Greenhouse Gas Emissions from Reservoir Water Surfaces: A New Global Synthesis

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Collectively, reservoirs created by dams are thought to be an important source of greenhouse gases (GHGs) to the atmosphere. So far, efforts to quantify, model, and manage these emissions have been limited by data availability and inconsistencies in methodological approach. Here, we synthesize reservoir CH_4, CO_2, and N_2O emission data with three main objectives: (1) to generate a global estimate of GHG emissions from reservoirs, (2) to identify the best predictors of these emissions, and (3) to consider the effect of methodology on emission estimates. We estimate that GHG emissions from reservoir water surfaces account for 0.8 (0.5–1.2) Pg CO_2 equivalents per year, with the majority of this forcing due to CH_4. We then discuss the potential for several alternative pathways such as dam degassing and downstream emissions to contribute significantly to overall emissions. Although prior studies have linked reservoir GHG emissions to reservoir age and latitude, we find that factors related to reservoir productivity are better predictors of emission.

Keywords: reservoir, methane, greenhouse gas, eutrophication, ebullition

The construction and operation of over 1 million dams globally (Lehner et al. 2011) has provided a variety of services important to a growing human population (e.g., hydropower, flood control, navigation, and water supply), but has also significantly altered water, nutrient, and ecosystem dynamics and fluxes in river networks. Much attention has been paid to negative impacts of dams on fish and other riverine biota, but the indirect effects on biogeochemical cycling are also important to consider. Although reservoirs are often thought of as “green” or carbon-neutral sources of energy, a growing body of work has documented their role as greenhouse gas (GHG) sources. Artificial reservoirs created by dams are distinct from natural systems in a number of key ways that may enhance GHG emissions from these systems. First, the flooding of large stocks of terrestrial organic matter may fuel microbial decomposition, converting the organic matter stored in above and below ground biomass to carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Second, reservoirs often experience greater fluctuations in water level than natural lakes. Drops in hydrostatic pressure during water level drawdowns can enhance CH_4 bubbling (e.g., ebullition) rates at least over the short term (Maeck et al. 2014). This enhanced ebullition may then decrease the fraction of CH_4 that is oxidized to CO_2, a less potent GHG, by methane oxidizing microbes (Kiene 1991). Finally, the high catchment area–to–surface area ratios and close proximity to human activities (Thornton et al. 1990) characteristic of many reservoirs are likely to increase the delivery of organic matter and nutrients from land to water (relative to natural lakes), potentially fueling additional decomposition.

St. Louis and colleagues (2000) raised the possibility that reservoir GHG emissions contribute significantly to global budgets (table 1). Since that influential review appeared, and in part because of the attention it generated, researchers have quantified GHG fluxes from more than 200 additional reservoirs, and have synthesized regional emissions (Demarty and Bastien 2011, Li et al. 2015) and emissions from particular types of reservoirs (i.e., hydroelectric; Barros et al. 2011, Hertwich 2013) paving the way for a new synthesis of global reservoir GHG emissions. In the sections that follow, we revisit the global magnitude and controls on reservoir GHGs presented by St. Louis and colleagues (2000). This includes (a) explicit incorporation of reservoir CH_4 ebullition measurements, (b) updated global estimates of the magnitude of GHG emissions from reservoir water surfaces including the first global estimates of reservoir N_2O emissions, (c) a discussion of the environmental controls on CO_2, CH_4, and N_2O emissions

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Table 1. The global surface area and GHG flux estimates from reservoirs compared with those of other freshwater ecosystems and other anthropogenic activities.

