Received: 20 January 2022 | Revised: 5 May 2022 | Accepted: 15 June 2022

DOI: 10.1111/gcb.16324

Global Change Biology WILEY

REVIEW

Anthropogenically driven climate and landscape change effects on inland water carbon dynamics: What have we learned and where are we going?

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Funding information
Oak Ridge National Laboratory; Office of Energy Efficiency and Renewable Energy; U.S. Department of Energy; Water Power Technologies Office

Abstract

Inland waters serve as important hydrological connections between the terrestrial landscape and oceans but are often overlooked in global carbon (C) budgets and Earth System Models. Terrestrially derived C entering inland waters from the watershed can be transported to oceans but over 83% is either buried in sediments or emitted to the atmosphere before reaching oceans. Anthropogenic pressures such as climate and landscape changes are altering the magnitude of these C fluxes in inland waters. Here, we synthesize the most recent estimates of C fluxes and the differential contributions across inland waterbody types (rivers, streams, lakes, reservoirs, and ponds), including recent measurements that incorporate improved sampling methods, small waterbodies, and dried areas. Across all inland waters, we report a global C emission estimate of 4.40 Pg C/year (95% confidence interval: 3.95–4.85 Pg C/year), representing a 13% increase from the most recent estimate. We also review the mechanisms by which the most globally widespread anthropogenically driven climate and landscape changes influence inland water C fluxes. The majority of these drivers are expected to influence terrestrial C inputs to inland waters due to alterations in terrestrial C quality and quantity, hydrological pathways, and biogeochemical processing. We recommend four research priorities for the future study of anthropogenic alterations to inland water C fluxes: (1) before-and-after measurements of C fluxes associated with climate change events and landscape changes, (2) better quantification of C input from land, (3) improved assessment of spatial coverage and contributions of small inland waterbodies to C fluxes, and (4) integration of dried and drawdown areas to global C flux estimates. Improved measurements of inland water C fluxes and quantification of uncertainty in these estimates will be vital to understanding both terrestrial C losses and the "moving target" of inland water C emissions in response to rapid and complex anthropogenic pressures.

KEYWORDS
anthropogenic change, carbon emissions, carbon fluxes, climate change, environmental effects, freshwater ecosystems, terrestrial carbon, terrestrial–aquatic interface

1 | INTRODUCTION

The role of inland waters as active, rather than passive, pipes in the global carbon (C) cycle has been well established for over 15 years (Cole et al., 2007). While previous global C budgets assumed that inland waters exported the same amount of C to oceans that was input into them with no internal C processing (Denman et al., 2007), internal C processes such as burial in and emissions from inland waters are now more realistically included in global C budgets (Ciais et al., 2013). Research on C fluxes in inland waters has emphasized the increasing magnitude of global C emissions due to improved measurements and spatiotemporal coverage, and as a response to anthropogenic change (Drake et al., 2018; Regnier et al., 2013). From this body of research, regional hotspots (Borges et al., 2015; Raymond et al., 2013; Sawakuchi et al., 2017) and temporal hot moments of C emissions from inland waters (Beaulieu et al., 2014; Demarty et al., 2011; Denfeld et al., 2018; Schilder et al., 2016; Vachon et al., 2017) have been highlighted globally. Some heterogeneity in C fluxes is due to the diverse set of waterbodies that inland waters encapsulate, including rivers, streams, lakes, reservoirs, and ponds. These waterbody types differentially contribute to total inland water C fluxes due to their diverse morphological, physical, chemical, and biological characteristics. There have been foundational efforts to estimate global C budgets across all inland waters (Battin et al., 2009; Cole et al., 2007; Drake et al., 2018; Tranvik et al., 2009) or to compare estimates of a subset of C fluxes (i.e., CO_2 and CH_4 emissions) in various inland waterbody types (Bastviken et al., 2011; Deemer et al., 2016; DelSontro et al., 2018; Mendonça et al., 2017; Raymond et al., 2013; Rosentreter et al., 2021; Tranvik et al., 2009). However, we currently lack a synthesis of all major C fluxes and the contribution by each waterbody type that highlights the mechanisms for their differential role to global inland water C fluxes and the effects of continued anthropogenic changes that alter C transport to and processing in inland waters.

Anthropogenically driven climate change and modification of the landscape are occurring globally at unprecedented scales. The impacts of climate change have affected billions of people and up to 80% of global land area (Callaghan et al., 2021), ranging from climate warming to sea level rise to increasingly frequent and more severe natural disasters (IPCC, 2013). These anthropogenic pressures influence terrestrial and inland water ecosystems by altering biogeochemical processing and hydrological connectivity via a variety of mechanisms that we detail in this review. In addition to climate change, anthropogenic landscape and land-use changes alter terrestrial C quantity and quality, and hydrologic pathways and connectivity, influencing C transport between land and water and the cycling of C within inland waters. Landscape changes (i.e., conversion from natural ecosystems to human-made or modified areas) have affected 32% of global land area since 1960 (Winkler et al., 2021). The large reductions in C-storing natural forest and wetland ecosystems (Davidson, 2014; Meiyappan & Jain, 2012) suggest major increases in lateral C fluxes from land to receiving inland waterbodies, which, in turn, have a high likelihood of being emitted from inland waters (Drake et al., 2018). The array of anthropogenically driven climate and landscape changes that affect different C fluxes in inland waters is vast, with the potential for significant interactions that can play an important role in current and future global C flux estimates.

For this review, we have three objectives. First, we synthesize the most recent global estimates of C fluxes in inland waters and the contributions of different inland waterbody types to global C fluxes, noting the disproportionate contributions of C emissions and C burial compared to relative surface area. Second, we review key anthropogenically induced climate and landscape changes that alter C transport and processing and that contribute to continued alterations in C fluxes in inland waters, highlighting patterns that may affect C fluxes for specific inland waterbody types. Third, we discuss important research priorities and the uncertainty in quantifying C fluxes in inland waters as a "moving target" due to global anthropogenic pressures. Finally, we conclude with a future outlook of quantifying C fluxes and estimating uncertainty in these dynamically altered inland waters.

2 | GLOBAL C FLUXES IN INLAND WATERS

2.1 | Overview of C flux components

C fluxes in inland waters include terrestrial inputs, aquatic primary production, burial, emissions, and export to oceans (see definitions in Table 1; Figure 1), with a simple mass balance stating that terrestrial C inputs plus aquatic C production equals the sum of C burial, emissions, and export in inland waters (Butman et al., 2018). The primary C input to inland waters comes from land and encompasses organic and inorganic C (OC and IC, respectively; Table 1) of terrestrial origin (soil, vegetation, respiration products) that enter via wind, surface and subsurface runoff, and groundwater. The secondary source of C is aquatic primary production via photosynthetic CO_2 fixation. This input of C is not well quantified at the global scale and is generally assumed to be low (i.e., 0.3 Pg C/year) (Cole et al., 2007; Drake et al., 2018; Regnier et al., 2013), and as such we do not discuss its response to anthropogenic climate and landscape change in detail. Once within inland waters, C has three fates: burial (storage), emission,

TABLE 1 Terms and specific definitions used in this paper related to inland water C processing and fluxes. When using "C," we are referring to all relevant forms of C for the specific flux or pathway being discussed (i.e., "C emissions" refers jointly to CO_2 emissions and CH_4 emissions), and clarify specific forms of C when reviewing specific pathways or examples

Term	Definition
Forms of C	
CO_2	Carbon dioxide, which is the primary form of dissolved inorganic carbon and one form of C emitted from inland waters
CH_4	Methane, which has a 34× greater global warming potential than CO_2, and one form of C emitted from inland waters
IC	Inorganic carbon, dominantly in dissolved (DIC) form as CO_2 in inland waters
OC	Organic carbon, which can be in dissolved (DOC, typically <0.7 μm) or particulate (POC, typically >0.7 μm) forms
C fluxes	
Input	Organic and inorganic C of terrestrial origin transferred laterally from land to inland waters via wind, surface and subsurface runoff, or groundwater
Aquatic primary production	Fixation of atmospheric CO_2 via photosynthesis by aquatic primary producers
Burial	Storage or sequestration of C (primarily organic, but can be inorganic) in sediments and removal of C from the active C cycle, at decadal to millennial time scales
Emissions (also evasion, outgassing)	Release of CO_2 and CH_4 from inland waters to the atmosphere via diffusion, ebullition, or degassing pathways ("C emissions" refers to CO_2 plus CH_4 emissions)
Export	Organic and inorganic C delivered from inland waters downstream to oceans
Additional terms	
C quality	Chemical composition of OC in inland waters that influences properties such as chromophoricity (color) and lability
Degassing	Pathway of CO_2 and CH_4 emissions via turbulent diffusion such as during water release through turbines or spillways, and with irrigation
Diffusion	Pathway of CO_2 and CH_4 emissions at the air–water interface following a chemical gradient
Ebullition	Pathway of CH_4 emissions via bubbles produced in sediments, transported through water column, and released to the atmosphere
Sedimentation	Decomposition or breakdown of OC followed by settling and deposition

or export to oceans (Table 1). C export to oceans was previously considered the sole C flux from inland waters in the outdated "passive pipe" model (Cole et al., 2007). While up to 1.06 Pg C/year is delivered to oceans from inland waters (Li et al., 2017), inland waters process significantly more C as "active pipes" (Cole et al., 2007; Drake et al., 2018). Burial in sediments sequesters C for long periods, preventing its return to the atmosphere or export to oceans for decades to millennia (Cole et al., 2007; Mendonça et al., 2017). C emissions from inland waters include evasion of CO_2 and CH_4 via diffusion, ebullition, and degassing (Table 1). Accurately quantifying these C fluxes is important for understanding changes in C emissions and estimating terrestrial C losses, but can be difficult due to the heterogeneous contributions of different inland waterbody types plus widespread anthropogenic pressures influencing these C fluxes (Butman et al., 2018).

