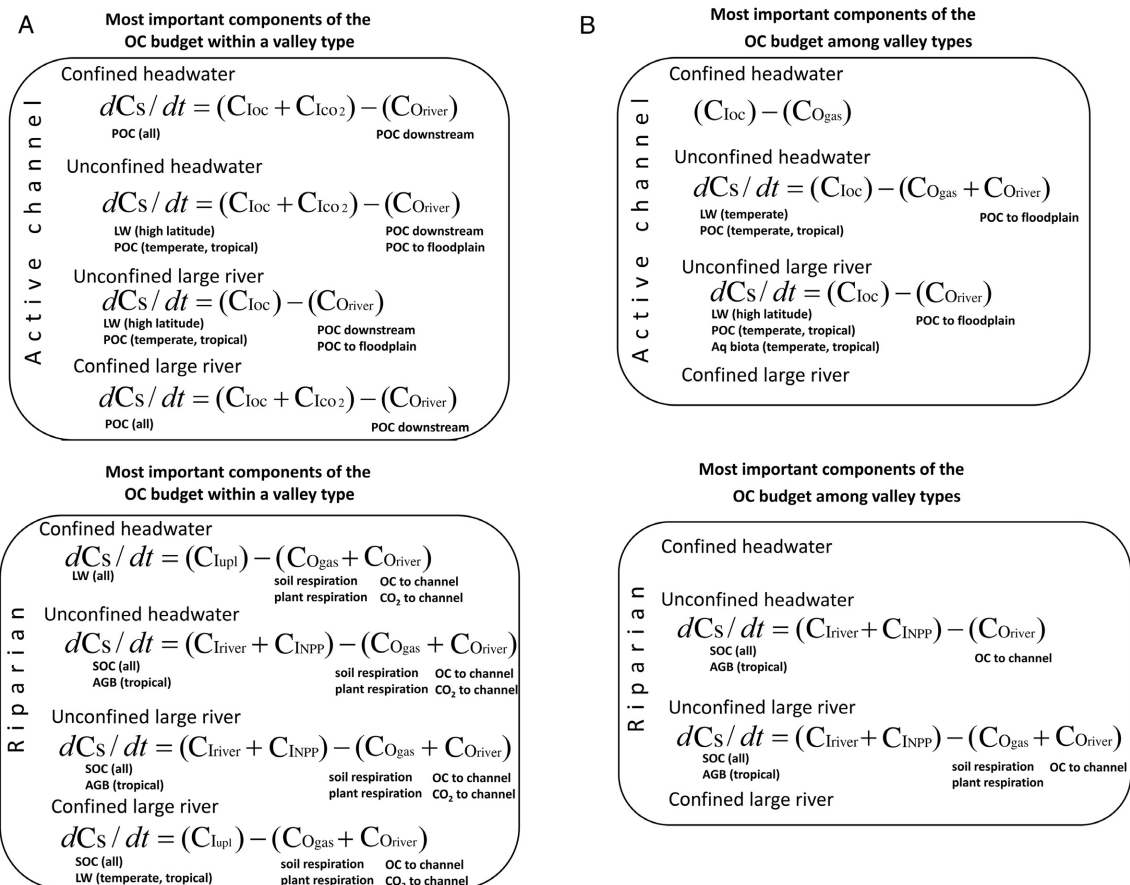

Fig. 4. Summary of Table 1 in terms of the OC budget equation from Fig. 2. (A) The OC budget equation is modified to include only the most important components of OC storage, inputs, and outputs within a valley type or at the reach scale. (B) The OC budget equation is modified to include only the most important components of OC storage, inputs, and outputs among valley types or at the network scale.
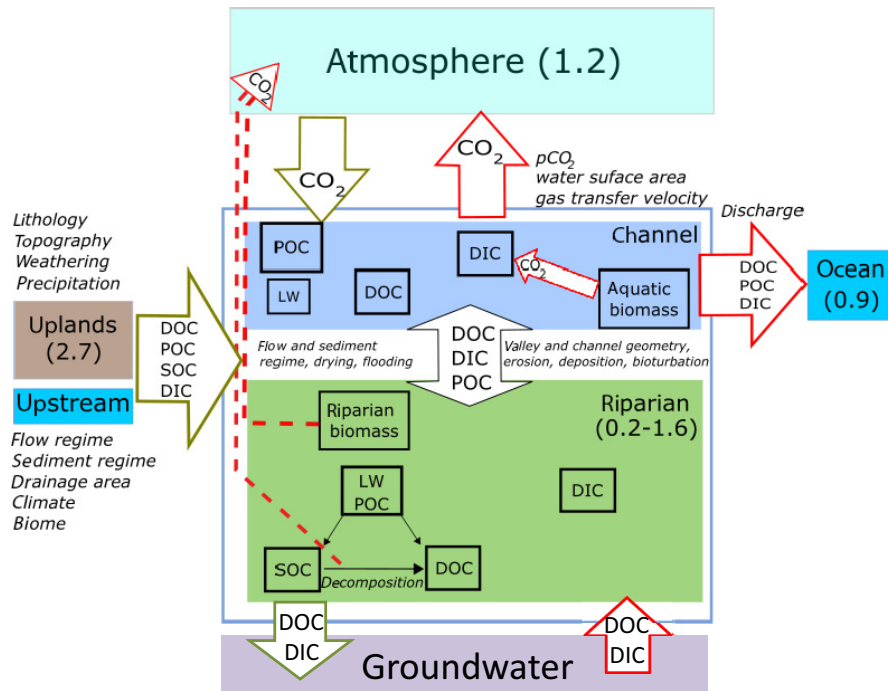
Concepts & Synthesis

Fig. 5. Schematic illustration of carbon dynamics in river corridors. Boxes are C reservoirs; arrows are C exchanges or fluxes (red indicates C outputs, green indicates C inputs); text in italic typeface lists factors that mediate fluxes. We have quantified fluxes where possible, using estimates from existing syntheses, but the large number of unquantified fluxes in this figure reflects data gaps. The numbers in parentheses are fluxes in Pg C/yr from uplands, into the atmosphere, into the ocean, and into the geosphere, as estimated by Aufdenkampe et al. (2011: Fig. 3).

Consequently, the relative magnitude of the pools and fluxes illustrated in Fig. 5 can vary within a river network (e.g., from headwaters to large floodplain rivers, or from confined to unconfined river corridor segments; Table 1; e.g., Thorp et al. 2006, Wohl et al. 2012*a, b*, Bellmore and Baxter 2014) and among river networks.

At a fundamental level, river segments in which most of the OC received from terrestrial sources is released as CO_2 to the atmosphere or efficiently transported to the ocean as DOC and POC are primarily OC sources. River segments that retain substantial amounts of OC in some pool (e.g., riparian sediment or biomass) along the river corridor are primarily OC sinks.

CO_2 emissions and OC transport are high within the active channel (Butman et al. 2016). CO_2 emissions and DOC and POC export are particularly high in the tropics, likely in association with high rates of terrestrial productivity and rapid erosion (e.g., Hilton et al. 2008*b*, 2012). The inferred mechanisms underlying observed high emissions and river fluxes of OC emphasize the strong influence of climate on upland biomass, SOC, and transfer of OC to river corridors. In this sense, the neutral pipe model of rivers has some basis in that factors external to the river corridor exert a primary influence on both terrestrial OC inputs and riverine OC outputs. OC storage, however, occurs primarily within riparian areas of unconfined river segments, and as such depends on fluxes of water and sediment between the

channel and riparian area, as well as NPP and soil development within the riparian area. Riparian areas retain OC and function as biogeochemical reactors that facilitate the speciation, transformation, and opportunities for both long-term storage of carbon and mineralization to the atmosphere. Consequently, it is with respect to riparian storage and biogeochemical processing of OC in aboveground biomass, LW, and SOC that river corridors are least likely to be adequately conceptualized as neutral pipes.

Examples from small, headwater streams in bedrock canyons and large alluvial rivers illustrate the importance of channel-riparian fluxes on OC storage. Relatively small (third order) rivers of the Colorado Front Range in unconfined river corridor segments with old-growth conifer forest, numerous logjams, and multiple secondary channels store ~250 Mg C/ha in SOC as measured down to the bedrock contact. Otherwise analogous unconfined river corridor segments with wet meadows and a single channel store ~500 Mg C/ha in SOC (Sutfin 2015). The higher values of SOC storage in single-channel river corridor segments may reflect continuously saturated soils (which retard mineralization) as a result of groundwater inputs, but also longer residence time of riparian sediment as a result of different magnitude and frequency of fluxes between the channel and riparian area. An example from a very large floodplain river illustrates how changes in the characteristics of the

riparian zone influence OC storage. Working in the alluvial river corridor of the lower Mississippi River valley in the United States, Hanberry et al. (2015) estimated that under contemporary agriculture the river corridor currently stores only about 2% of the inferred historical OC stock of 234 Tg associated with natural bottomland riparian forests.

A key point is that channels are connected to riparian zones and the exchange of OC from the channel to the riparian zone represents potential for storage of transported OC not included in either the neutral or active pipe model of OC dynamics in freshwater. Although changes in channel geometry and flow regime can alter carbon dynamics and especially short-term storage within a channel, as reviewed in the next section in the context of human activities, we conceptualize channels as primarily sources of OC to the atmosphere and ocean. In contrast, we conceptualize riparian areas as primarily sinks of OC over varying timescales and varying proportions of terrestrial and fluvial OC inputs.

Integrating channels and riparian areas, Yue et al. (2012) developed a framework for using sediment delivery ratio (ratio of sediment yield to the total eroded mass) and soil humin content in SOC to delineate river basins as CO_2 sources or sinks. Soil exchanges CO_2 with the atmosphere via chemical weathering of inorganic substances, OC formation, and decomposition by biota, all of which occur during soil erosion, re-deposition, and transport. Yue et al. (2012) propose that a river basin acts as an erosion-induced CO_2 sink with respect to the atmosphere when soil humin content is large relative to sediment delivery ratio, and as a source when sediment delivery ratio is large relative to soil humin content. Based on world average levels of these values, Yue et al. (2012) suggest that rivers cumulatively act as a global carbon sink. Again, this finding highlights shortcomings of the neutral and active pipe models of rivers, and illustrates the utility of both conceptualizing OC dynamics within river corridors in which active channels and riparian zones are coupled entities and conceptualizing river basins in source–sink terms.

HUMAN ALTERATIONS OF CARBON DYNAMICS IN RIVER CORRIDORS

Human activities affect every aspect of the carbon cycle. Although human alterations of the nitrogen cycle in freshwater environments have received more attention in the scientific literature (e.g., Rockström et al. 2009, Steffen et al. 2011) and it is widely acknowledged that human activities dominate global nitrogen fluxes (Howarth et al. 2002, Fenn et al. 2003), alterations of the carbon cycle in freshwater environments may be of similar magnitude.

Literature examining planetary sustainability with respect to the global carbon cycle focuses on fluxes of CO_2 into the atmosphere, equilibrium between the atmosphere and ocean and resulting effects such as ocean acidification, and terrestrial carbon sequestration in biomass and soils (e.g., Berhe et al. 2007, Battin et al. 2009, Aufdenkampe et al. 2011, Raymond et al. 2013). Studies examining altered OC fluxes from freshwaters at the regional to global scale either focus on carbon dioxide emissions (e.g., Butman and Raymond 2011, Raymond et al. 2013, Butman et al. 2016) or, when examining downstream fluxes to the ocean, ignore OC storage along river corridors except in lakes and reservoirs (e.g., Regnier et al. 2013). We contend that consideration of global carbon dynamics in the context of human alterations and sustainability should also include channel-riparian fluxes and storage of OC in riparian areas. Here, we discuss the multiple facets of human alterations that affect regional to global scale carbon dynamics within river corridors and consequently OC fluxes to the atmosphere, the oceans, and the geosphere.

Table 2 lists the primary human alterations of organic carbon dynamics, which we subdivide into upland alterations occurring outside of the river corridor and river alterations that involve modifications of flow regime, channel geometry, riparian areas, and riparian–channel connectivity within the river.

Indirect effects: upland alterations of carbon dynamics

Upland alterations affect carbon inputs to river corridors (the C_I term in Eq. 1). Pervasive examples include: changing sediment yield to river corridors by changing topography and land use; altered fire regimes; and climate change (Fig. 6A).

Changing topography and land use.—Alteration of topography can either increase or decrease OC inputs to river corridors. Topography is altered to facilitate construction of transportation corridors and to make the land surface more suitable for agriculture and urbanization. Humans now move more sediment than all geologic processes combined (Hooke 2000), and alterations of topography and land cover have caused an estimated additional 2.3 billion MT sediment input to rivers worldwide (Syvitski et al. 2005). An extreme example is wholesale changes to upland topography via mining (Ross et al. 2016). Reconfiguration of the land surface can increase POC fluxes to river corridors via bedrock and soil erosion if the reconfiguration results in decreased hillslope stability (e.g., roads increasing landslides; Larsen and Parks 1997). Increased terrestrial POC inputs to rivers can increase downstream transport of POC, increase burial within floodplain and delta sediments, or increase aquatic respiration and CO_2 emissions. Reconfiguration of the land surface can also decrease POC fluxes to river corridors if the alteration of topography results in highly stabilized, urban environments with minimal bedrock and soil erosion and reduced terrestrial vegetation biomass.