System Type	Surface Area (x 10^6 km^2)	Annual teragrams (Tg) C or N (Tg per year)			Areal Rates (milligrams per square meter per day)			Annual CO_2 Equivalents (Tg CO_2 Eq per year)			
		CH_4-C	CO_2-C	N_2O-N	CH_4-C	CO_2-C	N_2O-N	CH_4	CO_2	N_2O	Total
All Reservoirs (This Study)	0.31[a]	13.3	36.8	0.03	120	330	0.30	606.5	134.9	31.7	773.1
All Reservoirs (Other Work)	0.51–1.5[b,c]	15—52.5[b,d]	272.7[b]	–	82–96	498	–	680–2380	1000	–	
Hydroelectric Reservoirs	0.34[e]	3–14[e,f]	48—82[e,f]	–	24–112	386–660	–	136–635	176–301	–	
Lakes	3.7–4.5[c,g,h]	53.7[d]	292[g]	–	40	216	–	2434	1071	–	
Ponds	0.15– 0.86[i]	12[i]	571[i]	–	27[i]	422[i]	–	544	2094	–	
Rivers	0.36–0.65[d,g]	1.1–20.1[d,j]	1800[g]	–	6–98[j]	7954	–	50–911	6600	–	
Wetlands	8.6–26.9[k]	106—198[k]	–	0.97[l]	15—63[k]	–	0.1–0.31	4805–8976		908	
Other Anthropogenic Emissions (2000s)	N.A.	248[m]	9200[m]	6.9[m]	–	–	–	11243	33733	6462	51438

Note: The values presented are mean estimates; the ranges of mean values are reported when there are multiple relevant models. In cases in which the areal rates are not referenced, they were derived from dividing annual teragrams (Tg) of C or N by the global surface-area estimate. The annual CO_2 equivalents were calculated by multiplying the mass-based flux (in units of Tg CH_4, CO_2 or N_2O per year) by the 100-year global warming potential of each gas (1 for CO_2, 34 for CH_4 and 298 for N_2O). [a] (Lehner et al. 2011). [b] (St. Louis et al. 2000). [c] (Downing and Duarte 2009). [d] (Bastviken et al. 2011) . [e] (Barros et al. 2011). [f] (Li and Zhang 2014). [g] (Raymond et al. 2013). [h] (Verpoorter et al. 2014). [i] (Holgerson and Raymond 2016). [j] (Stanley et al. 2016). [k] (Melton et al. 2013). [l] (Tian et al. 2015). [m] (Ciais et al. 2013).

from reservoir water surfaces, (d) a discussion of the policy implications of these new findings, and (e) recommendations regarding fruitful avenues for future research. Although this synthesis focuses on GHG emissions from reservoir water surfaces, we also describe and discuss several important alternative pathways that can contribute significantly to reservoir GHG budgets (figure 1, supplemental table S1). Given the limited number of studies characterizing these pathways, we do not include them in this global analysis, but stress the need for additional study and eventual incorporation of relevant sources in future global analyses. Finally, we stress that the GHG emissions from reservoir water surfaces synthesized here represent gross fluxes such that CO_2 and CH_4 emissions should be considered alongside estimates of reservoir carbon burial for the purposes of carbon budgeting exercises.

From a GHG-management perspective, it is crucial to understand the relative role of CO_2, CH_4, and N_2O emissions as CH_4 and N_2O are more powerful GHGs than CO_2 (34 and 298 times the global warming potential on a 100-year timescale, respectively; Myhre et al. 2013). To describe the relative contribution of various GHG emissions to global warming, emissions were converted to CO_2 equivalents, a metric that relates the radiative forcing caused by 1 mass unit of trace GHG to that caused by the emission of 1 mass unit of CO_2 over a given time span. Although CH_4 emissions from reservoirs have been implicated as a particularly important source of CO_2 equivalents (Giles 2006), constraining and modeling these fluxes is complicated by the fact that common methodological approaches, which are effective for CO_2 and N_2O emissions, do not capture an important fraction of overall CH_4 flux: bubble-based (ebullitive) CH_4 emissions. Our synthesis confirms that CH_4 emissions are responsible for the majority of the radiative forcing from reservoir water surfaces (approximately 80% over the 100-year timescale and 90% over the arguably more policy-relevant 20-year timescale) and that modeling approaches that ignore ebullitive CH_4 flux may fail to accurately quantify the magnitude of fluxes. We find that more productive, nutrient-rich reservoirs tend to emit more CH_4 than their less productive, nutrient-poor counterparts. Our global estimates support previous assertions (e.g., St. Louis et al. 2000) that GHG fluxes from reservoirs are globally important (approximately 1.3% of anthropogenic CO_2 equivalent emissions over the 100-year timespan), with CH_4 emissions from reservoir water surfaces comparable to those from rice paddies or from biomass burning. Therefore, we suggest the utility of incorporating reservoir CH_4 emissions into Intergovernmental Panel on Climate Change (IPCC) budgets.