2.2 | Influence of inland waterbody types on relative C fluxes

Though inland waters cover only a small portion of the global land area (~4%), they play a large role in global C processing (Cole et al., 2007). At a global scale, lentic waters dominate the estimated area of inland waters (90%; Figure 2), with natural lakes alone accounting for approximately 5 million km^2 (Verpoorter et al., 2014). Reservoir surface area is estimated at 305,723 km^2 (Deemer et al., 2016), though a "global boom" of planned dam constructions (Zarfl et al., 2015) could increase this estimate substantially. Pond area, which is highly uncertain due to the small size and intermittency of ponds, is estimated between 148,000 and 862,000 km^2 (Holgerson & Raymond, 2016). In contrast, lotic waters, including rivers and streams, encompass about 10% of the world's inland water area, covering approximately 662,100 km^2

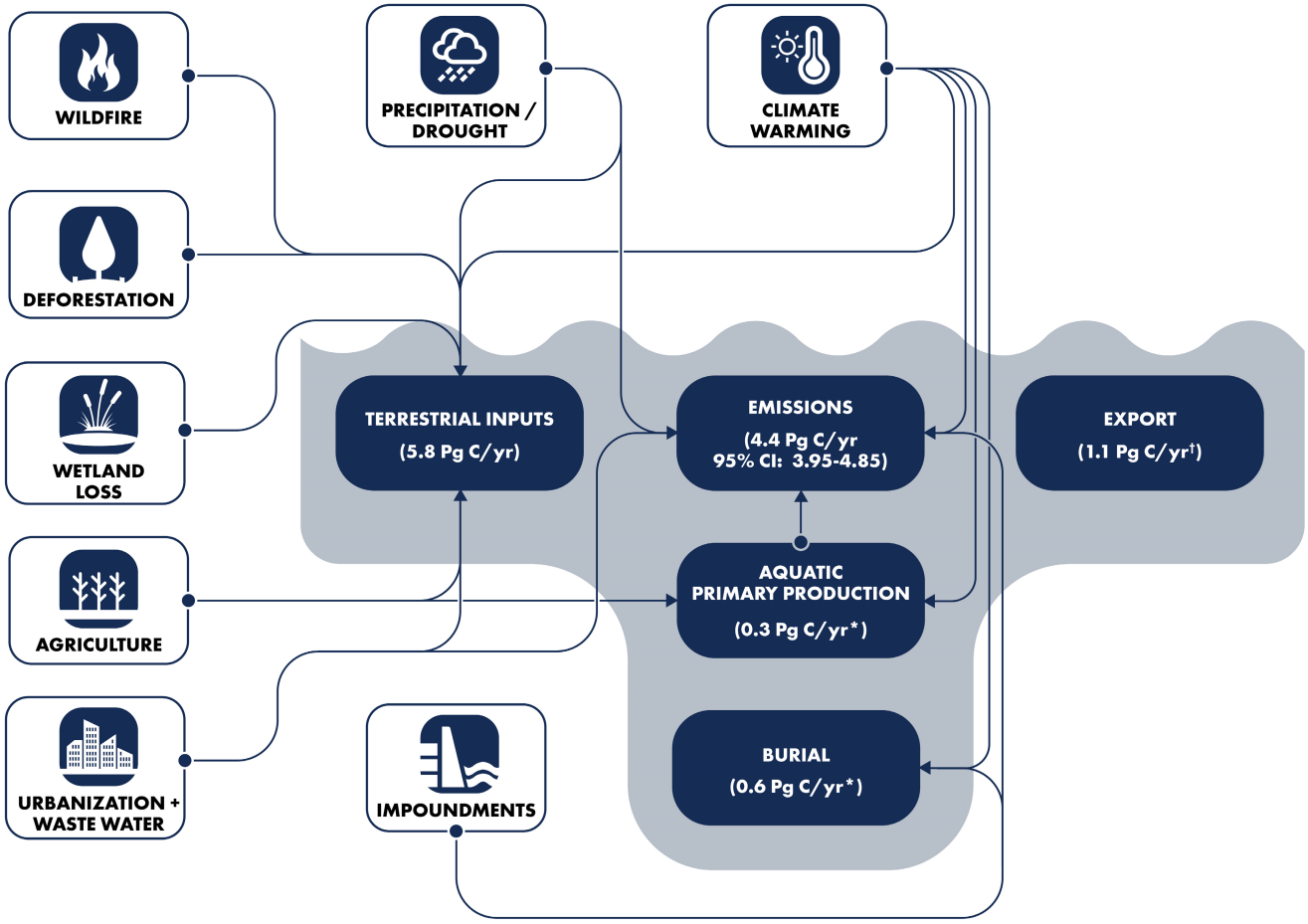

FIGURE 1 Conceptual diagram linking the described climate and landscape changes to the respective altered C fluxes in inland waters as described in detail in Section 3 and Table 3. Grey area represents inland waters as a whole, including rivers, streams, lakes, reservoirs, and ponds. Estimates of total C inputs from land and C emissions from inland waters are calculated from recently published global studies (see Table 2). Estimates for total aquatic primary production and burial (*) in inland waters are from Regnier et al. (2013); these estimates in particular are not well quantified in inland waters at a global scale nor based on waterbody type. Estimate for export (†) is from Li et al. (2017), assumed to be exclusively from streams and rivers. Climate-related changes in precipitation and temperature will likely interact with many landscape change drivers and disturbances in influencing C fluxes in inland waters at short (i.e., drought increases likelihood of wildfire) to longer time frames (i.e., warming alters vegetation and related C quality and storage in watersheds), which have the potential to be vast and therefore are not detailed in this diagram for clarity (see Box 2).

(Downing et al., 2012). However, global river length is 21 times longer than the cumulative perimeter of world lakes and 46 times the length of world coastlines (Downing et al., 2012), highlighting the considerable role that lotic systems play in interactions at the terrestrial–aquatic interface. The differences in morphological, physical, chemical, and biological properties across these inland waterbody types result in disproportionate contributions to inland water C fluxes compared to their relative surface area coverage (Table 2, Figure 2).

Globally, lotic waters contribute the vast majority of CO_2 emissions from inland waters (Deemer et al., 2016; Raymond et al., 2013)—more than five times all lentic systems combined (Table 2, Figure 2). Lotic waters are nearly always supersaturated with CO_2 (dissolved inorganic carbon, DIC; Table 1) (Lauerwald et al., 2015; Raymond et al., 2013), which can be especially pronounced in lower order streams (Davidson et al., 2010; Teodoru et al., 2009). The dominant source of DIC in most undisturbed streams and rivers is from the terrestrial watershed (Hotchkiss et al., 2015; Marx et al., 2017) via weathering of bedrock and soil respiration, which enters via groundwater, subsurface, or overland flowpaths (Regnier et al., 2013). The high turbulence and gas transfer velocity in lotic waters allows supersaturated CO_2 to be readily degassed into the atmosphere, resulting in lowered DIC downstream of source water inputs (Doctor et al., 2008). Despite generally having higher CO_2 emissions than larger lentic waters (Hotchkiss et al., 2015), small streams are more difficult to integrate into global estimates of inland water C fluxes given limited detection by satellite imagery at fine resolution (Raymond et al., 2013).

While CO_2 (as C) constitutes over 98% of total C emitted from inland waters, the high 100-year global warming potential of CH_4 (factor of 34) (Myhre et al., 2013) suggests that CH_4 comprises over 15% of CO_2-equivalent emissions from inland waters (Table 2). Unlike

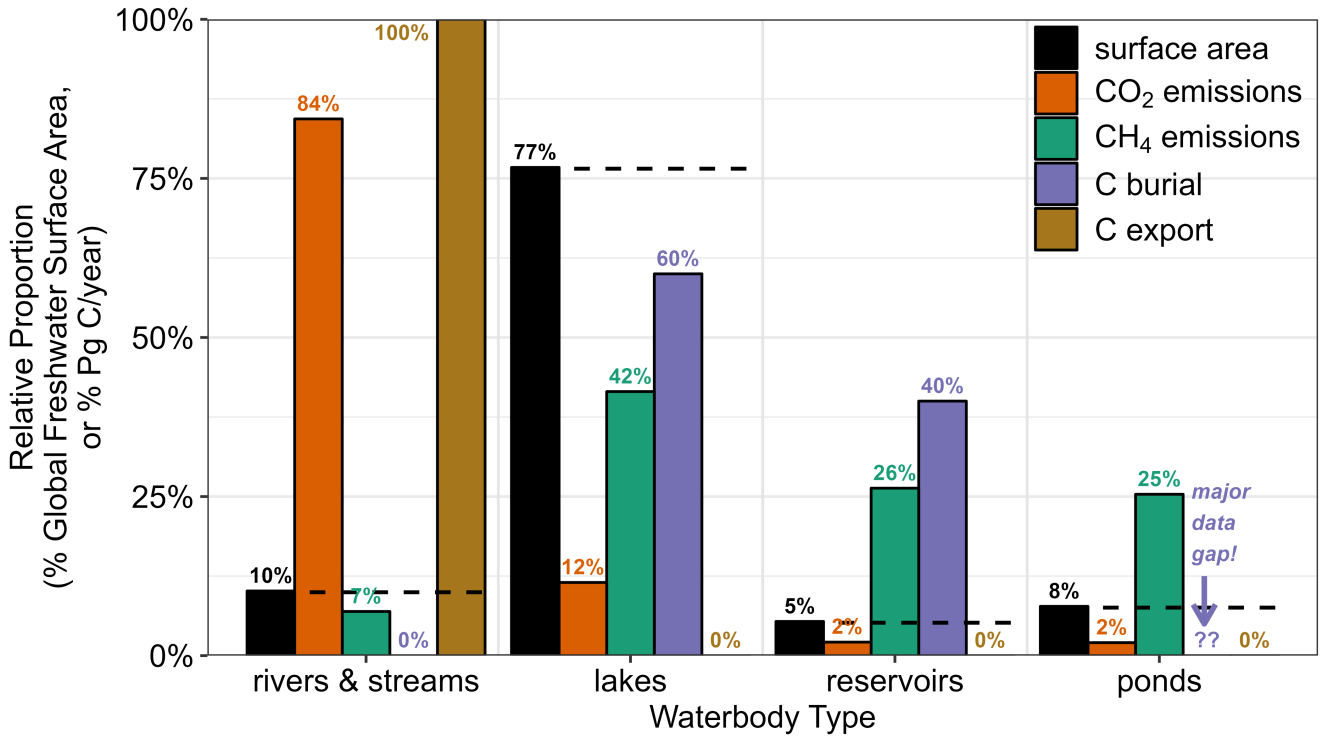

FIGURE 2 Relative proportion of estimated global surface area (black) by waterbody type, and relative proportion of estimated global C fluxes by waterbody type, including CO_2 emissions (orange), CH_4 emissions (diffusive + ebullitive + degassing [for reservoirs]; green), C burial (blue), and C export to oceans (brown). Estimates are from the most recent publications of global C flux by specific waterbody type that are listed in bold in Table 2. Dashed black lines indicate the expected C flux for each waterbody type if the flux was directly proportional to global surface area. Note that no global quantitative estimate of C burial in ponds has been conducted.

CO_2, most CH_4 in lentic waters is produced in-situ via anaerobic methanogenesis. Lentic systems have longer hydrologic residence times and greater potential for anoxic conditions that promote methanogenesis, making them large contributors of inland water CH_4 emissions (Figure 2) (Bastviken et al., 2011; Deemer et al., 2016; Rosentreter et al., 2021). Though natural lakes dominate the inland water surface area coverage, reservoirs and ponds contribute disproportionally to CH_4 emissions by five and three times greater than what would be expected given their relative surface areas, respectively (Figure 2), which is also reflected in the high contribution of CH_4 emissions (as CO_2-equivalents) relative to CO_2 in these water bodies (Figure 3). Reservoirs tend to have high primary productivity given their generally large watersheds often in agricultural areas (Knoll et al., 2003). This high productivity is associated with both a greater amount of autochthonous C preferred for CH_4 production (West et al., 2012) and low oxygen conditions following decomposition that promote CH_4 production. Research focused on CH_4 emissions from tropical reservoirs may bias these patterns as CH_4 emissions generally increase at lower latitudes (Barros et al., 2011), though more recent global syntheses with a similar spatial sampling bias reported no statistical difference between CH_4 emissions from tropical/subtropical vs. temperate reservoirs (Deemer et al., 2016; Johnson et al., 2021). One unique pathway by which CO_2 and CH_4 are emitted from many reservoirs is degassing (i.e., via turbines or spillways), which can account for over 50% of CH_4 emissions from reservoirs globally (Figure 3) (Harrison et al., 2021). In pond systems, frequent mixing of anoxic waters can result in large anoxic volumes that promote methanogenesis and CH_4 release (Holgerson & Raymond, 2016). CH_4 ebullitive emissions from lentic waters are high in shallow areas in particular (Bastviken et al., 2004; West et al., 2016), especially in ponds (Holgerson & Raymond, 2016), due to less time for CH_4 oxidation and lower hydrostatic pressure allowing for greater bubble release (ebullitive emissions) (Bastviken et al., 2008; Holgerson & Raymond, 2016; Ostrovsky et al., 2008). Lotic waters are often supersaturated with CH_4 but have generally low CH_4 emissions compared to CO_2 emissions (Figure 3) (Bastviken et al., 2011; Rosentreter et al., 2021; Stanley et al., 2016), and contribute only a fraction of inland water CH_4 emissions compared to lentic systems (Table 2, Figure 2).