Increased sediment yield to river corridors increases SOC inputs, whereas decreased sediment yield decreases

TABLE 2. Human alterations of organic carbon dynamics in river corridors.

| Human activity	Likely effect on OC in river corridors	Sample references
Upland alterations		
Changing topography	(+) OC inputs unless land surface is urbanized	Larsen and Parks (1997), Madej et al. (2013)
Clearing native land cover	(±) OC inputs; (−) inputs from reduced terrestrial C pools, but (+) inputs from exacerbated upland erosion	Madej (2010)
Crops	(±) OC inputs; (+) from enhanced soil erosion, (−) from lower NPP and depleted SOC; (+) N, P inputs and (+) CO_2 emissions	Davidson and Ackerman (1993), Robertson et al. (1999), Quinton et al. (2010)
Grazing	(+) OC inputs; Δ SOC is context-dependent (grazing tends to increase in SOC in C_4-dominated grasslands, but decrease SOC in C_3-dominated grasslands); intense grazing increases erosion	Trimble and Mendel (1995), Reeder et al. (2004), McSherry and Ritchie (2013)
Timber harvest and associated roads	(−) OC wood inputs, (+) OC sediment inputs	Madej (2010), Madej et al. (2013)
Urbanization	(+) DOC and DIC inputs from soil erosion and wastewater	Daniel et al. (2002), Sickman et al. (2007)
Groundwater withdrawal	(−) DOC inputs	Barlow and Leake (2012)
Fire	(±) OC inputs; (+) from increased upland erosion of SOC and charcoal; (−) from decreased LW and POM inputs, and from emission of CO_2	Pierce et al. (2004), Hicke et al. (2012)
Climate change	(±) OC inputs; (+) from greater upland productivity or greater soil erosion, (−) from greater oxidation of soil organic matter in uplands; (±) OC storage in river corridors; (+) from increased flooding and saturated riparian soils, (−) from decreased flooding and riparian productivity	Davidson and Janssens (2006)
River corridor alterations		
Burial of river segments	(−) OC inputs and storage	Elmore and Kaushal (2008)
Floodplain drainage	(−) OC storage on floodplain, increases oxidation and CO_2 emissions	Armentano (1980), Duffkova et al. (2005), Hanberry et al. (2015), Knox et al. (2015)
Construction of levees	(±) OC storage on floodplain; (−) fluvial inputs, can (+) NPP and SOC by reducing fluvial erosion and deposition	Bullinger-Weber et al. (2014), Hatten et al. (2014), Rieger et al. (2014); Sutfin (2015)
Channelization (dredging, straightening, bank stabilization)	(−) OC storage in channel and floodplain because of decreased channel-floodplain connectivity and river inputs of organic matter	Samaritani et al. (2011)
Log floating	(−) OC storage in channel and floodplain	Schama (1996), Comiti (2012), Wohl (2014), Nilsson et al. (2005a)
Removal of instream wood	(−) OC storage in channel and floodplain	Beckman and Wohl (2014), Wohl (2014)
Removal of native riparian vegetation	(±) OC inputs from floodplain and storage on floodplain in SOC and aboveground biomass, depending on what replaces native vegetation	DeLong and Brusven (1994), Giese (2001), Roberts and Bilby (2009), Hanberry et al. (2015)
Removal of beaver	(−) OC inputs and storage	Wohl (2013)
Mining within the river corridor	(±) OC storage; (−) at mining site as riparian vegetation and sediment are removed, (+) at downstream sites as sedimentation increases	Hilmes and Wohl (1995), James (1999)
Flow regulation (dams, diversions)	(−) OC storage on floodplain; (+) OC storage in reservoir sediment; (+) CO_2 + CH_4 emissions from some reservoirs, (−) CO_2 emissions from pools at low-head dams	Tranvik et al. (2009), Raymond et al. (2013), Regnier et al. (2013), Crawford et al. (2016)
Changes in aquatic biota	uncertain and likely highly variable between sites	Schmitz et al. (2014)
Changes in water chemistry (excess N, P)	(+) CO_2 emissions	Rosemond et al. (2015)
Delta engineering and indirect effects of flow regulation	(−) OC storage in channel and floodplain	Syvitski and Kettner (2011), Canuel et al. (2012)

CONCEPTS & SYNTHESIS
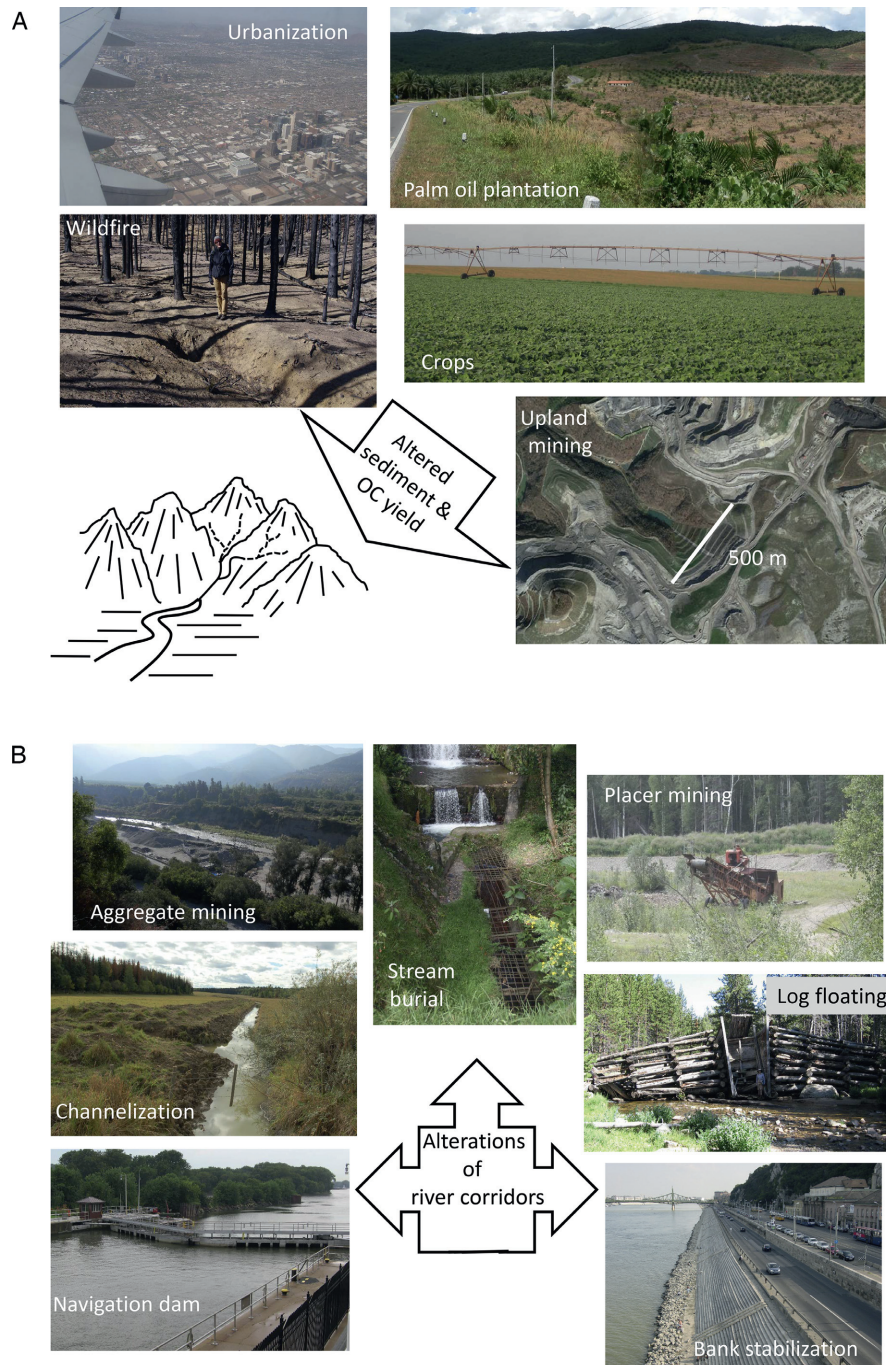
FIG. 6. (A) Sample illustrations of a few of the human alterations of watersheds that influence water, sediment, and organic carbon yields to river corridors. Illustrated here (clockwise, starting at upper left) is urbanization in Arizona, USA; a palm oil plantation in Borneo; mountaintop mining in West Virginia, USA; and burned forest in Colorado, USA. (B) Sample illustrations of human alterations within river corridors. Illustrated here (clockwise, from upper left) are aggregate mining within a channel in Chile; burial of a headwater stream entering an urban area in Colombia; placer mining in Alaska, USA; historical splash dam used in log floating in Wyoming, USA; bank stabilization along the Danube at Budapest, Hungary; a navigation lock and dam along the Mississippi River in Iowa, USA; and channelization of a small stream in Canada.

these inputs. Increases in sediment yield to river corridors resulting from changes in upland topography and land cover are particularly important because the flux of OC mobilized into river corridors is greater than the ability of most river corridors to stabilize the OC via mechanisms such as organo-mineral complexation or

burial in anoxic environments (Aufdenkampe et al. 2011). Rates of erosion, delivery, mixing of fresh mineral surfaces with organic matter, and storage within river corridors likely control watershed- to global-scale OC fluxes (Aufdenkampe et al. 2011). Whether human-induced soil erosion creates a net source or sink of OC in river corridors remains an open question (e.g., Hoffmann et al. 2013), the answer to which partly depends on the characteristics of the river corridors into which the eroded soil is transported. If the eroded soil is transported to the ocean or deposited in a riparian zone in which the SOC is oxidized and released to the atmosphere, human-induced soil erosion can create a net source of OC within river corridors. If the eroded soil is deposited in a reservoir or in saturated or rapidly aggrading floodplain soils that limit oxidation and respiration, the OC may be stored within the river corridor (Van Oost et al. 2012).

Replacement of native vegetation with forest plantations, crops, or grazing lands can reduce or increase OC fluxes to river corridors, but generally has the overall effect of increasing net OC flux to river corridors. Reduction of OC fluxes to river corridors due to clearing of native vegetation can occur via at least three mechanisms. First, comparisons of diverse forests subject only to natural disturbances vs. managed forests or other human-managed land cover indicate lower soil OC and total ecosystem OC pools in human-altered vegetation communities (e.g., Jaramillo et al. 2003). Similarly, tilled soils are typically depleted in SOC relative to untilled soils (Davidson and Ackerman 1993). Second, removal of OC in harvested products such as timber represents a decreased upland pool available for transport into river corridors. If the harvested products are mature trees, OC is lost directly via removal of the tree and indirectly via loss of large wood that could be recruited to the river corridor, where large wood can effectively trap finer organic matter in transport and facilitate overbank deposition of organic matter (Madej 2010, Beckman and Wohl 2014). Third, replacement of native vegetation with fertilized crops and grazing animals can result in greater nitrogen and phosphorus fluxes into river corridors. Excess nutrients can stimulate production of algae and increase mineralization of terrestrially derived OC (Rosemond et al. 2015). The magnitude of each of these effects can vary through time, particularly if land use extent or intensity within a watershed varies through time. However, we propose that the increased soil erosion and flux of SOC associated with removal of native upland vegetation communities is likely to be more important than reduced terrestrial OC pools, so that removal of native vegetation is likely to increase net OC flux to river corridors.

Replacement of native vegetation with urban areas likely reduces OC fluxes to river corridors by substantially decreasing organic matter inputs and sediment yield. For example, capping the land with impervious surfaces can greatly reduce water infiltration, disrupting the linkage between surface water and groundwater recharge and thus reducing fluxes of SOC into stream networks. Urban areas can also contribute substantial quantities of N and P to river corridors (Green et al. 2004, Divers et al. 2013), with effects similar to those described above for agricultural lands. DIC inputs from urban areas can be much higher than those from a naturally vegetated landscape (Barnes and Raymond 2009). OC inputs from urban areas via wastewater are likely to be relatively low in high-income countries, but could be greater in areas with minimal sewage treatment. Urban areas also represent a concentration of food, water, and other resources from a much larger area than lies within the urban boundary (Wackernagel et al. 2006, Sabo et al. 2010) and runoff and infiltration from this concentration can influence river corridors.

Upland mining, by disrupting surface vegetation and creating point sources in the form of tailings piles, can substantially increase sediment yield (e.g., Marcus et al. 2001), and associated fluxes of fossil OC and SOC, to river corridors. As with other forms of increased sediment yield to river corridors, whether this creates a net OC flux from, or storage within, the river corridor depends on whether the sediment is exported from or stored within the river corridor. Mountain top removal, in particular, creates such enormous increases in sediment yield to rivers that low-order streams are either buried under mine spoil or can experience substantial aggradation (Bernhardt and Palmer 2011, Jaeger 2015). Mining can also substantially alter the chemistry of surface water (Palmer et al. 2010). Mining and the resulting presence of heavy metals can influence complex speciation, pH, redox state, adsorption of organic matter to mineral facies, and the decomposition of OC (Brezonik and Arnold 2011).

Human-induced alterations of the chemistry of surface and subsurface runoff entering river corridors is diverse and difficult to generalize. Alterations include enhanced concentrations of N that indirectly influence OC dynamics within river corridors (Hill 1996, Devito et al. 2000, Hill et al. 2000). A more direct alteration of OC comes from increased C loading in sewage water from humans and domestic animals (Borges et al. 2005, Regnier et al. 2013).

Lowering regional water tables may decrease OC storage in uplands and inputs to river corridors. Groundwater withdrawal for agricultural irrigation, industrial manufacturing, and municipal consumption has lowered water tables in diverse regions of the world. Groundwater can contain substantial quantities of DOC and DIC (Hem 1985, Cai 2003), so lowered water tables that result in reduced groundwater fluxes into channels and floodplains can also reduce OC inputs to river corridors. In addition, reducing the spatial extent and depth of saturated soils in uplands can result in oxidation of organic matter in the soil and lower SOC content (Armentano 1980, Duffkova et al. 2005, Trumbore and Czimczik 2008).

Altered fire regimes.—Altered fire regimes influence OC dynamics in several ways that differ through time, making it difficult to generalize their net effect. Alterations of natural fire regime in uplands, either through fire suppression or enhanced fires during initial clearing of native vegetation, can significantly change terrestrial inputs of OC to river corridors, as well as inputs of large wood. The occurrence of a fire is commonly followed by an increase in water and sediment inputs to the river corridor because of decreased infiltration capacity and slope stability (Moody and Martin 2001, Shakesby and Doerr 2006). Inputs of charcoal to river corridors also typically increase for at least a year or two (Meyer et al. 1992), but inputs of organic matter may decrease for many years following a fire as litter and duff layers gradually reform (Kane et al. 2007). Inputs of large wood can decline for several decades following a fire (Bragg 2000), but this pattern largely depends on fire severity. Research from diverse environments suggests that at least some portion of the increased sediment mobilized from burned uplands can be deposited in floodplains and alluvial fans for periods of 10^2–10^3 yr (e.g., Oguchi 1997, Pierce et al. 2004), which can bury and store charcoal moving with the sediment. Fires also release substantial quantities of upland OC as CO_2 to the atmosphere (Hicke et al. 2012), which would otherwise eventually enter river networks.

Climate change.—The potential effects of climate change on OC dynamics within any river network will depend on the specific scenario of climate change likely to occur in the river network, as well as the existing characteristics of the network. Changes in temperature and precipitation will influence OC dynamics through changes in upland sediment yields, vegetation, land use, and human consumptive demand for water, especially for regions in which climate change will create warmer and/or drier conditions. The most immediate effects of changes in temperature and precipitation are likely to occur within the dynamics of soil OM and metabolism.

The net effect of climate warming on soil carbon remains uncertain due to complex interactions between soil burial, soil warming, and soil moisture content (Davidson and Janssens 2006). Organic matter persists in soil if it is physically isolated from decomposition by microbes via incorporation into soil aggregates or sorption into mineral (or other organic) surfaces (Trumbore and Czimczik 2008). Similarly, buried soil horizons and organic-rich lenses at depth are removed from the biologically active surface layer (Gurwick et al. 2008). Resistance to metabolism as a result of depth within the profile can be largely influenced by changes in temperature and soil moisture regime.

Carbon stored belowground could be transferred to the atmosphere by a warming-induced acceleration of its decomposition. On the other hand, increases of plant-derived carbon inputs to soils could exceed increases in decomposition. Part of the uncertainty arises from the complexity of soil OC, which includes thousands of different compounds, each with its own kinetic properties (Davidson and Janssens 2006). Uncertainty also arises from environmental constraints that affect temperature sensitivity of decomposition. Temperature also affects OC by influencing the rates and mechanisms of POC production (Canuel et al. 2012) and the occurrence of freezing that can reduce POC decomposition (Trumbore and Czimczik 2008) or thawing of frozen ground that can release aged OC from riparian and upland soils to rivers for microbial processing (Spencer et al. 2015).