Why methods matter

Aquatic GHG fluxes are measured using a variety of techniques (e.g., floating chambers, thin boundary methods, eddy covariance towers, acoustic methods, and funnels; supplemental figure S1) that provide varying degrees of spatial and temporal coverage and accuracy (St. Louis et al. 2000). Many commonly employed techniques for measuring aquatic GHG emissions focus on quantifying the diffusive flux of gases across the air–water interface. For CO_2 and N_2O, which are

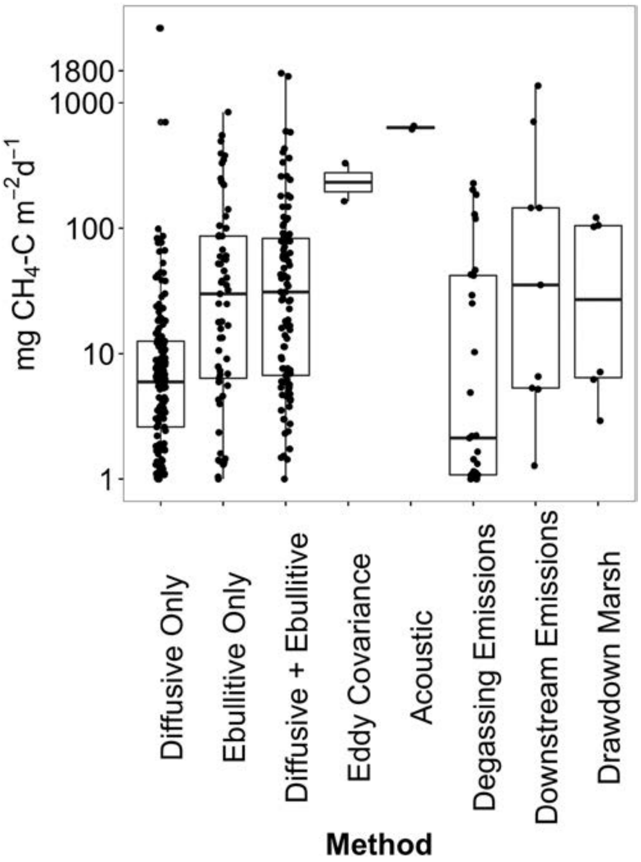

Figure 1. Areal CH_4 fluxes associated with reservoir: diffusive-only fluxes (via thin boundary layer and floating chamber with R^2 cutoff values > 0.85, n = 151), ebullitive-only fluxes (via funnels and floating chamber by subtraction, n = 58), diffusive + ebullitive fluxes (via traditional methods n = 89), total CH_4 emission via eddy covariance (n = 2), ebullitive emissions via acoustic measurements (n = 2), degassing emissions (n = 22), downstream emissions (n = 6), and drawdown marsh fluxes (n = 6, 5 from Three Gorges Reservoir). Each dot represents the mean flux from a single published paper. The lines within the boxes indicate median fluxes. The boxes demarcate the twenty-fifth and seventy-fifth percentiles; the whiskers demarcate the 95% confidence intervals.