Stagnant waters with stratified environments and low oxygen in deep waters promote higher OC burial rates and efficiency compared to flowing waters (Table 2) (Cole et al., 2007; Sobek et al., 2009). OC burial rates are amplified in reservoirs in particular, which have areal burial rates up to 6.5 times greater than natural lakes (Mendonça et al., 2017). Often, reservoirs receive large nutrient and C inputs associated with sediment from erosion (Downing et al., 2008; Quinton et al., 2010; Van Oost et al., 2007). These inputs stimulate primary production which can be directly buried (Mendonça et al., 2017) or indirectly lead to anoxia after decomposition, an environment that promotes OC burial over mineralization (Sobek et al., 2009). Ponds,

whether natural or manmade, are expected to have high burial rates given their high OC accumulation relative to their small size and depth (Downing et al., 2008; Mulholland & Elwood, 1982; Taylor et al., 2019), though no global estimate of total OC burial in ponds currently exists due to the lack of data on pond global areal coverage and scarce burial rates measured in-situ (Mendonça et al., 2017) (Figure 2). Small agricultural impoundments in particular have been shown to have some of the highest OC burial rates due to very high C inputs relative to water volume plus high primary production (Downing et al., 2008). However, dredging of reservoirs can remobilize OC and limit the time scale at which it is buried in sediment, and re-introduce it into the active C pool (Maeck et al., 2013).

Finally, effectively all C exported from inland waters to oceans is via lotic systems (Table 2, Figure 2) (Cole et al., 2007). C export is estimated to have increased by 20% since 1750 due to anthropogenic pressures (Regnier et al., 2013). The anthropogenically driven climate and landscape changes that impact C fluxes in inland waters (Regnier et al., 2013) are likely to differentially influence rivers, streams, lakes, reservoirs, and ponds, given their distinct contributions to global C fluxes.

2.3 | Global C flux estimates

C flux estimates from inland waters globally have received much research attention in recent years (Table 2, Figure 4). Estimates of C burial and export have remained largely constant, while estimates of C emissions have increased from the original estimate of 0.75 Pg C/year by Cole et al. (2007) to 3.88 Pg C/year by Drake et al. (2018) (Figure 4). Several refinements to global C emission estimates have been presented for inland waters, in part reflecting improvements to methods and upscaling techniques that better capture spatiotemporal variability (Bastviken et al., 2011; Battin et al., 2009; Deemer et al., 2016; DelSontro et al., 2018; Raymond et al., 2013; Tranvik et al., 2009) and new sources of C emissions that were not previously considered globally (Table 2). For example, diffusive C emissions from ponds (Holgerson & Raymond, 2016), CH_4 ebullition from lentic waterbodies (Rosentreter et al., 2021), degassing emissions from reservoirs (Harrison et al., 2021), and CO_2 emissions from dried inland waters (Keller et al., 2020) are all recently published updates (Figure 4). Recent estimates of CO_2 emissions from riverine hotspots in the Amazon (Sawakuchi et al., 2017) and sub-Saharan Africa (Borges et al., 2015) have also substantially increased the global inland water C emission estimate (Figure 4). On top of these, anthropogenic change has increased C input to inland waters by ~1 Pg C/year compared to pre-industrial input estimates (Regnier et al., 2013), suggesting alterations to C cycling and transport in inland waters.

By incorporating these refined data, we estimate the total global C emissions from inland waters to be 4.40 Pg C/year (95% CI: 3.95–4.85 Pg C/year [calculated via error propagation from uncertainty values in bold in Table 2 converted to estimates of standard deviation]), representing a 13% increase from the previous estimate in Drake et al. (2018). To balance this value and assuming the most recent estimates of aquatic primary production, burial, and export, terrestrial C inputs would need to be 5.76 Pg C/year (Figure 1). Of the C entering inland waters, 73% is emitted, while only 10% is buried and 17% is exported to oceans, where it may be emitted or buried. However, we are unable to provide a reasonable estimate of uncertainty for the global terrestrial C input flux due to very high uncertainty (>100% of mean) in literature estimates of aquatic primary production and C burial from Regnier et al. (2013) and no provided measure of uncertainty for C export to oceans in Li et al. (2017). Hence, we suggest caution in interpreting this global flux and further urge that studies report measures of uncertainty alongside mean estimates of global upscaled fluxes when appropriate.

Given the most recent estimate of the global net terrestrial C sink of 3.4 Pg C/year (standard deviation: ±0.9) (Friedlingstein et al., 2020), further increases in inland water C emission estimates and thereby terrestrial C inputs (subtracted from the net terrestrial C sink as lateral C loss) may considerably influence the estimated magnitude of the terrestrial C sink. These estimates of inland water C emissions and, correspondingly, C input from land should be considered a "moving target" (Drake et al., 2018) not simply due to improved sampling techniques and advancing methodology, but also because of the anthropogenic pressures affecting C fluxes in inland waters. In many cases, our knowledge and quantification of the anthropogenic impacts on inland water C fluxes are based on a few local case studies, on modeled estimates, or on cross-system comparisons, which complicate our ability to accurately upscale C fluxes resulting from anthropogenic pressures. As we review the influence of several globally widespread, anthropogenically driven climate and landscape changes on inland water C fluxes, four key research priorities emerge that have major potential consequences for improving our understanding and quantification of the "moving target" of inland water C fluxes (Box 1, Figure 4).

3 | CLIMATE AND LANDSCAPE CHANGE IMPACTS ON C FLUXES

In this section, we highlight some of the most significant and broad-scale anthropogenically driven climate and landscape changes that influence C fluxes into, within, and out of inland waters (Figure 1). Many of these anthropogenic pressures affect the quantity or quality of C entering inland waters from land, the hydrologic delivery of C to the receiving waterbodies, and, in some instances, both via different mechanisms. Some also influence C processing within inland waters via direct effects on in-situ environmental conditions, such as water temperatures or nutrient delivery. We highlight, where appropriate, the inland waterbody types that are likely to be most affected by specific drivers and mechanisms (Table 3), especially considering those waterbody types with disproportionate relative contributions to inland water C fluxes (Figure 2), as well as areas that are research priorities for understanding how C fluxes in inland waters respond anthropogenic pressures (Box 1). Additional climate-change-related impacts, such as extreme events, landscape-level impacts, salinity

TABLE 2 Estimates of global C fluxes by waterbody type. Mean global C flux values as presented in each study are listed, followed by minimum and maximum ranges, 95% confidence intervals, 25%–75% quantiles, or standard deviation (SD) if reported in the respective paper. For CH_4 emissions, "D" indicates diffusive CH_4 emissions, "E" indicates ebullitive CH_4 emissions, and "G" indicates degassing CH_4 emissions (in reservoirs). Bold indicates the most recent estimate of a C flux at a global scale for each waterbody type and these most recent estimates are used in Figure 2. Values for Rosentreter et al. (2021) are the median estimates (likely more conservative) when both median and mean were reported

C flux type	Estimated global C flux (Pg C/year)	Reference
Rivers & Streams		
CO_2 emissions	0.23 (range: 0.15–0.3)	Cole et al. (2007)
CO_2 emissions	0.32	Battin et al. (2009)
CO_2 emissions	0.55	Tranvik et al. (2009)
CO_2 emissions	0.92	Aufdenkampe et al. (2011)
CO_2 emissions	0.65	Lauerwald et al. (2015)
CO_2 emissions	**3.56 (1.80 + 0.37 + 1.39)**	**Raymond et al. (2013) + Borges et al. (2015) + Sawakuchi et al. (2017)**
CH_4 emissions (D + E)	0.001	Bastviken et al. (2011)
CH_4 emissions (D only)	0.020 (range: −0.026-0.759)	Stanley et al. (2016)
CH_4 emissions (D + E)	**0.004 (25%–75% quantiles: 0.001–0.016)**	**Rosentreter et al. (2021)**
C burial	assumed negligible	Cole et al. (2007)
C export to ocean	0.71	Cole et al. (2007)
C export to ocean	1.04	Cai (2011)
C export to ocean	0.95	Regnier et al. (2013)
C export to ocean	**1.06**	**Li et al. (2017)**
Lakes		
CO_2 emissions	0.11 (range: 0.07–0.15)	Cole et al. (2007)
CO_2 emissions	0.53	Tranvik et al. (2009)
CO_2 emissions	**0.485**	**Holgerson and Raymond (2016)**
CH_4 emissions (D + E)	0.054	Bastviken et al. (2011)
CH_4 emissions (D only)	0.007	Holgerson and Raymond (2016)
CH_4 emissions (D + E)	**0.026**	**Rosentreter et al. (2021)**
C burial	0.05 (range: 0.03–0.07)	Cole et al. (2007)
C burial	**0.09 (range: 0.04–0.18)**	**Mendonça et al. (2017)**
Reservoirs		
CO_2 emissions	0.273	St. Louis et al. (2000)
CO_2 emissions	0.28	Cole et al. (2007)
CO_2 emissions	0.037 (95% CI: 0.032–0.043)	Deemer et al. (2016)
CO_2 emissions	**0.090 (95% CI: 0.075–0.113)**	**Harrison et al. (2021)**
CH_4 emissions (D + E)	0.053	St. Louis et al. (2000)
CH_4 emissions (D + E)	0.015	Bastviken et al. (2011)
CH_4 emissions (D + E)	0.013 (95% CI: 0.009–0.022)	Deemer et al. (2016)
CH_4 emissions (D + E)	0.011 (25%–75% quantiles: 0.007–0.021)	Rosentreter et al. (2021)
CH_4 emissions (D + E)	0.008 (1 SD: 0.005–0.010)	Johnson et al. (2021)
CH_4 emissions (D + E + G)	**0.017 (95% CI: 0.010–0.044)**	**Harrison et al. (2021)**
C burial	0.18 (range: 0.16–0.2)	Cole et al. (2007)
C burial	**0.06 (range: 0.02–0.11)**	**Mendonça et al. (2017)**
Ponds		
CO_2 emissions	**0.086 (25%–75% quantiles: 0.03–0.13)**	**Holgerson and Raymond (2016)**
CH_4 emissions (D only)	0.005 (25%–75% quantiles: 0–0.008)	Holgerson and Raymond (2016)
CH_4 emissions (D + E)	**0.016 (25%–75% quantiles: 0.007–0.040)**	**Rosentreter et al. (2021)**
C burial	Unknown	