Soil moisture regime constitutes an environmental constraint on SOC that is subject to changes in climate. Flooding reduces oxygen diffusion to decomposition reaction sites, causing fewer degradative enzymatic pathways (Davidson and Janssens 2006). Anticipated decrease in annual average snowpack and earlier timing of snowmelt (Bates et al. 2008) are likely to result in drier conditions in riparian ecosystems, which could influence metabolism of SOC. Drying of otherwise saturated soils could increase decomposition. Poorly understood feedbacks are likely to occur among the numerous variables and processes influencing C dynamics. Rates of OC burial in northern lakes, for example, have increased significantly during the past century as a result of the combined effects of increased temperature, atmospheric nitrogen deposition, and other factors (Heathcote et al. 2015).

In summary, alterations of portions of a drainage basin outside of the river corridor may either increase or decrease OC inputs to river networks and the magnitude of these effects may be greater in smaller watersheds (Stackpoole et al. 2016). However, model simulations suggest that river transport of carbon has increased by approximately 20% since 1750 (Regnier et al. 2013) and measured data support this estimate (e.g., Meybeck 1982, Milliman and Meade 1983, Richey 2004). This increase is attributed primarily to deforestation and intensive agriculture that have increased sediment yield and OC inputs to rivers (Raymond et al. 2008), but it is worth noting that little attention has been given to how alterations of river corridors influence the ability of rivers to outgas or bury these increased OC inputs. For example, Regnier et al. (2013) assume that CO_2 outgassing and OC burial in river corridors scale linearly with the estimated increase in soil-derived carbon exported to oceans, but note that, although this assumption may be reasonable for the air-water flux, the change in carbon burial is likely to be more complex. Most of the attention to carbon burial thus far has focused on natural lakes and reservoirs (e.g., Mulholland and Elwood 1982, Cole et al. 2007), rather than on riparian zones. Limited studies indicate that SOC storage in riparian corridors of very small channels increased substantially following regional changes in land cover (e.g., Ricker et al. 2012), whereas SOC storage decreased substantially in the corridor of large alluvial rivers that have been affected by direct alterations in the form of floodplain drainage, flow regulation, and riparian deforestation (e.g., Hanberry et al. 2015).

Direct effects: alterations of river corridors

River alterations by humans involve changing channel and floodplain process and form in ways that can influence the terms C_S, C_{Ogas}, and C_{Oriver} in Eq. 1. Basic alterations in channel and river corridor geometry include alteration of channel form via burial of river segments, mining in the river corridor, channelization (dredging, straightening, bank stabilization), log floating, and removal of instream wood; alteration of hydrologic conditions, including floodplain drainage, construction of levees, and flow regulation; alteration of aquatic and riparian biota, including removal of beaver (*Castor canadensis* in North America and *Castor fiber* in Eurasia) and removal of native riparian vegetation; alterations of water chemistry; and alterations of delta form and process, all of which can substantially influence OC dynamics (Fig. 6B). The remainder of this section briefly reviews the effects of each category of alteration.

Altered channel form.—Burial of river segments reduces OC inputs and storage. Burial occurs when channels are placed into pipes or otherwise completely covered and disconnected from floodplains and adjacent uplands. Burial primarily occurs in relatively small channels and has been undertaken in rural and urban environments (Elmore and Kaushal 2008, Roy et al. 2009). Channel burial lowers rates of ecosystem processes such as nitrogen cycling (Beaulieu et al. 2015) and metabolism (Pennino et al. 2014). The first- and second-order channels that constitute most total channel length within most river networks (Downing et al. 2012) are important for processing and uptake of nutrients (Alexander et al. 2007), as well as CO_2 emissions of terrestrially derived OC and CO_2 (Hotchkiss et al. 2015).

An extreme example of channel burial occurs during mountain top mining for coal in the Central Appalachian Mountains, USA. Here, waste rock from surface mines is deposited in river valleys, directly burying thousands of kilometers of headwater streams (Lindberg et al. 2011), along with associated indirect effects of vegetation removal, loss of carbon-rich topsoil, increased erosion, and altered hydrologic flow paths between uplands, river corridors, and river channels (Palmer et al. 2010). All of these effects have myriad consequences (reviewed above) for OC dynamics within river corridors.

Other mining activities within the river corridor also alter OC dynamics by changing sediment flux and riparian vegetation, with a net effect that varies between different portions of an affected river. Mining within channels and riparian areas can be focused on metals disseminated among alluvial sediment (placer mining) or on alluvial sediment to be used for construction aggregate. Either type of mining can remove riparian vegetation and mobilize channel and riparian sediment (Hilmes and Wohl 1995, James 1999), thus decreasing OC storage at the mining site. Increased sediment deposition downstream, however, can result in increased OC storage.

Channelization, undertaken to increase the downstream conveyance of a channel, also reduces OC inputs and storage. Among the secondary effects of channelization are decreased overbank flows, which reduces the OC content of floodplains (Noe and Hupp 2005) by drying floodplain soils and increasing organic matter oxidation; and reduced physical complexity of the active channel. Physical complexity in the form of channel-margin irregularities (bank embayments, pools and riffles, bars) and secondary channels creates areas of flow separation and at least temporary retention of fine sediment and organic matter (Livers and Wohl 2015). Organic matter that is retained in areas of flow separation is more available for uptake by stream biota (Battin et al. 2008) and is more likely to be buried within river sediments and stored for periods of up to thousands of years.

Log floating, which primarily reduces OC storage in the channel and floodplain, is the floating of timber to downstream locations for milling. In commercial operations, hundreds of thousands of logs were floated down a river each year, and even the smallest headwater channels in a network have been used for log floating. Historical records of commercial log floating in Europe date to the Middle Ages (Schama 1996), and commercial log floating continued into the 20th century in much of Europe and North America (Comiti 2012, Wohl 2014). Like channelization, log floating simplifies and homogenizes channel geometry. Direct modifications undertaken to facilitate the downstream movement of logs include blocking off secondary channels and removing obstructions such as natural logjams. Channel geometry is also simplified through the erosion of channel boundaries associated with building and then dynamiting temporary splash dams or the abrasive action of enormous quantities of wood moving downstream (Young et al. 1994, Miller 2010, Ruffing et al. 2015). The net effect of simplifying and homogenizing channel geometry is to reduce OC storage in the channel and floodplain, although some logs that were commercially floated became saturated and were buried in the streambed.

Wood removal has decreased OC storage in channels and riparian zones. Removal of naturally occurring downed wood in channels and riparian areas has been undertaken for centuries to improve navigation, reduce overbank flooding, enhance fish passage, and remove obstacles during log floating. Large wood can be recruited to a river corridor from uplands or from riparian forests. Channels from headwater streams to major rivers such as the Mississippi have been altered through the removal of hundreds of millions of logs (Wohl 2014). Where large wood is present as dispersed individual pieces or as logjams, the wood increases hydraulic roughness (Shields and Smith 1992, Wilcox et al. 2011) and creates zones of flow separation and backwaters that retain sediment and particulate organic matter (Bilby and Likens 1980, Brooks et al. 2006, Beckman and Wohl 2014). Large wood creates diverse habitat for

organisms that can process organic matter, and the wood enhances hyporheic exchange (Hester and Doyle 2008, Sawyer et al. 2011), transient storage (Day 2015), and associated biological processing of carbon. Logjams, in particular, can enhance overbank flooding, channel avulsion, and formation of secondary channels (O'Connor et al. 2003, Sear et al. 2010, Wohl 2011, Collins et al. 2012) and promote channel–riparian connectivity. Wood thus both enhances retention of POC in channel and riparian areas, and itself serves as a source of POC and DOC as it decays (Ward and Aumen 1986). Wood that reaches the ocean also serves a vital function in nearshore and even offshore marine ecosystems by providing nutrients and habitat for diverse invertebrates and vertebrates (Gonor et al. 1988). Removal of large wood from the channel and riparian zone results in loss of channel–floodplain connectivity, channel complexity, and retention of OC in the form of POC and large wood (Beckman and Wohl 2014).

Altered hydrologic conditions.—Floodplain drainage reduces the OC content of floodplain sediments by decreasing soil moisture levels and facilitating oxidation of organic matter in the soil (Armentano 1980, Duffkova et al. 2005, Trumbore and Czimczik 2008). Floodplain drainage is commonly undertaken to facilitate agriculture or urbanization in the floodplain. Construction of levees reduces the OC content of floodplain sediments by laterally disconnecting the floodplain from the active channel (Bullinger-Weber et al. 2014), causing the river corridor to behave more like a confined river corridor segment. Levees limit overbank flows and the associated inputs of river-transported organic matter as well as causing drying and oxidation of floodplain sediments. Reduced overbank flows can also alter riparian vegetation, lowering inputs of organic matter from riparian vegetation. However, where precipitation or groundwater inputs are sufficient to support riparian forests or wet meadows, litterfall and downed wood may result in higher concentrations of riparian SOC from net primary production in leveed floodplains than SOC concentrations in portions of the floodplain subject to fluvial erosion and deposition (Rieger et al. 2014, Sutfin 2015). Finally, floodplain drainage can reduce or eliminate riparian groundwater fluxes into channels. Groundwater contains DOC, so reduced groundwater inputs represent another loss of OC fluxes within river corridors.

Flow regulation alters OC dynamics within river corridors in a variety of ways that remain poorly understood. These include (1) reduction of peak flows and associated reduction of lateral connectivity between channel and floodplain, (2) storage of particulate organic matter in reservoirs (Tranvik et al. 2009), (3) outgassing of CO_2 and, to a lesser extent, CH_4 (Tranvik et al. 2009, Bastviken et al. 2011, Fearnside and Pueyo 2012, Deemer et al. 2016) from reservoirs, which have a higher partial pressure of CO_2 during approximately the initial 15 yr after impoundment, but are then comparable to natural lakes (Raymond et al. 2013), (4) generation and export of OC in tailwater reaches downstream from dams (Ulseth and Hall 2015), and (5) downstream alteration of water chemistry and temperature, thus affecting internal carbon cycling. With regard to reduced lateral connectivity, Hatten et al. (2014) found that sources of organic matter in the riparian zone changed in relation to lateral connectivity. In hydrologically connected sites, sediment OC from eroded upland soils accumulated in riparian areas in a pattern correlated with distance to and discharge of the channel. Hydrologically disconnected sites had lower rates of C deposition and transfers of C within the riparian zone, and groundwater dynamics were more important than proximity to the channel.

Construction of dams and reservoirs has strong direct effects on OC dynamics in river corridors at the global scale. For example, even though OC river fluxes to the ocean show a net increase, much of the SOC eroded from uplands is buried in reservoir sediments (Smith et al. 2001). OC retention within reservoirs correlates strongly with water retention time. The net effect of reservoirs is to reduce transport and oxidation rates of allochthonous river OC and to increase removal of atmospheric CO_2 by primary producers such as algae and macrophytes: both of these processes cause reservoir sediments to serve as an OC sink (Mulholland and Elwood 1982). For example, Butman et al. (2016) estimate that total CO_2 efflux from U.S. lake and reservoir surfaces is 16 Tg C/yr, whereas total OC burial is 20.6 Tg C/yr (relative to an estimated total flux to oceans of 41.5 Tg C/yr). However, they note that the high level of uncertainty associated with burial of OC and emissions of CO_2 from lakes and reservoirs highlights a gap in our understanding of the processes involved.

Altered aquatic and riparian biota.—Alteration of aquatic and riparian animal assemblages can also influence OC dynamics, although these activities are not usually considered in large-scale carbon budgets (Schmitz et al. 2014). The net effect of these alterations on OC budgets depends on the specific scenario. Animals can directly alter C cycling via calcification and bioturbation and indirectly through food web effects and engineering. Calcifiers such as mollusks release a mole of CO_2 for each mole of carbonate fixed into their shells. This physiological process can release large amounts of CO_2 (Chauvaud et al. 2003), although these CO_2 fluxes are likely small relative to daily variation in gross metabolic fluxes (Hotchkiss and Hall 2010). Variation in food web structure via a trophic cascade can alter CO_2 emissions from lakes by controlling rates of CO_2 fixation (Schindler et al. 1997). Animal bioturbation of benthic sediments can also influence the amount of sediment OC released to the water column to be decomposed and released to the atmosphere (Schmitz et al. 2014). Bioturbation by fish can increase organic sediment transport

and the spatial scale of carbon cycling in streams (Taylor et al. 2006). Emerging aquatic insects can return OC to terrestrial environments (Scharnweber et al. 2014) and migrating fish can import marine-derived OC to river ecosystems (Bilby et al. 1995). In riparian environments, herbivores such as moose can indirectly control rates of primary productivity and heterotrophic respiration through browsing. Moose can also influence soil microbial decomposition via their dung and by altering the nutrient content of plant litter (Pastor et al. 1988). Studies in boreal forests indicate that high densities of moose can cause declines in CO_2 uptake by vegetation and storage in plants and soil, as well as influencing humidity, soil temperature and moisture, and fire regime (Schmitz et al. 2003). Within aquatic habitats, moose can stimulate nutrient fluxes through bioturbation of sediments, with potential effects on OC cycling (Bump et al. 2016). The net effect on OC dynamics of altering aquatic and riparian animal assemblages thus depends on the species affected and the intensity of the alteration, but such alterations have received relatively little attention in terms of their potential net effects on OC dynamics within river corridors.

Of all vertebrates, beaver have the largest effects on riverine C cycling. Removal of beaver from river corridors, undertaken as part of commercial harvest of beaver fur and to reduce flooding associated with beaver dams, has the net effect of decreasing OC inputs and storage by diminishing the floodplain wetlands associated with beaver ponds and the wet meadow complexes known as beaver meadows (Naiman et al. 1988). Numerous studies have shown that removal of beaver decreases attenuation of downstream fluxes of water, fine sediment, organic matter, and nutrients, and associated decreases in storage of OC in floodplain sediment and riparian vegetation (e.g., Naiman et al. 1988, Wohl 2013, Johnston 2014).

Removal of native riparian vegetation can influence OC dynamics in diverse ways. Biomass, rates of NPP, litterfall, and SOC typically correlate with type and age of riparian vegetation (DeLong and Brusven 1994, Giese 2001, Roberts and Bilby 2009). Removal of native vegetation can reduce inputs of POC in the form of downed wood and finer organic matter, analogous to the effect described for removal of native upland vegetation, although the magnitude of this effect depends on what type of land cover replaces the native vegetation. Removal of native riparian vegetation can also reduce hydraulic roughness of the floodplain and its ability to trap and store POC transported by overbank flows (e.g., Burkham 1976). Again, the magnitude of this effect depends on what replaces the native vegetation. In some cases, exotic riparian vegetation can create greater hydraulic roughness (e.g., Griffin et al. 2005) and sediment storage (e.g., Graf 1978). Replacement of native riparian vegetation by exotic species can also result in alterations of the composition and function of the soil microbial community (Wolfe and Klironomos 2005), and

presumably in associated soil carbon dynamics, although such alterations do not yet appear to have been explored in scientific literature.