quite soluble in water (mole fraction solubility of 7.07×10^{-4} and 5.07×10^{-4} respectively at 20°C), this is the dominant flux pathway, moving gasses to the atmosphere across the air–water interface. In contrast, CH_4 is relatively insoluble in water (mole fraction solubility of 2.81×10^{-5} at 20 °C), and is often emitted in the form of bubbles that rise directly from the sediments (Kiene 1991, Bastviken et al. 2004). Several common measurement methods do not capture ebullition (e.g., combining estimates of air–water gas exchange with measurements of dissolved GHG concentrations), whereas others may exclude ebullition events because they interfere with the linear accumulation of CH_4 within a sampling chamber (e.g., floating chambers; supplemental figure S2).

A second important challenge for accurate measurements of aquatic CH_4 ebullition is that fluxes are often highly variable in both time and space (Wik et al. 2016). Ebullition is most commonly measured using inverted funnel traps, which float beneath the surface of the water and capture bubbles as they rise through the water column. These funnel traps are typically deployed for relatively short periods of time (minutes to hours) in a relatively small number of locations (generally fewer than 10 sites per reservoir), making it difficult to capture the spatial and temporal variability of fluxes (see the Hot Spots and Hot Moments section below).

Several recent method developments improve the spatial and/or temporal resolution of CH_4 ebullition measurements in lakes and reservoirs. Modified funnel trap designs can support longer-term, temporally resolved data by (a) incorporating an airtight housing equipped with a differential pressure sensor or optical bubble size sensor for automated, high temporal resolution measurements of ebullition fluxes (Varadharajan et al. 2010, Delwiche et al. 2015), and (b) installing an electronic unit to empty the trap once it reaches full capacity so that traps don't fill faster than they can be sampled (cited in Maeck et al. 2014). Acoustic techniques can support higher spatial and temporal resolution ebullition measurements without the cumbersome and invasive field deployments associated with funnel traps. Following calibration of acoustic signal with bubble size (Ostrovsky et al. 2008), an echosounder can be mounted to a boat to estimate ebullition flux at a greater spatial resolution, or mounted to a stationary object for greater temporal resolution. Repeat daily or subdaily echosounder surveys provide a much higher degree of spatiotemporal coverage than that achieved via traditional methods, allowing for more accurate ebullitive flux estimates in survey zones (DelSontro et al. 2015). Still, echosounders are only effective within a certain depth range that depends on transducer frequency, beam angle, and survey boat speed (but generally ranges from 1 to 100 meters), provide no information about bubble CH_4 concentrations without ancillary measurements, and can also be cost prohibitive and challenging to calibrate (Ostrovsky et al. 2008, DelSontro et al. 2015). Eddy covariance techniques, which calculate GHG fluxes on the basis of mean air density and instantaneous deviations in vertical wind speed and gas concentrations, can also overcome some of the difficulty of capturing spatially and temporally variable emissions although they cannot zero in on hot spots for release unless combined with other methods. Currently, the use of eddy covariance systems over lakes and reservoirs is relatively new and poses several challenges. These challenges include (a) high instrument cost, (b) poor sensor performance during wet conditions, and (c) difficulty associated with estimating measurement footprints, especially in small, heterogeneous areas (Fassbinder et al. 2013, Peltola et al. 2013).