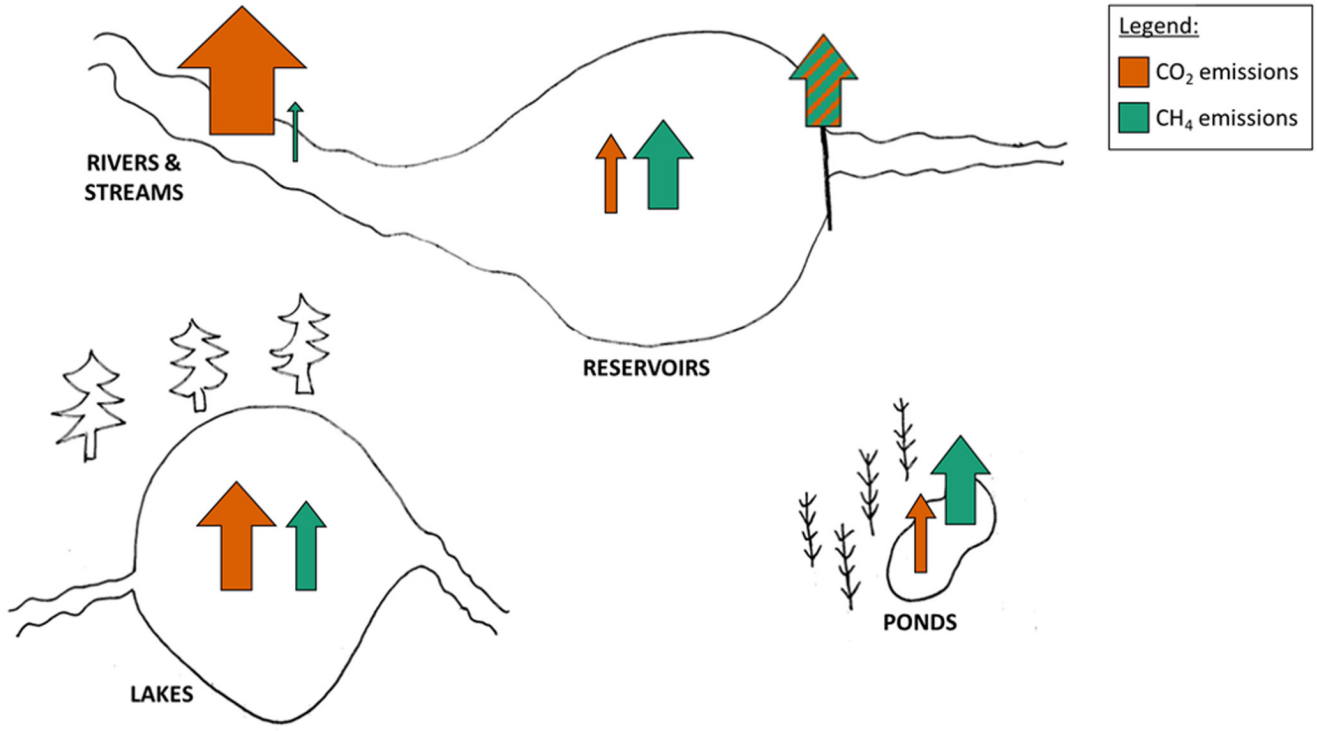

FIGURE 3 Conceptual comparison of the contribution of CO_2 (orange) vs. CH_4 (green; as CO_2-equivalents) emission pathways for each inland waterbody type. Size of arrows reflects relative contributions per waterbody type at a global scale and are for relative (not absolute) comparison within and across waterbody types only. Degassing emissions of CO_2 and CH_4 in reservoirs is understudied at a global scale compared to CO_2 and CH_4 emissions in the main reservoir, and relative emissions of CO_2 to CH_4 due to degassing have high uncertainty (thus are shown together as a hatched green/orange arrow).

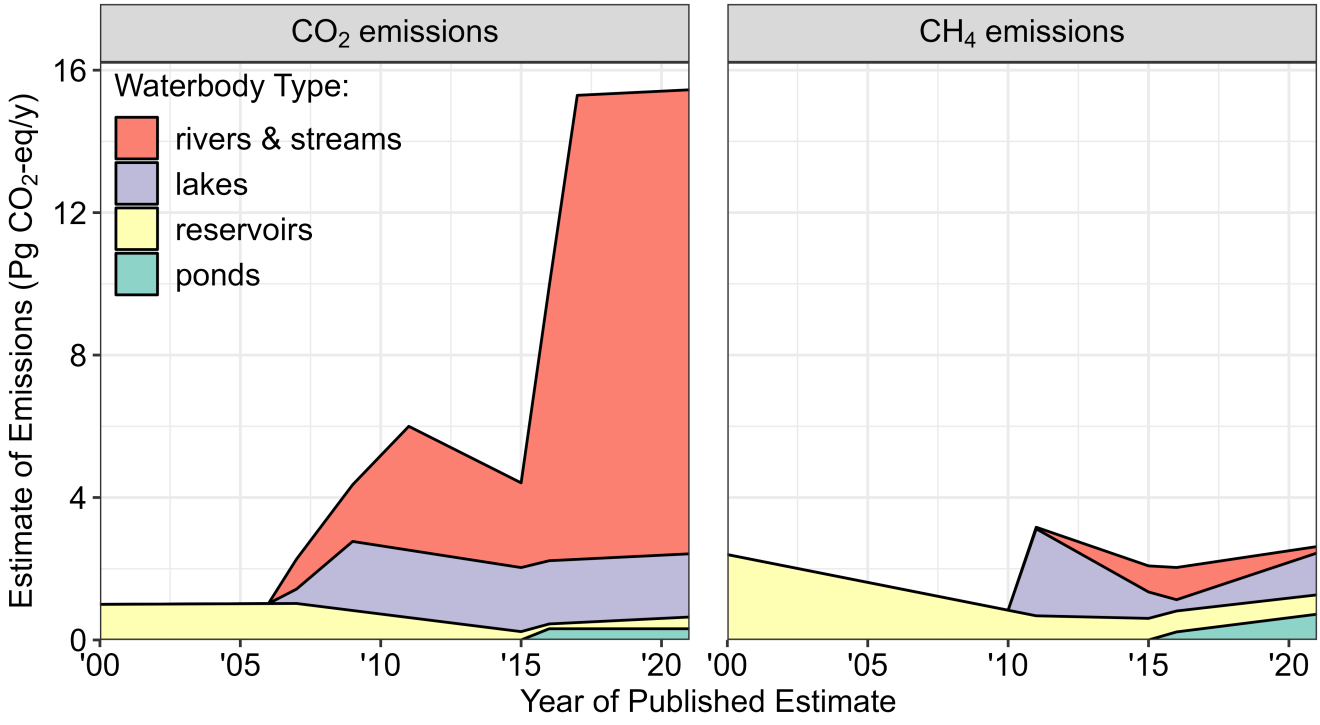

FIGURE 4 Time series of published C emission estimates from inland waters by waterbody type. Total height of each series indicates the total CO_2 or CH_4 (as CO_2-equivalents) emissions from all inland waters at the time of publication, and individual colors represent the relative contribution to each flux from each waterbody type.

BOX 1 Four key research priorities to improve the understanding of C fluxes in inland waters

1. Before-and-after measurements of C fluxes associated with climate change events and land-use change
2. Improved quantification of C inputs from land to inland waters
3. Assessment of the spatial coverage of small inland waterbodies (i.e., ponds, streams) and their contribution to global C fluxes
4. Integration of C fluxes from dried and drawdown areas

increases, and their interactions, have the potential to influence inland water C fluxes, but are generally less studied and therefore not included in this review (Box 2).

3.1 | Anthropogenically driven climate change

3.1.1 | Atmospheric CO_2 and temperature

Influences on C input

Global climate warming influences the terrestrial environment in several ways that can alter the quantity and quality of terrestrial C entering inland waters (Table 3). Atmospheric CO_2 fertilization and warming in some regions have been linked to terrestrial "greening," a response of longer growing seasons and increased foliage cover (Donohue et al., 2013; Myneni et al., 1997; Piao et al., 2020; Xu et al., 2013). Though this response is globally variable with high uncertainty regarding attribution to increasing atmospheric CO_2 concentrations (Walker et al., 2021), it is strongest in warm, arid regions where water use efficiency is high (Donohue et al., 2013). Such regions with greater OC in terrestrial standing stocks (Myneni et al., 2001; Piao et al., 2020; Xu et al., 2013) are projected to increase OC transport to receiving inland waters and thereby increase in-situ OC concentrations by up to 65% by 2100 (Finstad et al., 2016; Larsen et al., 2011a). Increased terrestrial evapotranspiration has been linked to altered hydrological connectivity and reduced streamflow (Lupon et al., 2018) that could affect the delivery of OC and IC from land to waterbodies. However, there is differing evidence for trends in evapotranspiration (Douville et al., 2013; Zeng et al., 2016; Zhang et al., 2015) vs. water use efficiency per unit C (Cheng et al., 2017) in response to CO_2 fertilization and terrestrial greening, with globally variable trends (Douville et al., 2013). The quality of OC that enters inland waters is also likely to be altered by shifts in terrestrial vegetation composition in the watershed (Ball et al., 2010; Butman et al., 2012), though less attention has been given to terrestrial OC quality compared to quantity (Kothawala et al., 2014). CO_2 fertilization may also influence aquatic primary production rates in inland waters, especially when nutrients and other resources are not limiting (Hamdan et al., 2018; Jansson et al., 2012), though it is highly dependent on in-situ CO_2 saturation levels (Vogt et al., 2017).

In northern latitudes, warming-induced permafrost melting leads to increased OC and IC inputs to inland waters. Permafrost melt, especially in organic-rich areas, releases "old" biolabile OC into northern inland water systems (Vonk et al., 2015; Wauthy et al., 2018). Increases in dissolved OC (DOC), particulate OC (POC), and DIC have been observed in northern lakes and rivers following permafrost melt (Tank et al., 2016; Vonk et al., 2015). Permafrost thaw also leads to altered hydrology and formation of new, C-rich thaw lakes (Vonk et al., 2015; Wauthy et al., 2018). Together, the increase in available C and formation or expansion of thaw lakes can lead to increased C production and emissions, especially as CH_4, from inland waters in northern regions (Cunada et al., 2021; Lapierre et al., 2013; Vonk et al., 2015; Walter et al., 2006; Walter Anthony et al., 2016). Many of these northern lakes have become greater sources of CH_4 and greater sinks of CO_2, but the two times greater rate of increase in CH_4 emissions combined with its higher global warming potential outweighs the CO_2 sink dynamics in these systems (Kuhn et al., 2021). These changes may contribute to an abrupt and critical feedback to global climate change (Turetsky et al., 2020). Alterations to C inputs to inland waters in response to climate warming are poorly quantified at local to global scales, with large consequences for the estimation of terrestrial C storage vs. loss (Drake et al., 2018) (Box 1).

Influences on C emissions and burial

Warming atmospheric temperatures can influence C emissions and burial within inland waters both directly and indirectly (Table 3). Surface water warming is a globally widespread phenomenon across inland waterbodies (Liu, Xie, et al., 2020; O'Reilly et al., 2015; Schneider & Hook, 2010; van Vliet et al., 2013). This warmer in-situ environment will lead to increased respiration rates (Gillooly et al., 2001; Marotta et al., 2014) and higher CO_2 and CH_4 emissions (Marotta et al., 2014; Vachon et al., 2017; Yvon-Durocher et al., 2014). Respiration increases more rapidly with warming temperatures compared to primary production, leading to an altered net C balance that could reduce net C sequestration by 13% with ~4°C of warming (Yvon-Durocher et al., 2010).

Globally, lentic systems have also experienced increases in the strength and duration of thermal stratification (Kraemer et al., 2015; Pilla et al., 2020; Woolway et al., 2021; Woolway & Merchant, 2019). These responses lead to longer periods of low oxygen in deep waters (Fang & Stefan, 2009; Foley et al., 2012; Knoll et al., 2018; Rösner et al., 2012) that promote anaerobic CH_4 production and deepwater accumulation of CO_2 and CH_4 via the physical barrier of stratification (Kankaala et al., 2006; Pu et al., 2020). In fact, total CO_2 emissions during the summer stratified period can be just 5% of that during mixing periods due to the accumulation of CO_2 in deep waters (Pu et al., 2020). The expected increasing accumulation of both CO_2 and CH_4 in deep waters during longer summer stratified periods may lead to greater C pulse emissions during autumn turnover (Ducharme-Riel et al., 2015; Kankaala et al., 2006; Kortelainen

et al., 2006; Vachon et al., 2017). While these longer stratified periods with anoxic conditions promote methanogenesis, they also promote OC burial over mineralization (Carey et al., 2018; Sobek et al., 2009). Hence, the relative rate of increase in CH_4 production vs. OC burial during anoxic conditions will ultimately determine the potential change in C source vs. sink dynamics.