Altered water chemistry.—Alteration of water chemistry in the form of increased levels of nitrogen and phosphorus can alter fixation of DIC and mineralization of OC, with varying effects on OC fluxes. Excess nutrients have the well-known effect of stimulating gross primary production of C-rich algal biomass (Schindler et al. 1997, Conley et al. 2009). Most research focuses on the water quality problems associated with algal blooms; only recently have ecologists studied the effect on OC cycling. Findings are that eutrophication can increase burial of OC in lakes (Heathcote and Downing 2012, Anderson et al. 2014) and cause lakes to be OC sinks rather than sources of CO_2 (Pacheco et al. 2013). Excess nutrients will have the opposite effect on ecosystems that receive mainly terrestrial inputs. Recent work shows enhanced OC loss via increased POC mineralization through microbial processing, with increased CO_2 emissions and lowered standing stocks of OC in headwater streams (Rosemond et al. 2015). At the watershed scale, it is possible that increased nitrogen deposition has increased DOC yield in the Hudson River, USA (Findlay 2005).

Altered delta form and process.—Alteration of delta form and process likely has the net effect of reducing OC storage in delta sediments. Alterations of deltas include construction of levees, channelization, and removal of native vegetation on the delta, as well as the downstream effects of flow regulation (e.g., reduced sediment inputs, altered magnitude and timing of flow, altered nutrient inputs; Canuel et al. 2012). Deltas are naturally dynamic environments subject to regular inputs of sediment and organic matter from distributary channels on the delta, and to reworking of sediment via waves on the lake or ocean in which the delta is formed. Deltas form a potentially enormous but poorly quantified reservoir of OC in the form of sediment and, to a lesser extent, vegetation. Flow regulation has reduced regular inputs of sediment and organic matter from upstream, while processes such as compaction, subsidence, and wave erosion continue, resulting in accelerated erosion of deltas around the world (Syvitski et al. 2009, Syvitski and Kettner 2011).

In summary, although enhanced hillslope instability and soil erosion add OC to river corridors, the net effect of most human activities, with the exception of reservoir construction, appears to be that of reducing the ability of river corridors to store OC within biota and sediment (Fig. 7). This increase in inputs and decrease in storage is reflected in increased OC fluxes to oceans (Regnier et al. 2013). Human-induced soil erosion has increased sediment transport by rivers by an estimated 2.3 ± 0.6 Pg/yr, but reduced the flux of sediment to coastlines by $\sim 1.4 \pm 0.3$ Pg/yr because of sediment storage within reservoirs (Syvitski et al. 2005). These numbers suggest that POC fluxes to the ocean have decreased. Estimates
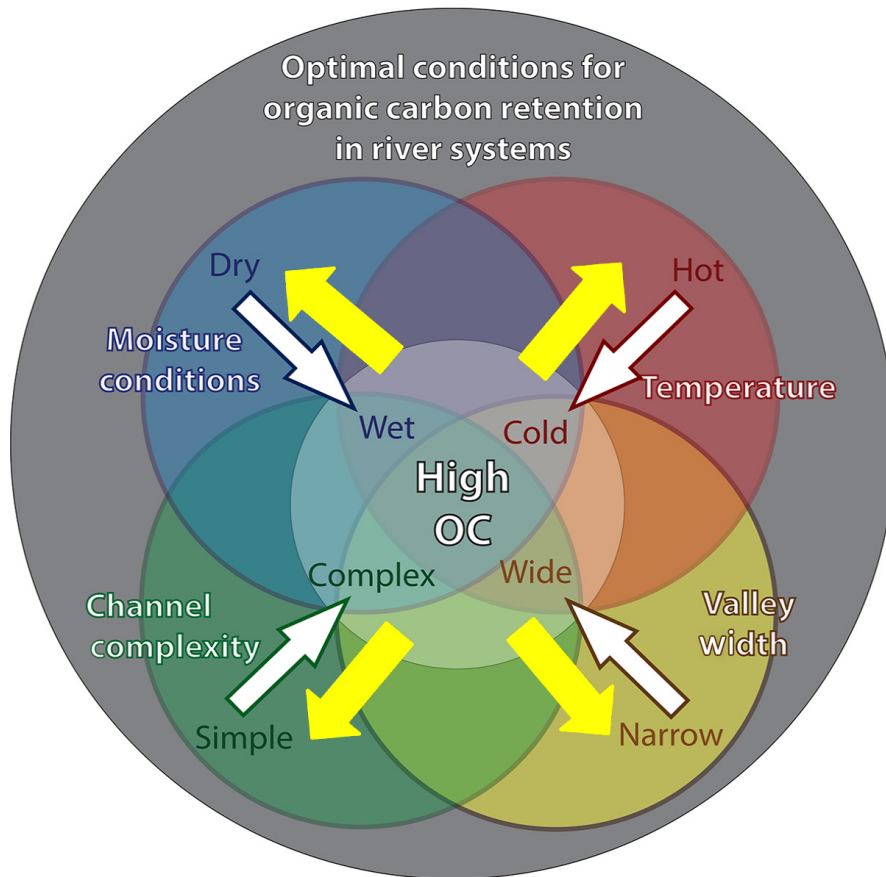
Fig. 7. Schematic illustration of hypothesized optimal conditions for OC storage in river corridors and human alterations of optimal conditions. White arrows indicate the gradient toward optimal conditions. Each component illustrated here can be influenced by numerous variables: soil moisture, for example, can be influenced by climate, flow regime, channel–floodplain connectivity, valley geometry, and so forth. Yellow arrows indicate the net effect of human activities, which tend to decrease the width and physical complexity of river corridors, reduce soil moisture and, indirectly, increase soil temperature via reduced shading and climate warming. Modified from Sutfin et al. (2016: Fig. 5).

of increased OC fluxes to oceans therefore likely represent primarily increased DOC fluxes. The net effect of human activities is therefore to effectively convert river corridors from sinks for OC in riparian soils and biomass to OC sources to the atmosphere and ocean. Engineering of river corridor and channel geometry and flow regulation reduce storage capacity in channels and, most importantly, in floodplains and deltas. Riparian sediment storage is a particularly critical component of OC dynamics. Only 5–25% of eroded sediment reaches the ocean. The great majority of sediment is stored in uplands, along river corridors, and in other inland freshwaters (lakes, reservoirs; Aufdenkampe et al. 2011), so that the cumulative global effect of reduced storage capacity along river corridors outside of reservoirs is likely to substantially influence global C dynamics. Human alterations of river corridors are also changing the spatial arrangement of OC storage within river networks, by increasing storage capacity within reservoirs but reducing the potentially much larger riparian storage outside of reservoirs.

Another way to consider human alterations of C dynamics in river corridors is to recognize that, as with global N dynamics, human activities now likely constitute the dominant influence on inputs, storage, and outputs of OC within river corridors. Although limited studies appear to have specifically evaluated historical changes in OC content of nearshore marine sediments in the context of land use, for example, there is evidence that land uses associated with soil erosion and the presence of reservoirs can dominate OC delivered to and preserved in continental margin sediment (Sampere et al. 2011).

Net effects of human activities on source–sink dynamics of OC in river corridors

The net effects of human activities are to increase OC inputs to rivers and decrease OC storage along the riparian portion of river corridors outside of reservoirs. The result of increased inputs and decreased storage is increased DOC fluxes to the ocean and CO_2 emissions to

the atmosphere (Fig. 8). Regnier et al. (2013) estimate that, of the 1.0 Pg C/yr entering inland waters as a result of human activities, ~0.4 Pg C/yr is emitted back to the atmosphere as CO_2 and ~0.5 Pg C/yr is sequestered in sediments of river corridors (including reservoirs), estuaries, and coastal waters. Regnier et al. (2013) note that the latter number is poorly constrained, and other syntheses have proposed different numbers: Raymond et al. (2013), for example, estimate a global evasion rate of 2.1 Pg C/yr. The details of where these human-induced changes in OC fluxes are greatest remain relatively poorly known, but we can infer likely trends based on available compilations of relevant human activities. For example, changes in land use/land cover associated with agriculture and urbanization have been greatest within the temperate latitudes and the tropics outside of sub-Saharan Africa and parts of the Amazon and Congo basins (Ramankutty and Foley 1999, Pielke et al. 2011). Elsewhere, only Antarctica and boreal/tundra areas in Siberia and North America have avoided extensive changes.

The most recent global syntheses of dams (Nilsson et al. 2005b, Lehner et al. 2011, Grill et al. 2015) indicate that most of the world's major river basins have at least some degree of flow regulation. Only a few basins, predominantly at high latitudes, are minimally impacted by dams. This synthesis does not include flow diversions,

which would increase the extent of river basins in which flow has been directly altered. Within the United States, for example, only about 2% of the 5.6 million kilometers of rivers is unaffected by dams, and these dams impound a volume of water approximately equal to the annual continental runoff (Graf 2001). Globally, impounded waters cover approximately 337,000 km^2 (Downing et al. 2006). In addition to altering downstream flow regime and channel-riparian connectivity, dams create reservoirs that can store OC in sediment and release CO_2 and CH_4 to the atmosphere. Although OC burial in reservoir sediments has increased by 300% whereas CO_2 emissions from inland waters have increased by 180%, CO_2 emissions are still estimated to represent a flux nearly twice that of OC burial (Regnier et al. 2013). The combined effects of increased sediment storage within reservoirs (Syvitski et al. 2005) and removal of LW likely have resulted in reduced POC fluxes to the ocean, although DOC fluxes may have increased (Regnier et al. 2013).

An important question in relation to OC storage in reservoir sediment is how human-induced changes in water bodies within river networks have changed the magnitude and distribution of stored OC. Annual burial rates of OC and IC tend to be highest in small, eutrophic lakes and impoundments and the concentration of OC in sediment is greatest in lakes with a low ratio of
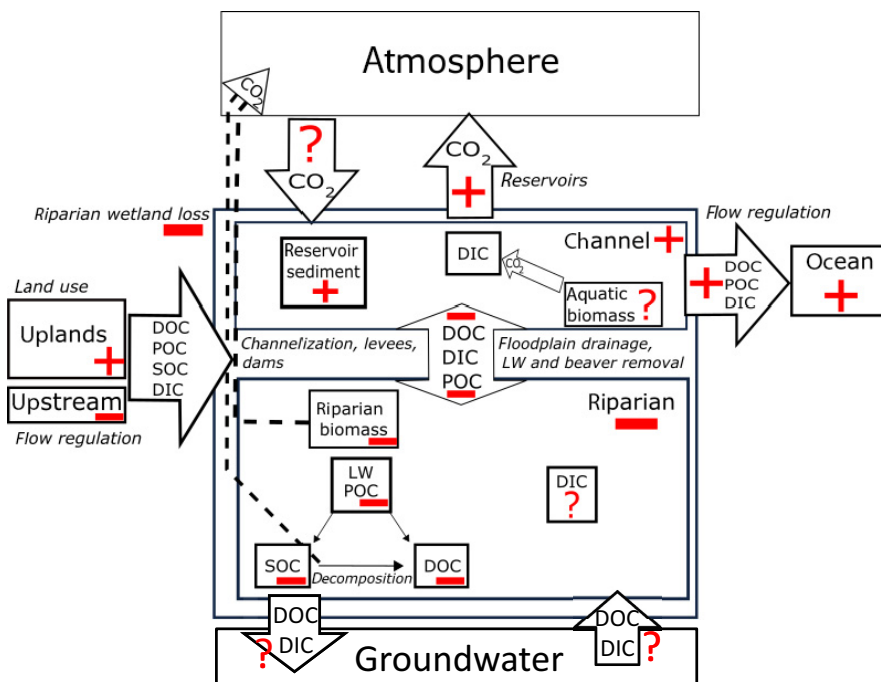

FIG. 8. A modified version of Fig. 5, highlighting the hypothesized net effects of human alterations as red plus (positive) or minus (negative) symbols or, where unknown, as red question marks. Human alteration of aquatic biomass, for example, includes increases in biomass associated with eutrophication and decreases associated with overfishing and alteration of channels in ways that minimize biomass (e.g., dredging). Consequently, the net global effect of human activities on aquatic biomass is unknown. Text in italic typeface lists the primary human activities resulting in the changes in carbon dynamics. The net effect of human activities is likely to be an increase in OC stored within reservoirs in channels and inundated riparian areas, and a decrease in OC stored in unimpounded riparian zones.

watershed to impoundment area (Downing et al. 2008). This finding suggests that small water bodies may be disproportionately important with respect to OC storage relative to their size. Severe reductions of beaver populations throughout Eurasia and North America have significantly reduced OC storage in beaver ponds, which have the potential for substantial OC concentrations in pond sediments and adjacent wet riparian areas (Naiman et al. 1986, 1988, Wohl 2013, Johnston 2014). On the other hand, the loss of beaver has been somewhat offset by construction of numerous small agricultural impoundments. These have greater sedimentation rates than natural lakes (Downing et al. 2008) and have presumably increased sediment storage of OC in river networks. Comparisons of natural and artificial impoundments indicate that reservoirs accumulate OC faster than natural lakes, although natural lakes cumulatively store a much greater volume of OC than reservoirs because the lakes are much older (Cole et al. 2007). The magnitude of OC once stored in small, natural beaver ponds and riparian wetlands vs. the magnitude of OC currently stored in small, agricultural impoundments and large reservoirs remains unknown.

Unlike impoundments, no global or even continental-scale analysis has quantified the geographic distribution or proportional or total values of channelization, levee construction, or alteration of riparian areas. Gleick et al. (2001) estimate that more than 500,000 km of inland waterways were altered for navigation globally during 1981–1990. They also estimate 50% wetland loss globally during 1900–1998, with regional values exceeding 50% in western Europe and the United States, although extensive wetland loss also occurred prior to 1900. These values are for all types of wetlands, including riparian or floodplain wetlands. We consider it reasonable to assume at least 50% floodplain wetland loss and alteration of native riparian vegetation along river corridors throughout high-income countries.

We originally proposed that dams create the most significant alteration of carbon dynamics within a channel, but alteration of riparian zones constitutes the most significant and altered aspect of carbon dynamics in river corridors. We use existing estimates of natural lake area (Verpoorter et al. 2014), reservoir area (St. Louis et al. 2000, Downing et al. 2006, Tranvik et al. 2009), and active channel area (Downing et al. 2012, Raymond et al. 2013) to roughly estimate the total global area of altered river corridors. Such estimates include substantial uncertainty because existing estimates of global channel area vary by a factor of 1.4 and estimates of reservoir area vary by a factor of 5.8 (Appendix S1: Table S1). Additional uncertainty comes from relatively limited measurements of riparian width. As a first approximation, we estimate global area in river corridors (active channel plus riparian zone) at somewhere between 2.6 and 6.2 million km^2. Using the existing upper and lower estimates of reservoir area and our upper and lower estimates of river corridor area indicates that reservoirs have altered anywhere from 4% to 70% of river corridors globally: we suggest that ~10% is a reasonably conservative estimate.