Of the studies compiled here, ebullition was measured in only 52% of cases in which reservoir CH_4 emissions were reported (figure 1). In the majority of cases, ebullition was measured with funnels or was lumped with diffusive flux via floating chamber measurements; however, in two studies, researchers estimated methane fluxes via eddy covariance (Eugster et al. 2011, Deshmukh et al. 2014), and in another two studies, researchers estimated ebullitive flux via acoustic methods (DelSontro et al. 2011, 2015). Mean ebullition + diffusion fluxes were over double that of diffusion-only fluxes (103 versus 43 mg CH_4-C per square meter, m^2, per day) and CH_4 fluxes varied significantly on the basis of whether or not ebullition was included (Kruskal Wallis test, $\chi^2 = 52.7$, $p < .001$; figure 1, supplemental table S2). On average ebullition contributed 65% of total diffusive + ebullitive flux ($n = 56$, standard deviation [SD] = 33.5). This is consistent with natural lakes where between 40% and 60% of CH_4 flux generally occurs via ebullition (Bastviken et al. 2004). The relative contribution of CH_4 ebullition to overall CH_4 flux was also highly variable, constituting anywhere from 0% to 99.6% of total CH_4 flux. This highlights how crucial it is to measure both types of CH_4 emission in order to estimate the total flux from reservoir surface waters. Although we did not explicitly address the temporal or spatial resolution of emission data from each system, it is notable that the few published acoustic and eddy covariance-based reservoir CH_4 flux estimates are quite high compared to the median CH_4 flux estimates from less temporally and/or spatially integrated measurement techniques (figure 1). Given the importance of CH_4 ebullition to overall CH_4 fluxes, we only use CH_4 emission estimates that incorporate both ebullition and diffusion in further sections of this article (i.e., to estimate the magnitude and controls on fluxes).

As with CH_4, many studies of CO_2 and N_2O emissions from reservoir water surfaces also suffer from low spatial and temporal resolution (therefore reducing the accuracy of emission estimates). Of the GHG estimates synthesized here, less than 25%, 3%, and 26% of temperate reservoir CH_4, CO_2, and N_2O emission estimates covered 6 months or more of the year. The majority of studies also had fewer than 10 sampling sites and measured fluxes over short periods of time (minutes to hours), often neglecting night sampling in favor of daytime measurements. A more extensive characterization of the spatial and temporal resolution of reservoir GHG sampling was beyond the scope of this analysis, but the role of sampling bias in upscaling efforts is discussed further below (see the section on Hot Spots and Hot Moments).

Patterns in areal fluxes

In total, we assembled areal CH_4, CO_2, and N_2O flux estimates from 161, 229, and 58 systems respectively, although only 75 reservoirs with CH_4 data met the methodological criteria for inclusion in our analyses (figure 2). In contrast to other recent reservoir GHG syntheses (Barros et al. 2011, Demarty and Bastien 2011, Hertwich 2013, Li et al. 2015), we include both hydroelectric and nonhydroelectric systems such as those used for flood control, irrigation, navigation, or recreation. Whereas previous synthesis efforts have lacked measurements from temperate and subtropical systems, our data set addresses this gap by including a number of recent GHG flux estimates from US, European, Australian, and Asian temperate and subtropical reservoirs (figure 2, table 2). This is important given a large number of dams that are either planned or under construction in temperate and subtropical zones (Zarfl et al. 2015). Several alternative flux pathways were not included in the areal flux estimates or the regression analysis, but are reported when available (see supplemental discussion and the Alternative Flux Pathways section below).

Here, we report mean areal (per unit surface area) CH_4 fluxes from reservoir water surfaces that are approximately 25% larger than previous estimates (120.4 mg CH_4-C per m^2 per day, SD = 286.6), CO_2 flux estimates that are approximately 30% smaller than previous estimates (329.7 mg CO_2-C per m^2 per day, SD = 447.7), and the first-ever global mean estimate of reservoir N_2O fluxes (0.30 mg N_2O-N per m^2 per day, SD = 0.9; table 1). The mean areal N_2O emissions reported here are approximately an order of magnitude less than those estimated for US reservoirs (Baron et al. 2013) and are consistent with the areal fluxes reported by Yang and colleagues (2014). 16% of reservoirs were net CO_2 sinks and 15% of reservoirs were net N_2O sinks, whereas all systems were either CH_4 neutral or CH_4 sources (figure 2). The average areal CH_4 emissions that we report from reservoirs are higher than average fluxes from natural lakes, ponds, rivers, or wetlands (table 1). On the basis of the mean areal GHG fluxes in our data set, the majority (79%) of CO_2 equivalents from reservoirs occurred as CH_4, with CO_2 and N_2O responsible for 17% and 4% of the radiative forcing, respectively, over the 100-year timespan.