Changes in ice cover duration can also influence C fluxes in inland waters (Benson et al., 2012; Magnuson et al., 2000; Sharma et al., 2019). In northern regions experiencing amplified climate warming (Cohen et al., 2014; IPCC, 2013) and rapid reductions in ice cover (Sharma et al., 2019), thermokarst lakes are estimated to emit 30% more CH_4 annually when ice cover is reduced by 20 days, due to a 13% longer open-water period corresponding to warmer in-situ temperatures that promote CH_4 production and emissions (Wik et al., 2016). Shorter ice cover duration can reduce the total accumulation of greenhouse gases under ice (Demarty et al., 2011) and can result in an earlier spring C emissions pulse that accounts for an average of 17% and 27% of annual CO_2 and CH_4 emissions, respectively (Cunada et al., 2021; Denfeld et al., 2018). Winter ice cover in Northern Hemisphere lotic systems can reduce CO_2 emissions by nearly half compared to the open water season (Liu et al., 2022), suggesting increased winter CO_2 emissions with decreasing river ice cover. These complex seasonal responses of C emissions are important to understanding both the phenology and potential increases in annual C emissions from inland waters given the global-scale trend of decreasing ice cover duration (Sharma et al., 2019).

3.1.2 | Precipitation

Changes in precipitation patterns primarily influence C transport from land throughout the inland water network. Climate change induced alterations to precipitation quantity and the frequency of extreme events are spatially heterogeneous globally (Table 3) (IPCC, 2013). Greater precipitation and discharge are linked with increased erosion in watersheds (Ludwig et al., 1996) and higher OC inputs to and OC concentrations in receiving waterbodies (de Wit et al., 2018; Dillon & Molot, 2005; Raymond & Saiers, 2010; Zhang et al., 2010). Extreme precipitation events lead to disproportionally high fluxes of DOC and POC from land to streams (Jennings et al., 2012; Zwart et al., 2016), which is pronounced during the rising hydrograph (Raymond & Saiers, 2010). The increased OC inputs during precipitation events follow a "pulse-shunt" concept (Raymond et al., 2016): the "pulse" increases terrestrial OC inputs to receiving waterbodies, but increased discharge can transport or "shunt" OC further downstream for processing, so OC is not necessarily processed near its point of entry. Furthermore, multiple, closely timed precipitation events can eventually decrease OC inputs as the OC source in the watershed becomes depleted (Dhillon & Inamdar, 2013). While increased precipitation and discharge lead to increased DOC and POC inputs (Lapierre et al., 2013; Larsen et al., 2011b), increased discharge generally has a dilution effect on CO_2 and CH_4 concentrations in rivers and streams (Dinsmore et al., 2013; Teodoru et al., 2009). In contrast, drought conditions disconnect surface and subsurface flowpaths (Kleine et al., 2021; Lake, 2003; Murphy et al., 2018) and have the opposite effects on C inputs. Lower precipitation and discharge reduce OC inputs (Blaurock et al., 2021; Dahm et al., 2003), while higher contributions from CO_2-rich groundwater during periods with low discharge result in higher in-situ CO_2 concentrations (Dinsmore & Billett, 2008; Öquist et al., 2009). Hence, both the intensity and frequency of precipitation events are important determinants of OC vs. IC inputs to inland waters.

C emissions from dried inland waters are an important, though often overlooked, component of the inland water C cycle. Intermittent or permanent losses of surface water are common in regions experiencing reduced precipitation and drought, such as the Middle East, central Asia, Australia, and the western United States (Pekel et al., 2016). For example, 18% of global inland waters experience intermittent drying (Keller et al., 2020), while the well-known drying of the Aral Sea has decreased surface area by 1278 km^2/year since the 1980s (Pekel et al., 2016). These dried areas are generally not included in either terrestrial or aquatic C budgets, representing a "blind spot" in global C accounting (Marcé et al., 2019). Dried areas generally have amplified CO_2 emission rates compared to their wetted condition (Gómez-Gener et al., 2016; Obrador et al., 2018), but lowered CH_4 emissions due to rapid oxidation and lack of an ebullitive pathway (Koschorreck, 2000; Marcé et al., 2019). Recent global syntheses of CO_2 emissions from dried inland waters reported that 0.12–0.22 Pg C/year should be added to the global CO_2 emission estimate (Table 4), a 6%–10% increase from recent global estimates (Keller et al., 2020; Marcé et al., 2019). Ponds are highly subject to seasonal drying (Marcé et al., 2019) and are hotspots of C emissions (Holgerson & Raymond, 2016), and incorporating dried areas of ponds increases their total C emission estimate by 26% (Table 4). Increases in C emissions from rivers, streams, and lakes are modest when dried areas are included in their global CO_2 emission estimates (Table 4), but represent a source of high variability in seasonal emissions (Liu et al., 2022). Furthermore, dried areas accumulate terrestrial leaf litter that, upon rewetting, can be rapidly mobilized and decomposed, contributing a CO_2 pulse equal to up to 10% from all permanent rivers and streams (Datry et al., 2018; del Campo et al., 2021). These changes in the spatial coverage and duration of dried areas, especially for highly variable small streams and ponds, are expected to increase in areas where reduced precipitation and drought are becoming more common with climate change. The resulting effects on C mobilization and processing have been rarely studied but likely represent an increasingly important component of inland water C fluxes (Keller et al., 2020) (Box 1). Additional drivers leading to dried areas in inland waters include drawdown in managed reservoirs, which we discuss below.

The chemical composition of precipitation has also changed in many regions of the world and has impacted the quality of C inputs, as DOC in particular, to and processing within inland waters. Reduced sulfur deposition throughout much of eastern North America and Europe (Monteith et al., 2007; Skjelkvåle et al., 2005;

TABLE 3 Examples of anthropogenic climate and landscape change drivers on C fluxes in inland waters

Driver variable	Response and mechanism	Expected C flux change	Most affected waterbody type(s)	Example and reference
Air temperature warming	Warmer temperatures and longer growing seasons promote terrestrial "greening"	+ C input	All	Increased vegetation coverage in Nordic watersheds associated with increased DOC in lakes due to greater allochthonous inputs (Finstad et al., 2016)
Air temperature warming	Melting permafrost releases formerly sequestered C and alters hydrology	+ C input + C emissions	All	Increases in DOC and POC during permafrost melt in organic-rich regions (Vonk et al., 2015); Thaw lakes are new locations of C cycling by inland waters and are hotspots of C emissions, especially CH_4 (Walter et al., 2006)
Air temperature warming	Warmer environment leads to increased metabolic rates and thereby faster production and respiration rates	+ C emissions	All	Sediment CO_2 and CH_4 production increased exponentially along a temperature gradient (Marotta et al., 2014); methanogenesis rates and ecosystem-level CH_4 emissions increased with temperature (Yvon-Durocher et al., 2014); increased CO_2 emissions in summer due to increased pelagic and benthic metabolism (Vachon et al., 2017)
Air temperature warming	Increased strength of stratification and prevalence of deepwater anoxia	+ C burial + CH_4 emissions	Lakes, reservoirs, and ponds	Warming inland waters result in stronger thermal stratification (Kraemer et al., 2015; Pilla et al., 2020) and is associated with increased prevalence of deepwater anoxic conditions (Jane et al., 2021) promoting OC burial (Carey et al., 2018; Sobek et al., 2009) and anaerobic CH_4 production
Air temperature warming	Reduced duration of ice cover	? C emissions	All (northern systems, primarily lentic)	Altered mixing phenology impacts pulse C emissions during spring and autumn in stratified systems (Denfeld et al., 2018); longer open-water periods with warmer waters promote greater metabolic rates, but also greater CO_2 and CH_4 accumulation in deep waters with long and strong thermally stratified periods (Kankaala et al., 2006; Pu et al., 2020)
Precipitation (amount)	Altered flux of C to inland waters due to increased runoff	+/− C input	Rivers & streams (→ downstream waterbodies)	(+) Stream water DOC & POC increases with discharge, and is disproportionately high during precipitation events (i.e., 57% of annual DOC flux from 5% of precipitation events, with up to 8 times greater POC fluxes) (Dhillon & Inamdar, 2013; Raymond & Saiers, 2010); (+/−) regions with increased precipitation experience increased OC input, and those prone to drought decreased OC input (Williamson et al., 2016); (−) increased discharge generally has dilution effect on in-situ CO_2 and CH_4 concentrations (Dinsmore et al., 2013)

(Continues)

TABLE 3 (Continued)

Driver variable	Response and mechanism	Expected C flux change	Most affected waterbody type(s)	Example and reference
Precipitation (acidity)	Soil recovery from acidification and reduced ionic strength	+ C input	Rivers & streams (→ downstream waterbodies)	Increasing soil acidity and reduced ionic strength leading to increased OC export from soils (Evans et al., 2012; Lawrence & Roy, 2021) and resulting in increased DOC concentrations in receiving waters (Monteith et al., 2007)
Wildfire	Aerial deposition of ash and particulate C	+ C input	All	Smoke plumes can transport and deposit ash and particulate OC to the surfaces of inland waters (Scordo et al., 2021)
Wildfire	Reduced vegetation cover and soil destabilization promotes erosion	+/− C input	Rivers & streams (→ downstream waterbodies)	POC inputs to receiving waters increase following wildfires (Smith, Sheridan, et al., 2011), while DOC inputs are more variable and may increase (Minshall et al., 2001), decrease (Evans et al., 2017), or show no significant change (Carignan et al., 2000; Mast & Clow, 2008) following wildfire
Landscape change	Deforestation reduces litterfall and evapotranspiration, and increases temperature and runoff from forest floors for faster decomposition	+/− C input	Rivers & streams (→ downstream waterbodies)	POC input to receiving waters generally decreases following deforestation (Kiffney & Richardson, 2010; Kreutzweiser et al., 2008; Santiago et al., 2011), while DOC input increases (Kreutzweiser et al., 2008)
Landscape change	Loss of wetland coverage	− C input	Rivers & streams (→ downstream waterbodies)	Loss of OC-rich wetlands (i.e., drained for agriculture) reduces DOC input into receiving waters (Royer & David, 2005)
Landscape change	Alteration of hydrological flowpaths for agricultural use	+ C input	Rivers & streams (→ downstream waterbodies)	Increased erosion (Pimentel et al., 1995) combined with tile drainage (Royer & David, 2005) and channelized agricultural streams (Blann et al., 2009) directly transport C from agricultural areas downstream
Landscape change	Runoff from lime application to agricultural fields	+ C input	Rivers & streams (→ downstream waterbodies)	Increased DIC fluxes in streams linked with agricultural practice of lime applications (Barnes & Raymond, 2009; Oh & Raymond, 2006)
Landscape change	Altered hydrology of engineered waterways and impervious surfaces in urban areas	+/− C input	Rivers & streams (→ downstream waterbodies)	(+) High connectivity in urban areas with engineered waterways facilitates OC transport (Kaushal & Belt, 2012); higher DOC transport to receiving waterbodies due to greater decomposition on impervious surfaces (Hobbie et al., 2013); (−) lower POC transport to receiving waterbodies due to removal of riparian zones and reduced litterfall (Roberts & Bilby, 2009), removal of yard waste (Templer et al., 2015)
Landscape change	Weathering of cement and high lawn CO_2 production	+ C input	Rivers & streams (→ downstream waterbodies)	DIC transport to receiving waterbodies increases in urban areas, up to 7.8 times more than forested watersheds (Barnes & Raymond, 2009)