The assumptions of 10% river corridor alteration by dams and 50% river corridor alteration through other human activities underlie our inferences regarding the primary direct human effects on carbon dynamics in river corridors. Estimates of increased OC flux from rivers to oceans (e.g., Regnier et al. 2013) also support our inferences that reduced riparian OC storage outside of reservoirs has, at a minimum, a slightly greater net effect on OC balances in river corridors than increased OC storage in reservoir sediments. We submit that alteration of land cover and consequent changes in soil erosion and OC influxes to river corridors are likely the largest indirect human effect on carbon dynamics in river corridors.

Returning to Eq. 1, we add upward and downward arrows to indicate an increase or reduction, respectively, in the magnitude of the terms in the equation as a result of the cumulative effects of human activities:

$$\downarrow \frac{dC_S}{dt} = \uparrow C_I - \uparrow C_{Ogas} + \uparrow C_{Oriver} \qquad (4)$$

Despite substantial uncertainties in the magnitude of individual human alterations and the interacting effects of multiple forms of alteration, the synthesis presented here suggests the shortcomings of the existing active and neutral pipe models of river channels in the global carbon cycle.

Conclusions

Diverse lines of evidence indicate the most likely scenario for cumulative effects of human activities on OC dynamics in river corridors. Widespread removal of native land cover and alteration of topography dramatically increase OC inputs to river corridors, primarily through increased soil erosion. Where these inputs affect small rivers without dams, levees, or continuing agricultural use of the riparian zone, the net effect may be increased OC storage in riparian soils (e.g., Ricker et al. 2012). Where increased upland inputs enter larger rivers, however, the net effect may be different. Channelized, physically simpler and more uniform channels provide fewer opportunities for temporary storage and biological processing of OC inputs (Battin et al. 2008, Aufdenkampe et al. 2011), and laterally disconnected riparian areas store far less OC in riparian soil and biomass (e.g., Bullinger-Weber et al. 2014, Hanberry et al. 2015). These changes in channels and riparian areas lead to increased downstream fluxes of POC and DOC and increased CO_2 fluxes to the atmosphere. However, POC is trapped within the numerous artificial reservoirs now present along most channels and stored in reservoir sediments. Model simulations suggest that the net effect of these changes within uplands and river networks is increased DOC flux to the ocean, increased CO_2 flux to the atmosphere, and a likely decrease in POC flux to the ocean.

CONCEPTS & SYNTHESIS

At least four fundamental knowledge gaps exist in our understanding of OC dynamics in river corridors. First, we lack quantitative estimates of how OC inputs to river corridors have varied over timescales of 10^1–10^3 yr as a result of human activities. This uncertainty applies to individual watersheds and regions, as well as globally. Although many studies have estimated changes in global sediment fluxes (Hooke 2000, Syvitski et al. 2005, Syvitski and Kettner 2011), these estimates have not been effectively coupled with knowledge of SOC levels and terrestrial biomass to quantify changes in OC fluxes to river corridors.

Second, we have only limited quantitative estimates of OC riparian storage, resulting from either OC fixed in the riparian zone or transported to the riparian zone from the channel, in relation to biome and position within a river network, as well as how such storage has changed under human modification of riparian areas. One might reasonably assume based on knowledge of SOC content by biome (Tarnocai et al. 2009, Schuur et al. 2015), for example, that large floodplain rivers at high latitudes would store most riparian OC. However, relatively small riparian areas within temperate latitudes can contain very high levels of SOC (Walter and Merritts 2008, Appling 2012, Wohl et al. 2012b), and 70–80% of total river (and riparian) length in most river networks is in lower order channels (Downing et al. 2012).

A third fundamental knowledge gap is our ignorance of the magnitude and geographic distribution of alterations in riparian areas at regional to global scales. A few studies estimate the spatial extent and magnitude of changes in riparian SOC content and biomass for a particular region (e.g., Hanberry et al. 2015), but no comprehensive, global synthesis has been undertaken to identify regions with extensive historical or ongoing riparian modification, or the implications of these modifications for carbon dynamics. Further, our admittedly coarse, first approximation of the effects of altering river corridors and disconnecting them from their channels for OC influxes illustrates the limitations of the existing active pipe and neutral pipe models of OC dynamics in rivers.

Finally, we lack adequate understanding of how diverse environmental variables might interact to influence the net effect of climate or land use change on Eq. 1 for diverse river segments and entire watersheds. Given the strong likelihood of nonlinear interactions among water and sediment yield to a river corridor, flow regime, soil moisture, primary productivity, CO_2 emissions, and SOC content in response to climate warming of 2°C, for example, we need many more studies focused on the mechanics of OC fluxes and transformations within river corridors.

The primary management implication to emerge from our study is the critical need to protect and restore the physical complexity of river corridors, including the lateral connectivity between channels and riparian areas. The importance of lateral connectivity and ecosystem integrity in riparian areas has previously been emphasized in many contexts, including enhanced flood controls (Opperman et al. 2010), reduced nutrient loading to channels (Zhang and Mitsch 2007), protecting fish stocks (Ogston et al. 2015), and reducing fine sediment concentrations in rivers (Fitzpatrick et al. 2009). To these we can now add OC storage in riparian soil and biomass. Current emphases on upland afforestation as a means of carbon sequestration (Van der Gaast et al. 2016) should be expanded to explicitly include riparian areas.

Literature Cited

Abril, G., et al. 2014. Amazon River carbon dioxide outgassing fueled by wetlands. Nature 505:395–398.

Alexander, R. B., E. W. Boyer, R. A. Smith, G. E. Schwarz, and R. B. Moore. 2007. The role of headwater streams in downstream water quality. JAWRA Journal of the American Water Resources Association 43:41–59.

Alin, S. R., R. Aalto, M. A. Goni, J. E. Richey, and W. E. Dietrich. 2008. Biogeochemical characterization of carbon sources in the Strickland and Fly Rivers, Papua New Guinea. Journal of Geophysical Research 113:F01S05.

Amon, R. M. W., et al. 2012. Dissolved organic matter sources in large Arctic rivers. Geochimica et Cosmochimica Acta 94:217–237.

Anderson, N. J., H. Bennion, and A. F. Lotter. 2014. Lake eutrophication and its implications for organic carbon sequestration in Europe. Global Change Biology 20:2741–2751.

Appling, A. P. 2012. Connectivity drives function: carbon and nitrogen dynamics in a floodplain-aquifer ecosystem. Dissertation. Duke University, Durham, North Carolina, USA.

Armentano, T. V. 1980. Drainage of organic soils as a factor in the world carbon cycle. BioScience 30:825–830.

Aufdenkampe, A. K., E. Mayorga, P. A. Raymond, J. M. Melack, S. C. Doney, S. R. Alin, R. E. Aalto, and K. Yoo. 2011. Riverine coupling of biogeochemical cycles between land, oceans, and atmosphere. Frontiers in Ecology and the Environment 9:53–60.

Barlow, P. M., and S. A. Leake. 2012. Streamflow depletion by wells—understanding and managing the effects of groundwater pumping on streamflow. Circular 1376. U.S. Geological Survey, Reston, Virginia.

Barnes, R. T., and P. A. Raymond. 2009. The contribution of agricultural and urban activities to inorganic carbon fluxes within temperate watersheds. Chemical Geology 266:318–327.

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

Bates, B. C., Z. W. Kundzewicz, and J. P. Palutikof. 2008. Climate change and water (Technical Report No. VI). IPCC, IPCC Secretariat, Geneva, Switzerland.

Battin, T. J., L. A. Kaplan, S. Findlay, C. S. Hopkinson, E. Marti, A. I. Packman, J. D. Newbold, and F. Sabater. 2008. Biophysical controls on organic carbon fluxes in fluvial networks. Nature Geoscience 1:95–100.

Battin, T. J., S. Luyssaert, L. A. Kaplan, A. K. Aufdenkampe, A. Richter, and L. J. Tranvik. 2009. The boundless carbon cycle. Nature Geoscience 2:598–600.

Beaulieu, J. J., et al. 2015. Urban stream burial increases watershed-scale nitrate export. PLoS ONE 10:e0132256.

Beckman, N. D., and E. Wohl. 2014. Carbon storage in mountainous headwater streams: the role of old-growth forest and logjams. Water Resources Research 50:2376–2393.

Bellmore, J. R., and C. V. Baxter. 2014. Effects of geomorphic process domains on river ecosystems: a comparison of floodplain and confined valley segments. River Research and Applications 30:617–630.

Berhe, A. A., J. Harte, J. W. Harden, and M. S. Torn. 2007. The significance of the erosion-induced terrestrial carbon sink. BioScience 57:337–346.

Bernhardt, E. S., and M. A. Palmer. 2011. The environmental costs of mountaintop mining valley fill operations for aquatic ecosystems of the Central Appalachians. Annals of the New York Academy of Sciences 1223:39–57.

Bilby, R. E., B. R. Fransen, and P. A. Bisson. 1995. Incorporation of nitrogen and carbon from spawning coho salmon into the trophic system of small streams: evidence from stable isotopes. Canadian Journal of Fisheries and Aquatic Sciences 53:164–173.

Bilby, R. E., and G. E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61:1107–1113.

Borges, A. V., G. Abril, F. Darchambeau, C. R. Teodoru, J. Deborde, L. O. Vidal, T. Lambert, and S. Bouillon. 2015. Divergent biophysical controls of aquatic CO_2 and CH_4 in the world's two largest rivers. Scientific Reports 5:15614.

Borges, A. V., B. Delille, and M. Frankignoulle. 2005. Budgeting sinks and sources of CO_2 in the coastal ocean: diversity of ecosystem counts. Geophysical Research Letters 32:1–4.

Bouillon, S., A. Yambele, R. G. M. Spencer, D. P. Gillikin, P. J. Hernes, J. Six, R. Merckx, and A. V. Borges. 2012. Organic matter sources, fluxes and greenhouse gas exchange in the Oubangui River (Congo River basin). Biogeosciences 9:2045–2062.

Bouillon, S., et al. 2009. Distribution, origin and cycling of carbon in the Tana River (Kenya): a dry season basin-scale survey from headwaters to the delta. Biogeosciences 6:2475–2493.

Bragg, D. C. 2000. Simulating catastrophic and individualistic large woody debris recruitment for a small riparian system. Ecology 81:1363–1394.

Brezonik, P. L., and W. A. Arnold. 2011. Water chemistry. Oxford University Press, New York, New York, USA.

Brooks, A. P., T. Howell, T. B. Abbe, and A. H. Arthington. 2006. Confronting hysteresis: wood based river rehabilitation in highly altered riverine landscapes of south-eastern Australia. Geomorphology 79:395–422.

Bullinger-Weber, G., R.-C. Le Bayon, A. Thebault, R. Schlaepfer, and C. Guenat. 2014. Carbon storage and soil organic matter stabilization in near-natural, restored and embanked Swiss floodplains. Geoderma 228–229:122–131.

Bump, J. K., B. G. Bergman, A. J. Schrank, A. M. Marcarelli, E. S. Kane, A. C. Risch, and M. Schütz. 2016. Nutrient release from moose bioturbation in aquatic ecosystems. Oikos. doi:10.1111/oik.03591

Burkham, D. E. 1976. Hydraulic effects of changes in bottomland vegetation on three major floods, Gila River in southeastern Arizona. Professional Paper 655-J, U.S. Geological Survey, Washington, D.C., USA.

Butman, D., and P. A. Raymond. 2011. Significant efflux of carbon dioxide from streams and river in the United States. Nature Geoscience 4:839–842.

Butman, D., S. Stackpoole, E. Stets, C. P. McDonald, D. W. Clow, and R. G. Striegl. 2016. Aquatic carbon cycling in the conterminous United States and implications for terrestrial carbon accounting. Proceedings of the National Academy of Sciences USA 113:58–63.

Cai, W. J. 2003. Riverine inorganic carbon flux and rate of biological uptake in the Mississippi River plume. Geophysical Research Letters 30:1032.

Canuel, E. A., S. S. Cammer, H. A. McIntosh, and C. R. Pondell. 2012. Climate change impacts on the organic carbon cycle at the land-ocean interface. Annual Review of Earth and Planetary Sciences 40:685–711.

Casas-Ruiz, J. P., N. Catalán, L. Gómez-Gener, D. von Schiller, B. Obrador, D. N. Kothawala, P. López, S. Sabater, and R. Marcé. 2017. A tale of pipes and reactors: controls on the in-stream dynamics of dissolved organic matter in rivers. Limnology and Oceanography. https://doi.org/10.1002/lno.10471

Catalán, N., A. M. Kellerman, H. Peter, F. Carmona, and L. J. Tranvik. 2015. Absence of a priming effect on dissolved organic carbon degradation in lake water. Limnology and Oceanography 60:159–168.

Chauvaud, L., J. K. Thompson, J. E. Cloern, and G. Thouzeau. 2003. Clams as CO_2 generators: the Potamocorbula amurensis example in San Francisco Bay. Limnology and Oceanography 48:2086–2092.

Cierjacks, A., B. Kleinschmit, M. Babinsky, F. Kleinschroth, A. Markert, M. Menzel, U. Ziechmann, T. Schiller, M. Graf, and F. Lang. 2010. Carbon stocks of soil and vegetation on Danubian floodplains. Journal of Plant Nutrition and Soil Science 173:644–653.

Cole, J. J., N. F. Caraco, G. W. Kling, and T. K. Kratz. 1994. Carbon dioxide supersaturation in the surface waters of lakes. Science 265:1568–1570.

Cole, J. J., et al. 2007. Plumbing the global carbon cycle: integrating inland waters into the terrestrial carbon budget. Ecosystems 10:171–184.

Collins, B. D., D. R. Montgomery, K. L. Fetherston, and T. B. Abbe. 2012. The floodplain large-wood cycle hypothesis: a mechanism for the physical and biotic structuring of temperate forested alluvial valleys in the North Pacific coastal ecoregion. Geomorphology 139–140:460–470.

Comiti, F. 2012. How natural are Alpine mountain rivers? Evidence from the Italian Alps. Earth Surface Processes and Landforms 37:693–707.

Conley, D. J., H. W. Paerl, R. W. Howarth, D. F. Boesch, S. P. Seitzinger, K. E. Havens, C. Lancelot, and G. E. Likens. 2009. Ecology controlling eutrophication: nitrogen and phosphorus. Science 323:1014–1015.

Crawford, J. T., M. M. Dornblaser, E. H. Stanley, D. W. Clow, and R. G. Striegl. 2015. Source limitation of carbon gas emissions in high-elevation mountain streams and lakes. Journal of Geophysical Research—Biogeosciences 120:952–964.

Crawford, J. T., L. C. Loken, E. G. Stanley, E. G. Stets, M. M. Dornblaser, and R. G. Striegl. 2016. Basin scale controls on CO_2 and CH_4 emissions from the Upper Mississippi River. Geophysical Research Letters 43:1973–1979.

Daniel, M. H. B., A. A. Montebelo, M. C. Bernardes, J. P. H. B. Ometto, P. de Camargo, A. V. Krusche, M. V. Ballester, R. L. Victoria, and L. A. Martinelli. 2002. Effects of urban sewage on dissolved oxygen, dissolved inorganic and organic carbon, and electrical conductivity of small streams along a gradient of urbanization in the Piracicaba River basin. Water, Air, and Soil Pollution 136:189–206.