The higher mean CH_4 emissions reported here are likely due to the exclusion of diffusive-only estimates and a preponderance of high CH_4 flux estimates in the recent literature. Particularly high CH_4 flux estimates have been reported for some temperate reservoirs (Maeck et al. 2013, Beaulieu et al. 2014) and subtropical reservoirs (Grinham et al. 2011, Sturm et al. 2014) that were not included in previous global estimates (St. Louis et al. 2000, Barros et al. 2011, Bastviken et al. 2011), indicating that midlatitude reservoirs can emit as much CH_4 as tropical systems. In fact, we found that CH_4 fluxes from Amazonian reservoirs were statistically indistinguishable from reservoir CH_4 fluxes in other regions (Mann Whitney test, $p = 0.25$; supplemental figure S3). These findings run counter to the common view that low latitude reservoirs (and Amazonian reservoirs in particular) support greater CH_4 emission rates than temperate systems (Barros et al. 2011), but are consistent with the recent influx of higher emission estimates from subtropical and temperate ecosystems mentioned above.

Previous efforts to identify predictors of reservoir GHGs

Reservoir age (Barros et al. 2011, UNESCO–IHA 2012, Hertwich 2013) and latitude (Barros et al. 2011) have been suggested as predictors of CO_2 and CH_4 flux from hydroelectric reservoirs. Elevated GHG emissions from young (less than 10 years) reservoirs are commonly observed