TABLE 3 (Continued)

Driver variable	Response and mechanism	Expected C flux change	Most affected waterbody type(s)	Example and reference
Landscape change	Wastewater and sewage inputs from urban areas	+ C input + C emissions	Rivers & streams (→ downstream waterbodies)	Wastewater imports high levels of labile OC downstream (Daniel et al., 2002), even if treated in many cases (Kim et al., 2019; Meng et al., 2013), and enhances levels of in-situ CO_2 and CH_4 (Alshboul et al., 2016); emissions of both CO_2 and CH_4 increase downstream of wastewater treatment plants (Alshboul et al., 2016), especially with untreated wastewater (Kim et al., 2019)
Landscape change	Construction of impoundments alters hydrological flows (i.e., lotic to lentic)	+ C burial + C emissions	Reservoirs	Impoundments have high rates of OC burial (Mendonça et al., 2017), especially those with small surface areas such as agricultural impoundments (Downing et al., 2008); impoundments are hotspots of C emissions (Deemer et al., 2016; Regnier et al., 2013), especially when considering CO_2 emissions from drawdown areas (Keller et al., 2021)

BOX 2 Anthropogenically driven climate and landscape changes with potential to alter C fluxes in inland waters not covered in this review

- Interactions between climate change drivers and landscape changes
- Eutrophication of inland waters affecting primary production and C sequestration, including the role of elevated atmospheric CO_2 on in-situ production
- Role of wetlands and wetland loss in global C cycling
- Salinity changes in inland waters
- Landslides
- Hurricanes (wind and precipitation)
- Degassing emissions from agricultural irrigation

Smith, van Aardenne, et al., 2011) has allowed for soil recovery from acidification over the past several decades. This has reduced soil ionic strength (Lawrence & Roy, 2021) and allowed for greater DOC export from soils to inland waters (Evans et al., 2012). Within inland waters, increasing pH can also lead to more rapid decomposition of terrestrial organic matter (Mulholland et al., 1987). In response, DOC concentrations in many streams and lakes have increased (Evans et al., 2006; Monteith et al., 2007). Contrarily, sulfur emissions and deposition in regions like East Asia rose exponentially from the 1950s to mid-2000s (Smith, van Aardenne, et al., 2011), but are beginning to decrease (Duan et al., 2016). In parts of China, for example, soil acidity has decreased by 1.1 pH units since the early 1980s (Duan et al., 2016), and soil recovery may be delayed by frequent regional droughts (Duan et al., 2013). The region-specific interactions among changing amount, intensity, and chemistry of precipitation on OC inputs from land is an important consideration for understanding C fluxes in inland waters.

3.1.3 | Wildfires

Warmer air temperatures and drought conditions interactively drive the increased prevalence of wildfires (Dennison et al., 2014; Melillo et al., 2014), which alter the availability and quality of terrestrial C and corresponding fluxes to inland waters. Wildfires transform and release terrestrial OC via direct C emissions, aerial transport via smoke and ash plumes, and burning of terrestrial OC on land (i.e., vegetation, soils), producing partially burned biomass, charcoal, and soot, collectively referred to as pyrogenic OC (Santín et al., 2016). The latter two pathways can directly influence OC inputs into inland waters at local to broad spatial scales (Table 3). OC in smoke and ash plumes can travel long distances from wildfires, even traversing continents and oceans, and can directly enter inland waterbodies (Williamson et al., 2016). For example, smoke plumes from wildfires in California, USA, in 2018 reached Castle Lake (45–160 km away), and ash and POC deposition resulted in 46% higher surface water POC concentrations in the lake relative to previous, non-fire years (Scordo et al., 2021).

Second, inland waters in watersheds experiencing wildfire receive increased OC loads, though this varies by form between POC and DOC (Table 3). Following burning of terrestrial vegetation and soil destabilization, erosion and sediment export from the watershed increases POC to receiving inland waters (Smith, Sheridan, et al., 2011). This elevated POC input to waterbodies

TABLE 4 Previous estimates of CO_2 emissions from inland waterbody types when dried, intermittent, or drawdown areas are excluded (see Table 2) vs. included; data from Keller et al. (2020) and Keller et al. (2021)

Waterbody type	Global CO_2 emission estimates (Pg C/year)			Relative increase
	From surface waters (without dried areas)	From dried areas	Total	
Rivers & Streams	3.560 (see Table 2)	0.047	3.607	1%
Lakes	0.485 (Holgerson & Raymond, 2016)	0.026[a]	0.511	5%
Reservoirs	0.090 (Harrison et al., 2021)	0.026	0.116	29%
Ponds	0.086 (Holgerson & Raymond, 2016)	0.022	0.108	26%

[a]Taken as the difference between the value for lakes + resesrvoirs presented in Keller et al. (2020) minus the reservoir-only estimate presented in Keller et al. (2021).

generally declines as vegetation reestablishes over time (Petticrew et al., 2006; Smith, Sheridan, et al., 2011) but is also tightly linked with post-fire precipitation patterns (Betts & Jones, 2009; Shakesby & Doerr, 2006). For instance, 90% of ash from a burned watershed was delivered to a local reservoir in New Mexico, USA, within 1 year following post-fire rain events (Reneau et al., 2007). In contrast to the increases in POC delivered to inland waters following wildfire, responses of DOC are more variable and difficult to predict. Some studies report increasing DOC input from land or in-situ concentrations in waters following wildfire (Allen et al., 2003; McEachern et al., 2000; Minshall et al., 2001), while others report a decrease (Betts & Jones, 2009; Evans et al., 2017) or no change (Carignan et al., 2000; Lamontagne et al., 2000; Mast & Clow, 2008; Olefeldt, Devito, & Turetsky, 2013; Wagner et al., 2015). A decrease or no change in DOC input from land following wildfire may be explained by reduced vegetation and litter inputs to soils combined with damaged soil microbial communities that lead to reduced DOC production despite abundant POC (Betts & Jones, 2009; Carignan et al., 2000; Evans et al., 2017). Decreased DOC input from land may also be linked to temporary reacidification of soils and increased ionic strength, which has been suggested in burned watersheds in the United Kingdom historically experiencing acidification (Evans et al., 2017). The extent or severity of watershed burning may explain some of this variability in DOC input from land, potentially as a non-monotonic response (Rhoades et al., 2019). The quality of DOC following wildfire can also be altered (Santín et al., 2016), with DOC from burned areas having higher coloration (Clay et al., 2012) and lower biodegradability (Olefeldt, Turetsky, & Blodau, 2013) than unburned areas. However, alteration of OC quality following wildfire and effects on C processing in inland waters are not well studied, suggesting an important area for future research. These impacts of wildfire are vital to understand, especially considering the positive feedback loop of between wildfire prevalence, direct C emissions, and climate change leading to worsening drought across the globe (Balshi et al., 2009; Hurteau et al., 2014). Here, improved monitoring of C fluxes before vs. after extreme climate-related events such as wildfires can resolve our understanding of the specific impacts of individual events across inland waterbodies (Box 1).

3.2 | Human alterations to the landscape

Anthropogenic modifications to the landscape alter hydrological connectivity across the terrestrial–aquatic interface, as well as the quantity and quality of C entering inland waterbodies (Figure 1). While forested areas decreased by 25% between 1900 and 2005, agricultural areas nearly doubled and urban areas quadrupled in size (Meiyappan & Jain, 2012). Globally, land use and land cover changes have resulted in a net total C flux of 145 Pg C since 1850 (Houghton & Nassikas, 2017) with a 59% increase in C inputs from land to water (Regnier et al., 2013). In this section, we discuss the roles and mechanisms that globally widespread human alterations to the landscape have played in affecting inland water C fluxes, and note an array of additional anthropogenic landscape changes and their potential interactions that are not included in this review (Box 2). Furthermore, it is important to note that anthropogenically driven landscape changes are occurring in conjunction with climate change, and additional research is needed to understand how these interactions affect C transport and processing in inland waters. Our understanding of the magnitude and mechanisms by which these landscape changes affect C fluxes, especially C input from land to inland waters, will be greatly improved by measurements of specific C fluxes before, during, and after land conversion (Box 1).

3.2.1 | Deforestation

Deforestation is a worldwide practice that removes trees for pulp and paper production or in favor of agriculture or urban areas. Unsurprisingly, POC transport to receiving waterbodies decreases following forest harvest due to reduced vegetation and litterfall (Kiffney & Richardson, 2010; Santiago et al., 2011; Webster

et al., 1990). In contrast, transport of DOC generally increases after harvest, but often declines within 3–5 years (Carignan et al., 2000; Kreutzweiser et al., 2008; Laudon et al., 2009; Schelker et al., 2012; Schelker et al., 2014). Here, clear-cutting can result in warmer soil temperatures and thereby faster decomposition rates producing DOC, while reduced evapotranspiration can result in greater runoff during precipitation events that delivers DOC to adjacent inland waterbodies (Table 3) (Kreutzweiser et al., 2008). The altered C inputs to waterbodies following deforestation can have important implications for their ecosystem structure and function (Carignan et al., 2000; Schelker et al., 2014), including potential increases in C emissions associated with higher DOC concentrations (Lapierre et al., 2013). In general, the long-term effects of deforestation on C fluxes to inland waters will depend on the future use of the land. For example, forested lands converted to agriculture or urban areas will experience significant changes (see sections below); on the other hand, forestry rotation may provide intermittent or cyclic effects on C fluxes due to forest regrowth and cutting cycles, as has been suggested for fast-growing *Eucalyptus* plantations (Santiago et al., 2011).

3.2.2 | Draining of wetlands

As with deforestation, drainage of natural wetlands for agricultural or other anthropogenic purposes has occurred at a global scale. Estimates suggest that over 87% of wetlands have been lost since 1700, leaving only 13% of natural wetlands intact globally (Davidson, 2014). C fluxes specifically in wetlands are not discussed in detail in this review, as they are often considered an intermediate between terrestrial and aquatic ecosystems (Cole et al., 2007; Drake et al., 2018). High watershed coverage of wetlands is associated with high DOC transport into and concentrations within receiving inland waterbodies (Eckhardt & Moore, 1990; Kortelainen, 1993; Mulholland & Kuenzler, 1979; Raymond et al., 2004). Hence, as wetland coverage decreases, DOC transport to inland waters also decreases (Royer & David, 2005). This reduction in OC inputs to inland waters can be especially important to consider given the additional impact of agriculture, the most common reason for wetland drainage (van Asselen et al., 2013), on inland water C fluxes.

3.2.3 | Agriculture

Land used for agriculture covers about one-third of terrestrial land globally (Ramankutty et al., 2018), a near doubling over the past century (Meiyappan & Jain, 2012). The conversion of land formerly covered by forests or wetlands, for example, influences hydrology and C transport to receiving inland waters as described above; however, there are several additional direct effects on C transported to inland waters due to agricultural activities following land conversion. Practices such as tile drainage and channelization of agricultural streams alter hydrological flowpaths (Table 3), and more efficiently and directly transport OC and IC to receiving waterbodies (Blann et al., 2009; Dalzell et al., 2011; Royer & David, 2005; Yang et al., 2022). Agricultural irrigation that sources groundwater can result in rapid degassing of CO_2 (Macpherson, 2009), compared to slower degassing of CO_2 from groundwater entering less-disturbed streams (Doctor et al., 2008). However, only one study has quantified this emissions pathway for a row-crop field in Michigan, USA (McGill et al., 2018), with degassing from irrigation contributing much less to agricultural C emissions than other components of irrigation (i.e., agricultural ponds, infrastructure) (Aguilera et al., 2019).