Davidson, E. A., and I. L. Ackerman. 1993. Changes in soil carbon inventories following cultivation of previously untilled soils. Biogeochemistry 20:161–193.

CONCEPTS & SYNTHESIS

Davidson, E. A., and I. A. Janssens. 2006. Temperature sensitivity of soil carbon decomposition and feedbacks to climate change. Nature 440:165–173.

Day, N. K. 2015. Nitrogen cycling in headwater streams. Unpublished Thesis. University of Wyoming, Laramie, Wyoming, USA.

Deemer, B. R., J. A. Harrison, S. Li, J. J. Beaulieu, T. Del Sontro, N. Barros, J. F. Bezerra-Neto, S. M. Powers, M. A. dos Santos, and J. A. Vonk. 2016. Greenhouse gas emissions from reservoir water surfaces: a new global synthesis. BioScience 66:949–964.

DeLong, M. D., and M. A. Brusven. 1994. Allochthonous input of organic matter from different riparian habitats of an agriculturally impacted stream. Environmental Management 18:59–71.

Devito, K. J., D. Fitzgerald, A. R. Hill, and R. Aravena. 2000. Nitrate dynamics in relation to lithology and hydrologic flow path in a river riparian zone. Journal of Environmental Quality 29:1075–1084.

Divers, M. T., E. M. Elliott, and D. J. Bain. 2013. Constraining nitrogen inputs to urban streams from leaking sewers using inverse modeling: implications for dissolved inorganic nitrogen (DIN) retention in urban environments. Environmental Science and Technology 47:1816–1823.

Doetterl, S., A. A. Berhe, E. Nadeu, Z. Wang, M. Sommer, and P. Fiener. 2016. Erosion, deposition and soil carbon: a review of process-level controls, experimental tools and models to address C cycling in dynamic landscapes. Earth-Science Reviews 154:102–122.

Doetterl, S., J.-T. Cornelis, J. Six, S. Bode, S. Opfergelt, P. Boeckx, and K. Van Oost. 2015. Soil redistribution and weathering controlling the fate of geochemical and physical carbon stabilization mechanisms in soils of an eroding landscape. Biogeosciences 12:1357–1371.

Dommain, R., J. Couwenberg, P. H. Glaser, H. Joosten, and I. N. N. Suryadiputra. 2014. Carbon storage and release in Indonesian peatlands since the last deglaciation. Quaternary Science Reviews 97:1–32.

Donato, D. C., J. B. Kauffman, D. Murdiyarso, S. Kurnianto, M. Stidham, and M. Kanninen. 2011. Mangroves among the most carbon-rich forests in the tropics. Nature Geoscience 4:293–297.

Downing, J. A., J. J. Cole, C. M. Duarte, J. J. Middelburg, J. M. Melack, Y. T. Prairie, P. Kortelainen, R. G. Striegl, W. H. McDowell, and L. J. Tranvik. 2012. Global abundance and size distribution of streams and rivers. Inland Waters 2: 229–236.

Downing, J. A., J. J. Cole, J. J. Middelburg, R. G. Striegl, C. M. Duarte, P. Kortelainen, Y. T. Prairie, and K. A. Laube. 2008. Sediment organic carbon burial in agriculturally eutrophic impoundments over the last century. Global Biogeochemical Cycles 22:GB1018.

Downing, J. A., et al. 2006. The global abundance and size distribution of lakes, ponds, and impoundments. Limnology and Oceanography 51:2388–2397.

Duffkova, R., T. Kvitek, and J. Voldrichova. 2005. Soil organic carbon and nitrogen characteristics in differently used grasslands at sites with drainage and without drainage. Plant Soil and Environment 51:165–172.

Elmore, A. J., and S. S. Kaushal. 2008. Disappearing headwaters: patterns of stream burial due to urbanization. Frontiers in Ecology and the Environment 6:308–312.

Fearnside, P. M., and S. Pueyo. 2012. Greenhouse-gas emissions from tropical dams. Nature Climate Change 2:382–384.

Fenn, M. E., et al. 2003. Ecological effects of nitrogen deposition in the Western United States. BioScience 53:404.

Findlay, S. E. 2005. Increased carbon transport in the Hudson River: Unexpected consequence of nitrogen deposition? Frontiers in Ecology and the Environment 3:133–137.

Fischlin, A., G. F. Midgley, J. T. Price, R. Leemans, B. Gopal, C. Turley, M. D. A. Rounsevell, O. P. Dube, J. Tarazona, and A. A. Velichko. 2007. Ecosystems, their properties, goods, and services. Pages 211–272 in M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden, and C. E. Hanson, editors. Climate change 2007: impacts, adaptation and vulnerability, contribution of working group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, UK.

Fisher, S., and G. Likens. 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. Ecological Monographs 43:421–439.

Fitzpatrick, F. A., J. C. Knox, and J. P. Schubauer-Berigan. 2009. Channel, floodplain, and wetland responses to flood and overbank sedimentation, 1846-2006, Halfway Creek Marsh, Upper Mississippi Valley, Wisconsin. Geological Society of America Special Paper 451:23–42.

Fontaine, S., S. Barot, P. Barre, N. Bdioui, B. Mary, and C. Rumpel. 2007. Stability of organic carbon in deep soil layers controlled by fresh carbon supply. Nature 450:277–280.

Galy, V., O. Beyssac, C. France-Lanord, and T. Eglinton. 2008a. Recycling of graphite during Himalayan erosion: a geological stabilization of carbon in the crust. Science 322: 943–945.

Galy, V., C. France-Lanord, and B. Lartiges. 2008b. Loading and fate of particulate organic carbon from the Himalaya to the Ganga-Brahmaputra delta. Geochimica et Cosmochimica Acta 72:1767–1787.

Galy, V., B. Peucker-Ehrenbrink, and T. Eglinton. 2015. Global carbon export from the terrestrial biosphere controlled by erosion. Nature 521:204–207.

Giese, L. A. B. 2001. Carbon pools and fluxes as an indicator of riparian restoration. Dissertation. Virginia Polytechnic Institute and State University, Blacksburg, Virginia, USA.

Gleick, P. H., A. Singh, and H. Shi. 2001. Threats to the world's freshwater resources. Pacific Institute for Studies in Development, Environment, and Security, Oakland, California, USA.

Gomez, B., W. T. Baisden, and K. M. Rogers. 2010. Variable composition of particle-bound organic carbon in steepland river systems. Journal of Geophysical Research 115:F04006.

Goñi, M. G., J. A. Hatten, R. A. Wheatcroft, and J. C. Borgeld. 2013. Particulate organic matter export by two contrasting small mountainous rivers from the Pacific Northwest, USA. Journal of Geophysical Research—Biogeosciences 118:112–134.

Gonor, J. J., J. R. Sedell, and P. A. Benner. 1988. What we know about large trees in estuaries, in the sea, and on coastal beaches. Pages 83–112 in C. Maser, R. F. Tarrant, J. M. Trappe, and J. F. Franklin, editors. From the forest to the sea: a story of fallen trees. General Technical Report PNW-GTR-229, USDA Forest Service, Portland, Oregon, USA.

Gordeev, V. V., and M. D. Kravchishina. 2009. River flux of dissolved organic carbon (DOC) and particulate carbon (POC) to the Arctic Ocean: What are the consequences of the global changes? Pages 145–160 in J. C. J. Nihoul, and A. G. Kostianov, editors. Influence of climate change on the changing Arctic and Sub-Arctic conditions. Springer Science, Dordrecht, The Netherlands.

Graf, W. L. 1978. Fluvial adjustments to the spread of tamarisk in the Colorado Plateau region. Geological Society of America Bulletin 89:1491–1501.

Graf, W. L. 2001. Damage control: restoring the physical integrity of America's rivers. Annals of the Association of American Geographers 91:1–27.

CONCEPTS & SYNTHESIS

Green, P. A., C. J. Vorosmarty, M. Meybeck, J. N. Galloway, B. J. Peterson, and E. W. Boyer. 2004. Pre-industrial and contemporary fluxes of nitrogen through rivers: a global assessment based on typology. Biogeochemistry 68:71–105.

Griffin, E. R., J. W. Kean, K. R. Vincent, J. D. Smith, and J. M. Friedman. 2005. Modeling effects of bank friction and woody bank vegetation on channel flow and boundary shear stress in the Rio Puerco, New Mexico. Journal of Geophysical Research 110:F04023.

Grill, G., B. Lehner, A. E. Lumsdon, G. K. MacDonald, C. Zarfl, and C. R. Liermann. 2015. An index-based framework for assessing patterns and trends in river fragmentation and flow regulation by global dams at multiple scales. Environmental Research Letters 10:015001.

Guenet, B., M. Danger, L. Abbadie, and G. Lacroix. 2010. Priming effect: bridging the gap between terrestrial and aquatic ecology. Ecology 91:2850–2861.

Gurwick, N. P., P. M. Groffman, J. B. Yavitt, A. J. Gold, G. Blazejewski, and M. Stolt. 2008. Microbially available carbon in buried riparian soils in a glaciated landscape. Soil Biology & Biochemistry 40:85–96.

Hall, R. O., J. A. Tank, M. A. Baker, E. J. Rosi-Marshall, and E. R. Hotchkiss. 2016. Metabolism, gas exchange, and carbon spiraling in rivers. Ecosystems 19:73–86.

Hanberry, B. B., J. M. Kabrick, and H. S. He. 2015. Potential tree and soil carbon storage in a major historical floodplain forest with disrupted ecological function. Perspectives in Plant Ecology, Evolution and Systematics 17:17–23.

Handique, S. 2015. A review on the riverine carbon sources, fluxes and perturbations. Pages 417–428 in M. Ramkumar, K. Kumaraswamy, and R. Mohanraj, editors. Environmental management of river basin ecosystems. Springer Earth System Sciences, Switzerland.

Harmon, M. E., et al. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133–302.

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

Hatten, J. A., M. A. Goñi, and R. A. Wheatcroft. 2012. Chemical characteristics of particulate organic matter from a small, mountainous river system in the Oregon Coast Range, USA. Biogeochemistry 107:43–66.

Hatten, J., J. Sloan, B. Frey, J. Straub, R. Kaminski, and A. Ezell. 2014. Soil and sediment carbon and nitrogen in Mississippi alluvial valley and interior flatwoods bottomlands. Soil Science Society of America Journal 78:S248–S260.

Heathcote, A. J., N. J. Anderson, Y. T. Prairie, D. R. Engstrom, and P. A. del Giorgio. 2015. Large increases in carbon burial in northern lakes during the Anthropocene. Nature Communications 6:10016.

Heathcote, A. J., and J. A. Downing. 2012. Impacts of eutrophication on carbon burial in freshwater lakes in an intensively agricultural landscape. Ecosystems 15:60–70.

Hem, J. D. 1985. Study and interpretation of the chemical characteristics of natural water. Third edition. Water Supply Paper 2254. U.S. Geological Survey, Reston, VA.

Hester, E. T., and M. W. Doyle. 2008. In-stream geomorphic structures as drivers of hyporheic exchange. Water Resources Research 44:W03417.

Hicke, J. A., et al. 2012. Effects of biotic disturbance on forest carbon cycling in the United States and Canada. Global Change Biology 18:7–34.

Hill, A. R. 1996. Nitrate removal in stream riparian zones. Journal of Environmental Quality 25:743–755.

Hill, A. R., K. J. Devito, S. Campagnolo, and K. Sanmugadas. 2000. Subsurface denitrification in a forest riparian zone: interactions between hydrology and supplies of nitrate and organic carbon. Biogeochemistry 51:193–223.

Hilmes, M. M., and E. E. Wohl. 1995. Changes in channel morphology associated with placer mining. Physical Geography 16:223–242.

Hilton, R. G., V. Glay, J. Gaillardet, M. Dellinger, C. Bryant, M. O'Regan, D. R. Gröcke, H. Coxall, J. Bouchez, and D. Calmels. 2015. Erosion of organic carbon in the Arctic as a geological carbon dioxide sink. Nature 524:84–87.

Hilton, R. G., A. Galy, and N. Hovius. 2008a. Riverine particulate organic carbon from an active mountain belt: importance of landslides. Global Biogeochemical Cycles 22:GB1017.

Hilton, R. G., A. Galy, N. Hovius, M.-C. Chen, M.-J. Horng, and H. Chen. 2008b. Tropical-cyclone-driven erosion of the terrestrial biosphere from mountains. Nature Geoscience 1:759–762.

Hilton, R. G., A. Galy, N. Hovius, M.-J. Horng, and H. Chen. 2011a. Efficient transport of fossil organic carbon to the ocean by steep mountain rivers: an orogenic carbon sequestration mechanism. Geology 39:71–74.

Hilton, R. G., A. Galy, N. Hovius, S.-J. Kao, M.-J. Horng, and H. Chen. 2012. Climatic and geomorphic controls on the erosion of terrestrial biomass from subtropical mountain forest. Global Biogeochemical Cycles 26:GB3014.

Hilton, R. G., P. Meunier, N. Hovius, P. J. Bellingham, and A. Galy. 2011b. Landslide impact on organic carbon cycling in a temperate montane forest. Earth Surface Processes and Landforms 36:1670–1679.

Hoffmann, T., S. Glatzel, and R. Dikau. 2009. A carbon storage perspective on alluvial sediment storage in the Rhine catchment. Geomorphology 108:127–137.

Hoffmann, T., V. R. Thorndycraft, A. G. Brown, T. J. Coulthard, B. Damnati, V. S. Kale, H. Middelkoop, B. Notebaert, and D. E. Walling. 2010. Human impact on fluvial regimes and sediment flux during the Holocene: review and future research agenda. Global and Planetary Change 72:87–98.

Hoffmann, T., et al. 2013. Short communication: humans and the missing C-sink: erosion and burial of soil carbon through time. Earth Surface Dynamics Discussions 1:93–112.

Hooke, R. L. B. 2000. On the history of humans as geomorphic agents. Geology 28:843–846.

Hotchkiss, E. R., and R. O. Hall. 2010. Linking calcification by exotic snails to stream inorganic carbon cycling. Oecologia 163:235–244.

Hotchkiss, E. R., R. O. Hall, and M. A. Baker. 2014. Modeling priming effects on microbial consumption of dissolved organic carbon in rivers. Journal of Geophysical Research—Biogeosciences. https://doi.org/10.1002/2013jg002599

Hotchkiss, E. R., R. O. Hall, R. A. Sponseller, D. Butman, J. Klaminder, H. Laudon, M. Rosvall, and J. Karlsson. 2015. Sources of and processes controlling CO_2 emissions change with the size of streams and rivers. Nature Geoscience 8:696–699.

Hovius, N., A. Galy, R. G. Hilton, R. Sparkes, J. Smith, K. Shuh-Ji, C. Hongey, L. In-Tian, and A. J. West. 2011. Erosion-driven drawdown of atmospheric carbon dioxide: the organic pathway. Applied Geochemistry 26:5285–5287.

Howarth, R. W., E. W. Boyer, W. J. Pabich, and J. N. Galloway. 2002. Nitrogen use in the United States from 1961–2000 and potential future trends. AMBIO: A Journal of the Human Environment 31:88–96.

Hugelius, G., et al. 2014. Estimated stocks of circumpolar permafrost carbon with quantified uncertainty ranges and identified data gaps. Biogeosciences 11:6573–6593.

Hunter, W. R., R. Niederdorfer, A. Gernand, B. Veuger, J. Prommer, M. Mooshammer, W. Wanek, and T. J. Battin. 2016. Metabolism of mineral-sorbed organic matter and microbial lifestyles in fluvial ecosystems. Geophysical Research Letters 43:1582–1588.

CONCEPTS & SYNTHESIS

IPCC. 2001. The carbon cycle and atmospheric carbon dioxide. Page 188 *in* IPCC, editor. Climate change 2001. Cambridge University Press, Cambridge, UK.

Jaeger, K. L. 2015. Reach-scale geomorphic differences between headwater streams draining mountaintop mined and unmined catchments. Geomorphology 236:25–33.

Jahnke, R. A. 1996. The global ocean flux of particulate organic carbon: areal distribution and magnitude. Global Biogeochemical Cycles 10:71–88.

James, L. A. 1999. Time and the persistence of alluvium: river engineering, fluvial geomorphology, and mining sediment in California. Geomorphology 31:265–290.

Jaramillo, V. J., J. B. Kauffman, L. Rentería-Rodríguez, D. L. Cummings, and L. J. Ellingson. 2003. Biomass, carbon, and nitrogen pools in Mexican tropical dry forest landscapes. Ecosystems 6:609–629.

Jeung, J. J., S. Bartsch, J. H. Felckenstein, E. Matzner, J. D. Tenhunen, S. D. Lee, S. K. Park, and J. H. Park. 2012. Differential storm responses of dissolved and particulate organic carbon in a mountainous headwater stream, investigated by high-frequency, in situ optical measurements. Journal of Geophysical Research—Biogeosciences 117:G03013.

Johnson, M. S., J. Lehmann, S. J. Riha, A. V. Krusche, J. E. Richey, J. P. H. B. Ometto, and E. G. Couto. 2008. CO_2 efflux from Amazonian headwater streams represents a significant fate for deep soil respiration. Geophysical Research Letters 35:L17401.

Johnston, C. A. 2014. Beaver pond effects on carbon storage in soils. Geoderma 213:371–378.

Jung, B. J., J. K. Lee, H. Kim, and J. H. Park. 2014. Export, biodegradation, and disinfection byproduct formation of dissolved and particulate organic carbon in a forested headwater stream during extreme rainfall events. Biogeosciences 11:6119–6129.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Special Publications of the Canadian Journal of Fisheries and Aquatic Sciences 106:110–127.

Kane, E. S., E. S. Kasischke, D. W. Valentine, M. R. Turetsky, and A. D. McGuire. 2007. Topographic influences on wildfire consumption of soil organic carbon in interior Alaska: implications for black carbon accumulation. Journal of Geophysical Research—Biogeosciences 112:G03017.

Kindle, E. M. 1921. Mackenzie River driftwood. Geographical Review 11:50–53.

Knox, S. H., C. Sturtevant, J. H. Matthes, L. Koteen, J. Verfaillie, and D. Baldocchi. 2015. Agricultural peatland restoration: effects of land-use change on greenhouse gas (CO_2 and CH_4) fluxes in the Sacramento-San Joaquin Delta. Global Change Biology 21:750–765.

Kramer, N., E. Wohl, B. Hess-Homeier, and S. Leisz. *In press.* The pulse of driftwood over multiple timescales in a great northern river. Water Resources Research 53. doi:10.1002/2016WR019260

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304:1623–1627.

Larsen, M. C., and J. E. Parks. 1997. How wide is a road? The association of roads and mass-wasting in a forested montane environment. Earth Surface Processes and Landforms 22:835–848.

Lauerwald, R., J. Hartmann, W. Ludwig, and N. Moosdorf. 2012. Assessing the nonconservative fluvial fluxes of dissolved organic carbon in North America. Journal of Geophysical Research—Biogeosciences 117. https://doi.org/10.1029/2011jg001820

Lehner, B., D. R. Liermann, C. Revenga, C. Vörösmarty, B. Fekete, P. Crouzet, P. Doll, M. Endejan, K. Frenken, J. Magome, C. Nilsson, et al. 2011. High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. Frontiers in Ecology and the Environment 9:494–502.

Leithold, E. L., N. E. Blair, and D. W. Perkey. 2006. Geomorphologic controls on the age of particulate organic carbon from small mountainous and upland rivers. Global Biogeochemical Cycles 30:GB3022.

Leithold, E. L., N. E. Blair, and K. W. Wegmann. 2015. Source to sink sedimentary systems and the global C-cycle: a river runs through it. Earth-Science Reviews 153:30–42.

Lindberg, T. T., E. S. Bernhardt, R. Bier, A. Helton, R. B. Merola, A. Vengosh, and R. T. Di Giulio. 2011. Cumulative impacts of mountaintop mining on an Appalachian watershed. Proceedings of the National Academy of Sciences USA 108:20929–20934.

Lininger, K. B., E. Wohl, N. A. Sutfin, and J. R. Rose. *In press.* Floodplain downed wood volumes: a comparison across three biomes. Earth Surface Processes and Landforms. doi:10.1002/esp.4072

Livers, B., and E. Wohl. 2015. An evaluation of stream characteristics in glacial versus fluvial process domains in the Colorado Front Range. Geomorphology 231:72–82.

Lloret, E., C. Dessert, L. Pastor, E. Lajeunesse, O. Crispi, J. Gaillardet, and M. F. Benedetti. 2013. Dynamic of particulate and dissolved organic carbon in small volcanic mountainous tropical watersheds. Chemical Geology 351:229–244.

Lyons, W. B., C. A. Nezat, A. E. Carey, and D. M. Hicks. 2002. Organic carbon fluxes to the ocean from high-standing islands. Geology 30:443–446.

Madej, M. A. 2010. Redwoods, restoration, and implications for carbon budgets. Geomorphology 116:264–273.

Madej, M. A., J. Seney, and P. van Mantgem. 2013. Effects of road decommissioning on carbon stocks, losses, and emissions in north coastal California. Restoration Ecology 21:439–446.

Mann, P. J., T. I. Eglinton, C. P. McIntyre, N. Zimov, A. Davydova, J. E. Vonk, R. M. Holmes, and R. G. M. Spencer. 2015. Utilization of ancient permafrost carbon in headwaters of Arctic fluvial networks. Nature Communications 6:7856.

Marcus, W. A., G. A. Meyer, and D. R. Nimmo. 2001. Geomorphic control of persistent mine impacts in a Yellowstone Park stream and implications for the recovery of fluvial systems. Geology 29:355–358.

Marwick, T. R., F. Tamooh, C. R. Teodoru, A. V. Borges, F. Darchambeau, and S. Bouillon. 2015. The age of river-transported carbon: a global perspective. Global Biogeochemical Cycles 29:122–137.

McClelland, J. W., et al. 2016. Particulate organic carbon and nitrogen export from major Arctic rivers. Global Biogeochemical Cycles 30:629–643.

McSherry, M. E., and M. E. Ritchie. 2013. Effects of grazing on grassland soil carbon: a global review. Global Change Biology 19:1347–1357.

Mertes, L. A. K., T. Dunne, and L. A. Martinelli. 1996. Channel-floodplain geomorphology along the Solimoes-Amazon River, Brazil. Geological Society of America Bulletin 108:1089–1107.

Meybeck, M. 1982. Carbon, nitrogen, and phosphorus transport by world rivers. American Journal of Science 282:401–450.

Meybeck, M. 1993. Riverine transport of atmospheric carbon: sources, global typology and budget. Water, Air, and Soil Pollution 70:443–463.

Meybeck, M. 2003. The global occurrence of major elements in rivers. Pages 207–223 *in* J. Drever, editor. Treatise on geochemistry. Elsevier, Dodrecht, The Netherlands.

Meyer, J. L., A. C. Benke, R. T. Edwards, and J. B. Wallace. 1997. Organic matter dynamics in the Ogeechee River, a blackwater river in Georgia, USA. Journal of the North American Benthological Society 16:82–87.

Meyer, G. A., S. G. Wells, R. C. Balling, and A. J. T. Jull. 1992. Response of alluvial systems to fire and climate change in Yellowstone National Park. Nature 357:147–150.

Miller, R. R. 2010. Is the past present? Historical splash dam mapping and stream disturbance detection in the Oregon Coastal Province. Thesis. Oregon State University, Corvallis, Oregon, USA.

Milliman, J., and R. Meade. 1983. World-wide delivery of river sediment to the oceans. Journal of Geology 91:1–21.

Montgomery, D. R. 1999. Process domains and the river continuum. Journal of the American Water Resources Association 35:397–410.

Montgomery, D. R., and J. M. Buffington. 1997. Channel reach morphology in mountain drainage basins. Geological Society of America Bulletin 109:596–611.

Moody, J. A., and D. A. Martin. 2001. Initial hydrologic and geomorphic response following a wildfire in the Colorado Front Range. Earth Surface Processes and Landforms 26:1049–1070.

Mulholland, P. J., and J. W. Elwood. 1982. The role of lake and reservoir sediments as sinks in the perturbed global carbon cycle. Tellus 34:490–499.

Naiman, R. J., C. A. Johnston, and J. C. Kelley. 1988. Alteration of North American streams by beaver. BioScience 38:753–762.

Naiman, R. J., J. M. Melillo, and J. E. Hobbie. 1986. Ecosystem alteration of boreal forest streams by beaver (*Castor canadensis*). Ecology 67:1254–1269.

Naiman, R. J., J. M. Melillo, M. A. Lock, T. E. Ford, and S. R. Reice. 1987. Longitudinal patterns of ecosystem processes and community structure in a subarctic river continuum. Ecology 68:1139–1156.

Nilsson, C., et al. 2005*a*. Forecasting environmental responses to restoration of rivers used as log floatways: an interdisciplinary challenge. Ecosystems 8:779–800.

Nilsson, C., C. A. Reidy, M. Dynesius, and C. Revenga. 2005*b*. Fragmentation and flow regulation of the world's large river systems. Science 308:405–408.

Noe, G. B., and C. R. Hupp. 2005. Carbon, nitrogen and phosphorus accumulation in floodplains of Atlantic Coastal Plain rivers, USA. Ecological Applications 15:1178–1190.

O'Connor, J. E., M. A. Jones, and T. L. Haluska. 2003. Flood plain and channel dynamics of the Quinault and Queets Rivers, Washington, USA. Geomorphology 51:31–59.

Ogston, L., S. Gidora, M. Foy, and J. Rosenfeld. 2015. Watershed-scale effectiveness of floodplain habitat restoration for juvenile coho salmon in the Chilliwack River, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 72:479–490.

Oguchi, T. 1997. Late Quaternary sediment budget in alluvial-fan-source-basin systems in Japan. Journal of Quaternary Science 12:381–390.

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

Opperman, J. J., R. Luster, B. A. McKenney, M. Roberts, and A. W. Meadows. 2010. Ecologically functional floodplains: connectivity, flow regime, and scale. Journal of the American Water Resources Association 46:211–226.

Pacheco, F. S., F. Roland, and J. A. Downing. 2013. Eutrophication reverses whole-lake carbon budgets. Inland Waters 4:41–48.

Palmer, M. A., et al. 2010. Mountaintop mining consequences. Science 327:148–149.

Pastor, J., R. T. Naiman, B. Dewey, and P. McInnes. 1988. Moose, microbes and the boreal forest. BioScience 38:770–777.

Pennino, M. J., S. S. Kaushal, J. J. Beaulieu, and P. M. Mayer. 2014. Effects of urban stream burial on nitrogen uptake and ecosystem metabolism: implications for watershed nitrogen and carbon fluxes. Biogeochemistry 121:107–126.

Pielke, R. A., et al. 2011. Land use/land cover changes and climate: modeling analysis and observational evidence. Wiley Interdisciplinary Reviews: Climate Change 2:828–850.

Pierce, J. L., G. A. Meyer, and A. J. T. Jull. 2004. Fire-induced erosion and millennial- scale climate change in northern ponderosa pine forests. Nature 432:87–90.

Quinton, J. N., G. Govers, K. Van Oost, and R. D. Bardgett. 2010. The impact of agricultural soil erosion on biogeochemical cycling. Nature Geoscience 3:311–314.

Ramankutty, N., and J. A. Foley. 1999. Estimating historical changes in global land cover: croplands from 1700 to 1992. Global Biogeochemical Cycles 13:997–1027.

Ramos Scharrón, C. E., C. J. Castellanos, and C. Restrepo. 2012. The transfer of modern organic carbon by landslide activity in tropical montane ecosystems. Journal of Geophysical Research—Biogeosciences 117:G03016.

Rasera, M. F. F. L., A. V. Krusche, J. E. Richey, M. V. R. Ballester, and R. L. Victória. 2013. Spatial and temporal variability of pCO_2 and CO_2 efflux in seven Amazonian Rivers. Biogeochemistry 116:241–259.

Rathbun, S., G. Bennett, E. Wohl, C. Briles, B. McElroy, and N. Sutfin. *In press*. The fate of sediment, wood, and organic carbon eroded during an extreme flood, Colorado Front Range, USA. Geology. https://doi.org/10.1130/g38935.1

Raymond, P. A., and J. E. Bauer. 2001. Use of ^{14}C and ^{13}C natural abundances for evaluating riverine, estuarine, and coastal DOC and POC sources and cycling: a review and synthesis. Organic Geochemistry 32:469–485.

Raymond, P. A., N. H. Oh, R. E. Turner, and W. Broussard. 2008. Anthropogenically enhanced fluxes of water and carbon from the Mississippi River. Nature 451:449–452.

Raymond, P. A., et al. 2013. Global carbon dioxide emissions from inland waters. Nature 503:355–359.

Reeder, J. D., G. E. Schuman, J. A. Morgan, and D. R. Lecain. 2004. Response of organic and inorganic carbon and nitrogen to long-term grazing of the shortgrass steppe. Environmental Management 33:485–495.

Regnier, P., et al. 2013. Anthropogenic perturbation of the carbon fluxes from land to ocean. Nature Geoscience 6:597–607.

Richey, J. E. 2004. Pathways of atmospheric CO_2 through fluvial systems. Pages 329–340 in C. B. Field, and M. R. Raupach, editors. The global carbon cycle, integrating humans, climate, and the natural world. Volume 17. Island Press, Washington, DC.

Ricker, M. C., S. W. Donohue, M. H. Stolt, and M. S. Zavada. 2012. Development and application of multi-proxy indices of land use change for riparian soils in southern New England, USA. Ecological Applications 22:487–501.

Ricker, M. C., and B. G. Lockaby. 2015. Soil organic carbon stocks in a large eutrophic floodplain forest of the southeastern Atlantic Coastal Plain, USA. Wetlands 35:291–301.