Globally, approximately 133 Pg C from soils has been lost due to land conversion to agriculture, which has been especially pronounced in the past 200 years (Sanderman et al., 2017). Common agricultural practices such as tilling can increase the transport of total C in agricultural watersheds due to high erosion rates (Kelsey et al., 2020; Walmsley et al., 2011), which can be three to four orders of magnitude higher than in undisturbed forested watersheds (Berhe et al., 2018; Pimentel et al., 1995). However, the effect on individual C forms can vary, with traditional tillage sometimes reducing the flux of DIC from soils to streams compared to reduced tillage due to greater losses of CO_2 due to soil aeration (Walmsley et al., 2011), while other studies indicated increased DIC flux in traditional compared to non-tilled agricultural fields (Kelsey et al., 2020). Similarly, tillage effects on DOC transported to receiving inland waters can vary from more than doubling in traditional vs. non-tilled fields (Kelsey et al., 2020) to having no clear effect (Van Gaelen et al., 2014; Walmsley et al., 2011). Various agricultural practices can also alter the composition of terrestrial OC that can enter water bodies (Graeber et al., 2012; Stanley et al., 2012; Wilson & Xenopoulos, 2008), as well as the production and cycling of DOC within agriculturally influenced inland waters (Giling et al., 2014). For example, streams in agricultural catchments compared to those in forested watersheds had higher seasonal variability in DOC concentrations with a more complex humic molecular signature, suggesting lower biolability of exported DOC for in-situ processing (Graeber et al., 2012). In agricultural areas used for livestock, animal waste containing high OC and DIC can enter nearby waterbodies, such as agricultural ponds or streams via runoff; however, this source of C to inland waters has not been quantified despite the near doubling of livestock production and associated animal waste globally since the 1960s (Thornton, 2010; United States EPA, 2015). These studies highlight how interactions between altered hydrologic pathways (i.e., irrigation, tile drainage) and C quantity and quality (i.e., animal waste, crop detritus, agricultural soils) together can influence the flux of C from agricultural lands to adjacent inland waters.

Agricultural ponds that receive high C inputs have some of the highest OC burial rates reported for inland waters (Downing et al., 2008; Mendonça et al., 2017). However, the high C and nutrient inputs that result in high primary productivity and eutrophic conditions, combined with the small size and depth of these agricultural ponds (Downing et al., 2008), suggest that they are important sources of both CO_2 and CH_4 emissions (Beaulieu et al., 2019; DelSontro et al., 2018; Holgerson & Raymond, 2016).

While fertilizer application in agricultural watersheds leads to high nutrient runoff with implications for eutrophication downstream, agricultural liming can result in increased DIC fluxes (Barnes & Raymond, 2009; Oh & Raymond, 2006) compared to undisturbed systems where bedrock weathering is the primary DIC input (Regnier et al., 2013). Lime application increases chemical weathering and interacts with fertilizer application to increase lime dissolution, both contributing to increased bicarbonate fluxes downstream (Barnes & Raymond, 2009; Oh & Raymond, 2006). This effect of agricultural liming accounts for 29% of bicarbonate transport from agricultural watersheds (Oh & Raymond, 2006), which is up to four times more DIC than transported from undisturbed forested watersheds (Barnes & Raymond, 2009). Small inland waterbodies, such as agricultural ponds, have a less understood spatial distribution and understudied C fluxes despite their disproportionately large role in C burial and C emissions (Figure 2), warranting future research in this area (Box 1).

3.2.4 | Urbanization, wastewater, and sewage

As with land conversion to agriculture, land conversion to urban areas is associated with many anthropogenic practices that influence C fluxes to inland waters (Table 3). Altered hydrological flowpaths, whether by modification or burial of headwater streams or creation of engineered waterways, have high connectivity with the urban landscape that facilitates efficient C transport downstream (Kaushal & Belt, 2012). Impervious surfaces, which dominate urban landscapes, have rapid rates of organic matter decomposition (Hobbie et al., 2013) that can result in increased DOC transport to highly connected urban streams (Kaushal & Belt, 2012). In contrast to DOC, POC transport from urban areas is generally low in part due to this rapid decomposition (Hobbie et al., 2013), but also due to removal of vegetation, especially in riparian zones (Roberts & Bilby, 2009), and yard waste (Templer et al., 2015) that reduces the POC availability. Though, grass clippings from mowing can produce POC for transport even in the absence of trees and larger vegetation (Roberts & Bilby, 2009). Urban areas also transport DIC to receiving waterbodies from weathering of cement, high CO_2 production in fertilized lawns, and wastewater and sewage inputs (Baker et al., 2008; Barnes & Raymond, 2009; Daniel et al., 2002; Davies et al., 2010; Moore et al., 2017; Scott et al., 2006). In fact, urban transport of DIC to receiving inland waters is two times greater than in agricultural watersheds used for croplands and nearly eight times greater than in undisturbed forested watersheds where bedrock weathering is the dominant DIC source (Barnes & Raymond, 2009; Regnier et al., 2013).

Important in heavily populated urban areas is the export of wastewater and sewage. OC from wastewater tends to be very labile, so it can be rapidly decomposed and emitted as CO_2 and CH_4 while traveling downstream (Abril et al., 2002; Daniel et al., 2002; Hu et al., 2018; Meng et al., 2013; Zhang et al., 2021). Globally, approximately 52% of wastewater is treated, though this proportion varies substantially by country largely as a function of economic class (Jones et al., 2021). Untreated wastewater, accounting for ~48% of total wastewater generated globally (Jones et al., 2021), can be up to 31 times more labile than treated wastewater (Kim et al., 2019), but even treated wastewater transports large quantities of labile OC downstream (Kim et al., 2019; Meng et al., 2013). Secondary treatment of wastewater can remove an additional 85% of organic matter (United States EPA, 1998), but rates of secondary treatment of wastewater globally are unknown due to inconsistent reporting (Jones et al., 2021). Highly labile OC inputs from wastewater subsequently increase in-situ CO_2 and CH_4 concentrations in the water column (Alshboul et al., 2016). In turn, emissions of both CO_2 and CH_4 increase downstream of wastewater treatment plants (Alshboul et al., 2016) and are especially high when wastewater is not treated (Kim et al., 2019). The increased OC and IC inputs from these anthropogenic practices in urban areas will likely amplify the C emissions from inland waters both within and downstream of urban areas.

3.2.5 | Impoundments

Construction of impoundments for drinking water, irrigation, flood control, or hydropower has been an anthropogenic alteration of hydrological flows for decades (Table 3) (Lehner et al., 2011). Impoundments effectively transition lotic waters to lentic systems, and, as such, C emissions and burial rates increase following impoundment (Deemer et al., 2016; Regnier et al., 2013). Young reservoirs may have especially high CO_2 and CH_4 emissions since flooded labile terrestrial C becomes available for rapid decomposition (Barros et al., 2011; Bastien et al., 2011; Demarty & Tremblay, 2019; St. Louis et al., 2000). Planned construction of new dams and impoundments has surged, with over 3700 dams slated for construction during the 2020s and 2030s (Zarfl et al., 2015), suggesting a potentially important role for young reservoirs in altering C fluxes in inland waters. Though C emissions can be high from reservoirs (Deemer et al., 2016; Regnier et al., 2013), they are also hotspots of OC burial, with rates approximately 6.5 times greater than natural lakes (Mendonça et al., 2017). OC burial rates increase with decreasing surface area (Clow et al., 2015) and can be especially high in small impoundments in agricultural areas due to high sediment input and rapid sedimentation rates (Downing et al., 2008; Mendonça et al., 2017). While increased OC burial may be a positive consequence of impoundment construction, dredging of reservoir sediments can limit the time scale in which OC is buried by mobilizing it downstream or removing it from waterbodies entirely, potentially re-introducing OC into the active pool for processing and emission (Maeck et al., 2013). The consequences of dredging on C burial and emissions are largely unstudied, contributing to high uncertainty of OC burial rates in reservoirs and the time scale on which they operate. Combined,

the balance of both increased C emissions and burial in impoundments is a key consideration for the role of reservoirs as net C sinks vs. sources.

Reservoirs are unique in their water-level management, which has important implications for reduced water depth and drawdown areas. During drawdown, CO_2 and CH_4 can be emitted via degassing through turbines and spillways that may have been otherwise processed or buried within the reservoir (Bastien et al., 2011)—mechanistically similar to degassing of high-DIC groundwater in streams (Doctor et al., 2008) and due to irrigation practices (McGill et al., 2018). However, this anthropogenic degassing pathway can more than double total C emissions from reservoirs (Harrison et al., 2021). Estimates of CO_2 and CH_4 degassing emissions are poorly constrained due to high uncertainty (Harrison et al., 2021) and warrant further study. Dried areas following water-level drawdown have heightened CO_2 emissions (Keller et al., 2020; Marcé et al., 2019), which can be over six times greater than wetted conditions (Kosten et al., 2018). Dried areas following drawdown account for approximately 15% of global reservoir surface area (Keller et al., 2021). When drawdown areas are accounted for, global CO_2 emissions from reservoirs increase by 29% (Table 4) (Keller et al., 2020), and in fact flip reservoirs from net sinks to sources of C at a global scale (Keller et al., 2021). Furthermore, while drawdown areas generally contribute little to CH_4 emissions (Keller et al., 2021), decreases in water level can lead to increases in ebullitive CH_4 emissions elsewhere in the reservoir (Beaulieu et al., 2018; Harrison et al., 2017). The reduced hydrostatic pressure and time for CH_4 oxidation in the water column can lead to temporary increases in CH_4 emissions by 1.4–77 times (Beaulieu et al., 2018). In some systems, CH_4 emissions from the drawdown period represent up to 90% of annual CH_4 emissions (Harrison et al., 2017). Hence, a complete understanding of total C emissions from reservoirs must integrate the drawdown impacts on dried areas and lowered water depth, but this is only beginning to be explored at a global scale (Box 1).

4 | FUTURE RESEARCH DIRECTIONS AND OUTLOOK

Our updated global C emissions estimate from inland waters of 4.40 Pg C/year (95% CI: 3.95–4.85 Pg C/year) incorporates the most recently available information on the impacts of anthropogenically driven climate and landscape changes on C fluxes in inland waters (Figure 1). However, we have highlighted several responses to anthropogenic pressures that remain understudied but that can strongly influence these estimates of C fluxes to and from inland waters. These emerging research priorities and related methodological advancements will be vital to understanding the full suite of anthropogenic pressures and their interactions on inland water C fluxes, the associated mechanisms, and specific estimates used for upscaling of global C fluxes, especially C input from land and C emissions.

4.1 | Key research priorities

We suggest four specific future research directions that will fill notable data gaps and improve understanding of global C fluxes from inland waters (Box 1), including some key effects of anthropogenic climate and landscape changes that are understudied and thus poorly quantified at this time (Box 2). First, before-and-after measurements of C fluxes can contribute to better quantification of the net changes in C cycling directly associated with climate change events or land-use change (Eddy & Gergel, 2015). Few studies have collected measurements of C fluxes prior to human alteration of inland waters (Bastien et al., 2011; Demarty & Tremblay, 2019), but these comprehensive measurements more accurately assess the direct net impact of land-use change on C fluxes (Prairie et al., 2018). However, the paucity of data collected before an anthropogenic disturbance or modification highlights the difficulty in doing this in practice (Jager et al., 2022; Prairie et al., 2018); alternatives for assessing the net alterations to C fluxes due to land-use change include modeling counterfactual scenarios, comparison with similar non-altered ecosystems, or evaluation along an impact gradient.

Second, we must improve the understanding of anthropogenic impacts specifically on C inputs to inland waters and effects on C emissions downstream (Figure 1). Nearly all of the climate and landscape change drivers of C fluxes reviewed here influence C inputs, but responses of both the quantity and quality of C inputs are understudied at local to global scales, with global estimates considered highly uncertain (Drake et al., 2018). Global estimates of C input from land are typically calculated as the remainder of the mass balance equation of C fluxes in inland waters, rather than direct measurements. Incorporating direct measurements of C input from land can improve C flux balancing for inland waters, but also refine terrestrial C accounting with estimates of C loss across the terrestrial–aquatic interface. Studies measuring terrestrial C cycling and net ecosystem C balance often do not incorporate lateral loss of C from land to water (Aufdenkampe et al., 2011; Liu et al., 2022; Webb et al., 2018); however, estimates of lateral C flux to inland waters (often using the mass balance approach described in Section 2) are increasing, especially in watershed-scale studies (Billett et al., 2004; D'Acunha et al., 2019; Genereux et al., 2013; Nakhavali et al., 2021; Song & Wang, 2021; Zhang et al., 2014).

Third, small waterbodies, including headwater streams and ponds, are hotspots of C cycling, especially emissions (Holgerson & Raymond, 2016; Rosentreter et al., 2021), but their role globally remains difficult to accurately assess given the limitations of satellite imagery in quantifying their areal coverage (McManamay et al., 2018). Ponds likely have high C burial, with man-made agricultural ponds having among the highest rates of C burial across all lentic waterbodies (Downing et al., 2008; Mendonça et al., 2017; Taylor et al., 2019). However, despite the large number of ponds worldwide (Downing, 2010; Holgerson & Raymond, 2016), no comprehensive global estimate of C burial in ponds has been conducted (Figure 2).

Finally, measurements of dried and drawdowns areas must be integrated into global C flux estimates for inland waters. Across the

United States, for example, intermittent streams are drying earlier and for longer periods throughout the year (Stokstad, 2021; Zipper et al., 2021). However, these currently represent a significant "blind spot" in understanding terrestrial–aquatic C linkages and fluxes (Marcé et al., 2019), with only one existing estimate of global C emissions from dried and drawdown areas (Keller et al., 2020). A recent international initiative has begun to research intermittent rivers and ephemeral streams globally (Datry et al., 2016), resulting in publications highlighting organic matter accumulation, processing, and export in dried areas from joint experiments (Datry et al., 2018; Schiller et al., 2019; Shumilova et al., 2019). Beyond this effort, alterations to C fluxes in dried areas, specifically the increase in CO_2 emissions and potential reduction in C burial, have not been broadly assessed. With these suggestions, long-term monitoring at high spatiotemporal resolutions will be able to fill in data gaps in the global inland water C cycle.

4.2 | Improved resolution of spatial and temporal variability

At regional to global scales, the spatial variability of C fluxes in inland waters requires further study. There are known spatial biases in measurements of CH_4 emissions from reservoirs, for example, with a bias toward measurements from reservoirs in the tropics and few from Africa or Australia (Deemer et al., 2016). Improved spatially resolved sampling efforts can identify hotspots of C emissions, such as in the Amazon (Sawakuchi et al., 2017) and sub-Saharan Africa (Borges et al., 2015). Conversely, identifying regions that, despite model predictions (Raymond et al., 2013), are not actually hotspots is an important step for global upscaling efforts. For example, rivers in southeast Asia were modeled to have high CO_2 emissions (Raymond et al., 2013), but empirical measurements indicated these rivers were not hotspots of CO_2 emissions (Wit et al., 2015). Such data can improve empirical relationships between C emissions and waterbody characteristics used in global upscaling estimations (DelSontro et al., 2018), alongside refined estimates of inland water surface area from geospatial mapping tools (McManamay et al., 2018). One challenge is the inclusion of small streams, ponds, and dried and drawdown areas (Holgerson & Raymond, 2016; Marcé et al., 2019; Rosentreter et al., 2021), which have a largely unknown spatial distribution and are difficult to identify from satellite imagery. The variable spatial extent and intermittency of these systems complicate global upscaling efforts but are significant contributors to inland water C fluxes (Holgerson & Raymond, 2016; Keller et al., 2020; Rosentreter et al., 2021).

Within-system spatiotemporal variability of C fluxes can be high, as small sample sizes in space or time can introduce bias to system-wide C flux estimates. For example, tributary regions and areas near inflows of reservoirs have amplified CH_4 emissions (Beaulieu et al., 2016), fast sedimentation rates, and high C burial rates (Koren & Klein, 2000; Mendonça et al., 2016; Sobek et al., 2009; Xu et al., 2009) compared to open-water regions. In rivers, CO_2 emissions generally decrease downstream while CH_4 emissions increase (Battin et al., 2008; Crawford et al., 2016) following the changing sources of CO_2 and CH_4 longitudinally throughout river networks (Hotchkiss et al., 2015). Temporal variability must also be considered given the impacts of climate change on seasonality, interannual variability, and extreme events. Periods of spring and autumn mixing can be hot moments of C emissions in many systems (Demarty et al., 2011; Denfeld et al., 2018; Schilder et al., 2016; Vachon et al., 2017), while others, like a midlatitude eutrophic reservoir, experience the highest C emissions in summertime (Beaulieu et al., 2014). C inputs to inland waters can also vary temporally, related to wet–dry seasonality or to less predictable extreme rain or drought events (Dahm et al., 2003; Jennings et al., 2012; Raymond & Saiers, 2010; Zwart et al., 2016). Temporal variability also plays an important role in the extent of dried and drawdown areas, which are large contributors to CO_2 emissions (Keller et al., 2020; Keller et al., 2021; Liu et al., 2022; Marcé et al., 2019). Advancing sampling designs to capture the high spatiotemporal variability of C fluxes within inland waters will be key to accurately upscaling C fluxes regionally and globally.

4.3 | Methodological advancements

Improved methods for comprehensively measuring spatiotemporal variability of C fluxes in inland waters are necessary to quantify their active role at a global scale. In lakes, ponds, and reservoirs, CH_4 ebullition must be accounted for in addition to CO_2 and CH_4 diffusion, otherwise total C emissions can be underestimated by more than 50% (Deemer et al., 2016). Additional or alternative methods for capturing C emissions pathways include acoustic measurements of CH_4 ebullition (e.g., DelSontro et al., 2015; Linkhorst et al., 2020; Liu, Yang, et al., 2020; Ostrovsky et al., 2008; Wilkinson et al., 2019) and eddy covariance flux towers (Deshmukh et al., 2014; Eugster et al., 2011; Huotari et al., 2011; Schubert et al., 2012; Vesala et al., 2006), though each has their own costs and challenges. Including estimates of CO_2 and CH_4 degassing emissions can increase global C emission estimates from reservoirs by 62% (Harrison et al., 2021), though this pathway is rarely quantified (Abril et al., 2005; Soued & Prairie, 2020). Advances in high-resolution and high-frequency technologies to measure various pathways of C emissions will be vital to understanding their relative contributions and spatiotemporal patterns.

Advancing methods to improve measurements of C fluxes in inland waters over space and time can be applied to inform inland water C processing in coupled empirical-model approaches to improve the outdated "passive pipe" model (Cole et al., 2007; Webb et al., 2018). While incorporation of inland waters in C budgets and Earth System Models has improved (i.e., Du et al., 2020; Nakayama, 2017; Nakayama, 2020; Tian et al., 2015), relevant estimates of their C fluxes beyond C export to oceans remain limited at a global scale. For example, inland waters were not included in the global C cycle in the IPCC report in 2007 except to transport

C directly to oceans (Denman et al., 2007); in the 2013 report, emissions and burial were incorporated, though high uncertainty in emission estimates were noted (Ciais et al., 2013). Furthermore, terrestrial CO_2 flux estimates based on upscaled eddy covariance measurements (e.g., Zeng et al., 2020) can be biased because they do not account for lateral loss of C across the terrestrial–aquatic interface. As hydrological connectors of land and oceans, inland waters and their C fluxes should be more explicitly accounted for in such models with the growing body of available data, especially given the complex influences of climate and landscape changes that interact with C fluxes through land, inland waters, and oceans. In fact, C processing by inland waters can offset 1%–590% of terrestrial net ecosystem production depending on watershed ecosystem type (Webb et al., 2018), again stressing the important and active role of inland waters in accounting for terrestrial C at the watershed scale, global C cycling, and Earth System Models.

5 | CONCLUSIONS

Inland waters play a dynamic role in the global C cycle, and their C fluxes will continue to be impacted by the complex and interactive effects of anthropogenically driven climate and landscape changes. The uncertainty in current estimates of inland water C fluxes due to high spatiotemporal variability, combined with the future changes in C fluxes due to anthropogenic pressures, underscores the importance of accurately quantifying and upscaling C fluxes in inland waters globally. C emission estimates from inland waters should be focal, as these estimates have risen substantially due to improved methods capturing spatiotemporal heterogeneity, finer-resolution upscaling, and inclusion of small and dried waterbodies. Our current C emission estimate of 4.40 Pg C/year (95% CI: 3.95–4.85 Pg C/year) is nearly six times greater than the initial estimate from Cole et al. (2007) (Figure 4). Continued increases in C inputs to inland waters due to climate and landscape change (Regnier et al., 2013) have a very high likelihood of returning to the atmosphere via emissions from inland waters. This is especially important if emitted as potent CH_4, where freshwaters account for 17% of global CH_4 emissions despite covering only ~4% of the global land area (Saunois et al., 2016). This large C flux from inland waters returned back to the atmosphere alters our understanding of the source–sink balance of these waterbodies (Keller et al., 2021) and the efficacy of C sequestration in terrestrial systems (Webb et al., 2018). Four much needed future research priorities (Box 1) for understanding C fluxes in inland waters and their role in the global C cycle are: (1) before-and-after measurements of C fluxes associated with climate change events and landscape changes, (2) quantification of C input from land, (3) improved assessment of spatial coverage and contributions of small inland waterbodies to C fluxes, and (4) integration of dried and drawdown areas to global C flux estimates. Achieving these will require improved sampling designs and methodological and modeling advancements, as well as measurements that capture spatiotemporal variability within and across inland waters. The expected responses of C fluxes in inland waters to the vast array of anthropogenic pressures will alter global C cycling, including the net C emissions from inland waters feeding back to anthropogenic climate change.

ACKNOWLEDGMENTS

This research was supported by the U.S. Department of Energy (DOE), Office of Energy Efficiency and Renewable Energy, Water Power Technologies Office, and the Climate Change Science Institute and Environmental Sciences Division at Oak Ridge National Laboratory (ORNL). ORNL is managed by UT-Battelle, LLC, for the U.S. DOE under contract DE-AC05-00OR22725. We thank Dr. J. Fellows and Dr. S. Wullschleger for project support and for championing Early Career initiatives, A. Malin for graphics support, and Dr. A. Walker and two reviewers for useful comments that greatly improved earlier versions of this manuscript.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

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How to cite this article: Pilla, R. M., Griffiths, N. A., Gu, L., Kao, S.-C., McManamay, R., Ricciuto, D. M., & Shi, X. (2022). Anthropogenically driven climate and landscape change effects on inland water carbon dynamics: What have we learned and where are we going? *Global Change Biology*, *00*, 1–29. https://doi.org/10.1111/gcb.16324