Ricker, M. C., M. H. Stolt, S. W. Donohue, G. A. Blazejewski, and M. S. Zavada. 2013. Soil organic carbon pools in riparian landscapes of southern New England. Soil Science Society of America Journal 77:1070–1079.

Ricker, M. C., M. H. Stolt, and M. S. Zavada. 2014. Comparison of soil organic carbon dynamics in forested riparian wetlands and adjacent uplands. Soil Science Society of America Journal 78:1817–1827.

Rieger, I., F. Lang, I. Kowarik, and A. Cierjacks. 2014. The interplay of sedimentation and carbon accretion in riparian forests. Geomorphology 214:157–167.

Roberts, M. L., and R. E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. Journal of the North American Benthological Society 28:941–954.

CONCEPTS & SYNTHESIS

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

Rockström, J., et al. 2009. A safe operating space for humanity. Nature 461:472–475.

Rosemond, A. D., J. P. Benstead, P. M. Bumpers, V. Gulis, J. S. Kominoski, D. W. P. Manning, K. Suberkropp, and J. B. Wallace. 2015. Experimental nutrient additions accelerate terrestrial carbon loss from stream ecosystems. Science 347:1142–1145.

Ross, M. R., B. L. McGlynn, and E. S. Bernhardt. 2016. Deep impact: effects of mountaintop mining on surface topography, bedrock structure, and downstream waters. Environmental Science and Technology 50:2064–2074.

Roy, A. H., A. L. Dybas, K. M. Fritz, and H. R. Lubbers. 2009. Urbanization affects the extent and hydrologic permanence of headwater streams in a midwestern US metropolitan area. Journal of the North American Benthological Society 28:911–928.

Ruesch, A., and H. K. Gibbs. 2008. New IPCC Tier-1 global biomass carbon map for the year 2000. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, Oak Ridge, Tennessee, USA.http://cdiac.ornl.gov

Ruffing, C. M., M. D. Daniels, and K. A. Dwire. 2015. Disturbance legacies of historic tie drives persistently alter geomorphology and large wood characteristics in headwater streams, southeast Wyoming. Geomorphology 231:1–14.

Sabo, J. L., et al. 2010. Reclaiming freshwater sustainability in the Cadillac Desert. Proceedings of the National Academy of Sciences USA 107:21263–21270.

Samaritani, E., J. Shrestha, B. Fournier, E. Frossard, F. Gillet, C. Guenat, P. A. Niklaus, K. Tockner, E. A. D. Mitchell, and J. Luster. 2011. Heterogeneity of soil carbon pools and fluxes in a channelized and a restored floodplain section (Thur River, Switzerland). Hydrology and Earth System Sciences Discussions 8:1059–1091.

Sampere, T. P., T. S. Bianchi, and M. A. Allison. 2011. Historical changes in terrestrially derived organic carbon inputs to Louisiana continental margin sediments over the past 150 years. Journal of Geophysical Research 116:G01016.

Sanders, L. M., K. H. Taffs, D. J. Stokes, C. J. Sanders, J. M. Smoak, A. Enrich-Prast, P. A. Macklin, I. R. Santos, and H. Marotta. 2017. Carbon accumulation in Amazonian floodplain lakes: A significant component of Amazon budgets? Limnology and Oceanography 2:29–35.

Sarmiento, J. L., and F. T. Sundquist. 1992. Revised budget of the oceanic uptake of anthropogenic carbon dioxide. Nature 356:589–593.

Sawyer, A. H., M. B. Cardenas, and J. Buttles. 2011. Hyporheic exchange due to channel-spanning logs. Water Resources Research 47:W08502.

Schama, S. 1996. Landscape and memory. Vintage, Visalia, CA.

Scharnweber, K., M. J. Vanni, J. Syvaranta, and T. Mehner. 2014. Boomerang ecosystem fluxes: organic carbon inputs from land to lake are returned to terrestrial food webs via aquatic insects. Oikos 123:1439–1448.

Schindler, D. E., S. R. Carpenter, J. J. Cole, J. F. Kitchell, and M. L. Pace. 1997. Influence of food web structure on carbon exchange between lakes and the atmosphere. Science 277:248–251.

Schlesinger, W. H., and J. M. Melack. 1981. Transport of organic carbon in the world's rivers. Tellus 33:172–187.

Schmitz, O. J., E. Post, C. E. Burns, and K. M. Johnston. 2003. Ecosystem responses to global climate change: moving beyond color-mapping. BioScience 53:1199–2005.

Schmitz, O. J., et al. 2014. Animating the carbon cycle. Ecosystems 17:344–359.

Schumm, S. A. 1977. The fluvial system. WH Freeman and Co., San Francisco, California, USA.

Schuur, E. A. G., et al. 2015. Climate change and the permafrost carbon feedback. Nature 520:171–179.

Sear, D. A., C. E. Millington, D. R. Kitts, and R. Jeffries. 2010. Logjam controls on channel:floodplain interactions in wooded catchments and their role in the formation of multi-channel patterns. Geomorphology 116:305–319.

Sedell, J. R., J. E. Richey, and F. J. Swanson. 1989. The river continuum concept: a basis for the expected ecosystem behavior of very large rivers. Pages 49–55 in D. P. Dodge, editor. Proceedings of the International Large River Symposium. Canadian Special Publications of Fisheries and Aquatic Sciences 106, Ottawa, Canada.

Shakesby, R. A., and S. H. Doerr. 2006. Wildfire as a hydrological and geomorphological agent. Earth-Science Reviews 74:269–307.

Shields, F. D., and R. H. Smith. 1992. Effects of large woody debris removal on physical characteristics of a sand-bed river. Aquatic Conservation: Marine and Freshwater Ecosystems 2:145–163.

Sickman, J. O., M. J. Zanoli, and H. L. Mann. 2007. Effects of urbanization on organic carbon loads in the Sacramento River, California. Water Resources Research 43:W11422.

Smith, J. C., A. Galy, N. Hovius, A. M. Tye, J. M. Turowski, and P. Schleppi. 2013. Runoff-driven export of particulate organic carbon from soil in temperate forested uplands. Earth and Planetary Science Letters 365:198–208.

Smith, S. V., W. H. Renwick, R. W. Buddemeier, and C. J. Crossland. 2001. Budgets of soil erosion and deposition for sediments and sedimentary organic carbon across the conterminous United States. Global Biogeochemical Cycles 15:697–707.

Smittenberg, R. H., T. I. Eglinton, S. Schouten, and J. S. Sinninghe Damste. 2006. Ongoing buildup of refractory organic carbon in boreal soils during the Holocene. Science 314:1283–1286.

Spencer, R. G. M., P. J. Mann, T. Dittmar, T. I. Eglinton, C. McIntyre, R. M. Holmes, N. Zimov, and A. Stubbins. 2015. Detecting the signature of permafrost thaw in Arctic rivers. Geophysical Research Letters 42:2830–2835.

St. Louis, V. L., C. A. Kelly, E. Duchemin, J. W. M. Rudd, and D. M. Rosenberg. 2000. Reservoir surfaces as sources of greenhouses gases to the atmosphere: a global estimate. BioScience 50:766–775.

Stackpoole, S. M., et al. 2016. Spatial and temporal patterns of dissolved organic matter quantity and quality in the Mississippi River Basin, 1997–2013. Hydrological Processes 2016:1–14.

Steffen, W., J. Grinevald, P. Crutzen, and J. McNeill. 2011. The Anthropocene: conceptual and historical perspectives. Philosophical Transactions of the Royal Society A 369:842–867.

Strahler, A. N. 1952. Hypsometric (area-altitude) analysis of erosional topography. Geological Society of America Bulletin 63:1117–1142.

Sutfin, N. A. 2015. Spatiotemporal variability of floodplain sediment and organic carbon retention in mountain streams of the Colorado Front Range. Dissertation. Colorado State University, Fort Collins, Colorado, USA.

Syvitski, J. P. M., and A. J. Kettner. 2011. Sediment flux and the Anthropocene. Philosophical Transactions of the Royal Society A 369:957–975.

Syvitski, J. P. M., C. J. Vorosmarty, A. J. Kettner, and P. Green. 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. Science 308:376–380.

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

CONCEPTS & SYNTHESIS

Syvitski, J. P. M., et al. 2009. Sinking deltas due to human activities. Nature Geoscience 2:681–686.

Tarnocai, C., J. G. Canadell, E. A. G. Schuur, P. Kuhry, G. Mazhitova, and S. Zimov. 2009. Soil organic carbon pools in the northern circumpolar permafrost region. Global Biogeochemical Cycles 23:GB2023.

Taylor, B. W., A. S. Flecker, and R. O. Hall. 2006. Loss of a harvested fish species disrupts carbon flow in a diverse tropical river. Science 313:833–836.

Thorp, J. H., and M. D. Delong. 1994. The riverine productivity model: an heuristic view of carbon sources and organic processing in large river ecosystems. Oikos 70:305–308.

Thorp, J. H., M. C. Thoms, and M. D. Delong. 2006. The riverine ecosystem synthesis: biocomplexity in river networks across space and time. River Research and Applications 22:123–147.

Tranvik, L. J., et al. 2009. Lakes and reservoirs as regulators of carbon cycling and climate. Limnology and Oceanography 54:2298–2314.

Trimble, S. W., and A. C. Mendel. 1995. The cow as a geomorphic agent – a critical review. Geomorphology 13:233–253.

Trumbore, S. E. 1993. Comparison of carbon dynamics in tropical and temperate soils using radiocarbon measurements. Global Biogeochemical Cycles 7:275–290.

Trumbore, S. E., and C. I. Czimczik. 2008. An uncertain future for soil carbon. Science 321:1455–1456.

Ulseth, A. J., and R. O. Hall Jr. 2015. Dam tailwaters compound the effects of reservoirs in the longitudinal transport of organic carbon in an arid river. Biogeosciences 12:4345–4359.

USCCSP (United States Carbon Cycle Science Program). 2016. State of the Carbon Cycle Report-2 (SOCCR-2). A special scientific assessment of the state of the carbon cycle in the United States and surrounding North American region—a draft prospectus. https://carboncyclescience.us/sites/default/files/cciwg/SOCCR-2

Van der Gaast, W., R. Sikkema, and M. Vohrer. 2016. The contribution of forest carbon credit projects to addressing the climate change challenge. Climate Policy. https://doi.org/10.1080/14693062.2016.1242056

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130–137.

Van Oost, K., G. Verstraeten, S. Doetterl, B. Notebaert, F. Wiaux, N. Broothaerts, and J. Six. 2012. Legacy of human-induced C erosion and burial on soil-atmosphere C exchange. Proceedings of the National Academy of Sciences 109:19492–19497.

Verpoorter, C., T. Kutser, D. A. Seekel, and L. J. Tranvik. 2014. A global inventory of lakes based on high-resolution satellite imagery. Geophysical Research Letters 41:6396–6402.

Wackernagel, M., J. Kitzes, D. Moran, S. Goldfinger, and M. Thomas. 2006. The ecological footprint of cities and regions: comparing resource availability with resource demand. Environment and Urbanzation 18:103–112.

Walter, R. C., and D. J. Merritts. 2008. Natural streams and the legacy of water-powered mills. Science 319:299–304.

Ward, J. V. 1989. The four-dimensional nature of lotic ecosystems. Journal of the North American Benthological Society 8:2–8.

Ward, G. M., and N. G. Aumen. 1986. Woody debris as a source of fine particulate organic matter in coniferous forest stream ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 43:1635–1642.

Ward, J. V., and J. A. Stanford. 1983. The serial discontinuity concept of lotic ecosystems. Pages 29–42 in T. D. Fontaine, and S. M. Bartell, editors. Dynamics of lotic ecosystems. Ann Arbor Science, Ann Arbor, Michigan, USA.

Ward, J. V., and K. Tockner. 2001. Biodiversity: towards a unifying them for river ecology. Freshwater Biology 46:807–819.

Warton, D. I., I. J. Wright, D. S. Falster, and M. Westoby. 2006. Bivariate line-fitting methods for allometry. Biological Reviews 81:259–291.

West, A. J., C.-W. Lin, T.-C. Lin, R. G. Hilton, S.-H. Liu, C.-T. Chang, K.-C. Lin, A. Galy, R. B. Sparkes, and N. Hovius. 2011. Mobilization and transport of coarse woody debris to the oceans triggered by an extreme tropical storm. Limnology and Oceanography 56:77–85.

Wilcox, A. C., E. E. Wohl, F. Comiti, and L. Mao. 2011. Hydraulics, morphology, and energy dissipation in an alpine step-pool channel. Water Resources Research 47:W07514.

Wohl, E. 2011. Threshold-induced complex behavior of wood in mountain streams. Geology 39:587–590.

Wohl, E. 2013. Landscape-scale carbon storage associated with beaver dams. Geophysical Research Letters 40:1–6.

Wohl, E. 2014. A legacy of absence: wood removal in US rivers. Progress in Physical Geography 38:637–663.

Wohl, E. 2015. Particle dynamics: the continuum of bedrock to alluvial river segments. Geomorphology 241:192–208.

Wohl, E., et al. 2012a. The hydrology of the humid tropics. Nature Climate Change 2:655–662.

Wohl, E., B. P. Bledsoe, R. B. Jacobson, N. L. Poff, S. L. Rathburn, D. M. Walters, and A. C. Wilcox. 2015. The natural sediment regime: broadening the foundation for ecosystem management. BioScience 65:358–371.

Wohl, E., K. Dwire, N. Sutfin, L. Polvi, and R. Bazan. 2012b. Mechanisms of carbon storage in mountainous headwater rivers. Nature Communications 3:1263.

Wohl, E., and F. L. Ogden. 2013. Organic carbon export in the form of wood during an extreme tropical storm, Upper Rio Chagres, Panama. Earth Surface Processes and Landforms 38:1407–1416.

Wolfe, B. E., and J. N. Klironomos. 2005. Breaking new ground: soil communities and exotic plant invasion. BioScience 55:477–487.

Worrall, F., and T. Burt. 2004. Time series analysis of long-term river dissolved organic carbon records. Hydrological Processes 18:893–911.

Young, M. K., D. Haire, and M. A. Bozek. 1994. The effect and extent of railroad tie drives in streams of southeastern Wyoming. Western Journal of Applied Forestry 9:125–130.

Yue, Y., J. Ni, A. G. L. Borthwick, and C. Miao. 2012. Diagnosis of river basins as CO_2 sources or sinks subject to sediment movement. Earth Surface Processes and Landforms 37:1398–1406.

Zhang, S., X. X. Lu, H. Sun, J. Han, and D. L. Higgitt. 2009. Geochemical characteristics and fluxes of organic carbon in a human-disturbed mountainous river (the Luodingjiang River) of the Zhujiang (Pearl River), China. Science of the Total Environment 407:815–825.

Zhang, L., and W. J. Mitsch. 2007. Sediment chemistry and nutrient influx in a hydrologically restored bottomland hardwood forest in Midwestern USA. River Research and Applications 23:1026–1037.

CONCEPTS & SYNTHESIS

SUPPORTING INFORMATION

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecm.1261/full