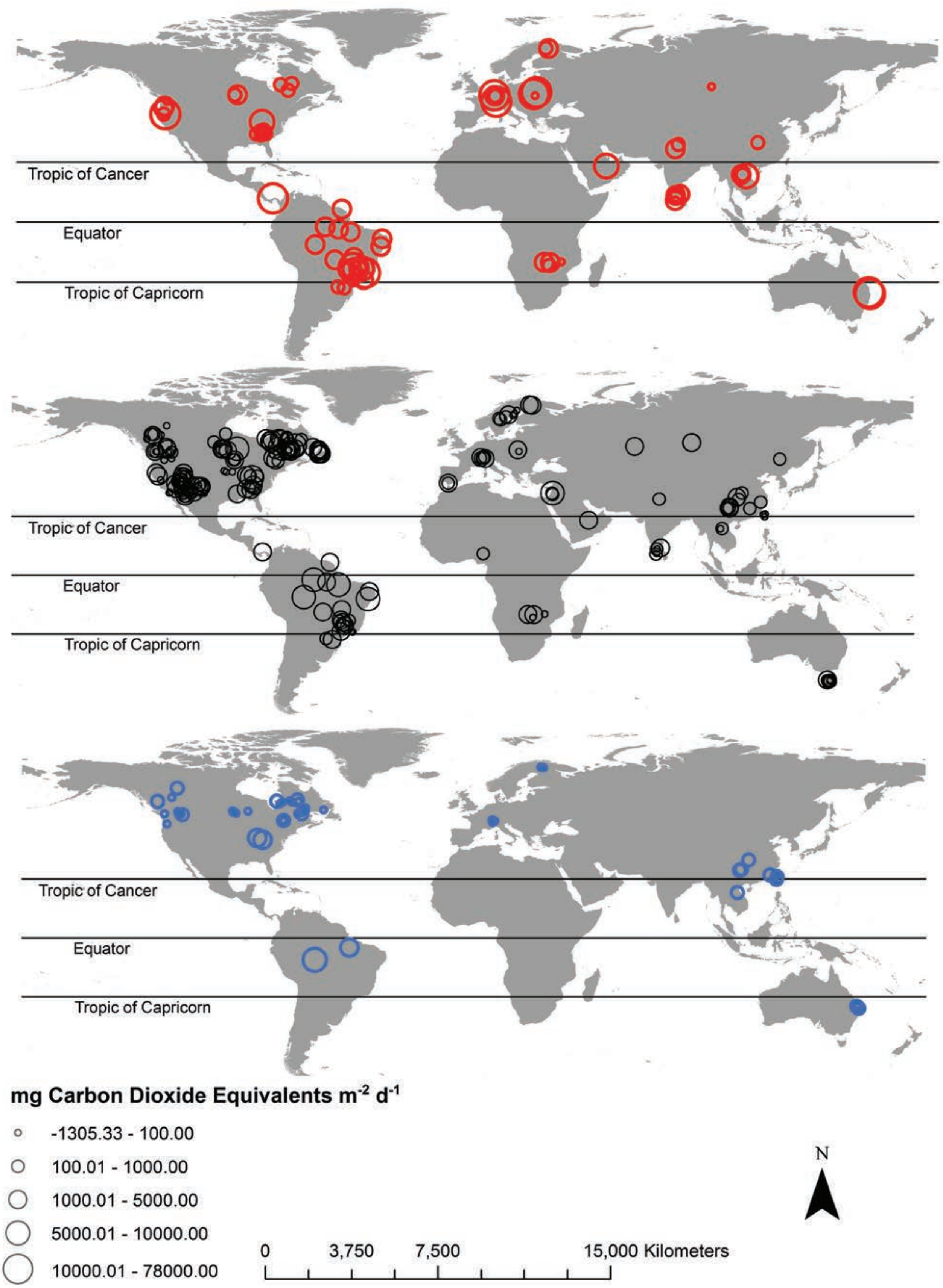

Figure 2. Diffusive + ebullitive methane (top), carbon dioxide (middle), and nitrous oxide (bottom) emissions from reservoirs on a CO_2-equivalent basis (100-year horizon). Few reservoirs had measurements for all three gases.

Table 2. The number of reservoirs with surface water GHG emission estimates by continent, as well as a break down of the number of CO_2, ebullitive + diffusive (E+D) CH_4, diffusive only (D) CH_4, and N_2O emission estimates by continent.

Continent	CO_2	CH_4 (E +D)	CH_4 (D)	N_2O	Total number of reservoirs with any GHG emission estimates
North America	144	23	56	37	158
South America	22	21	1	2	23
Africa	5	4	0	0	5
Europe	18	11	10	7	31
Asia	30	14	6	8	36
Australia	10	2	12	4	14
World	**229**	**75**	**85**	**58**	**267**

(Abril et al. 2005, Bastien et al. 2011, Teodoru et al. 2012) and are thought to be due to rapid decomposition of the most labile terrestrial organic matter, although some reservoirs may continue to have elevated GHG emissions at least 20 years after flooding (Kemenes et al. 2011). Measurements in an oligotrophic system in Canada's boreal zone have shown that heterogeneity in preflood carbon stocks can affect young reservoir CO_2 fluxes, with greater rates of sediment CO_2 production in higher carbon sediments (Brothers et al. 2012). However, the experimental flooding of high, medium, and low carbon boreal forests yielded no discernible relationship between the soil or sediment carbon stock and GHG production over a 3-year time span (Hendzel et al. 2005, Matthews et al. 2005). Reservoir GHG emissions can also be positively correlated with temperature (DelSontro et al. 2010, UNESCO–IHA 2012). Consequently, the negative correlation between latitude and hydroelectric GHG emissions reported in previous work could reflect higher average water temperatures at low latitudes. In addition, lower latitude regions typically experience higher rates of terrestrial net primary production (NPP), a factor that has been positively correlated with GHG emissions from hydroelectric reservoirs (Hertwich 2013). High rates of NPP may promote enhanced leaching of dissolved organic matter (DOM), fueling additional decomposition of terrestrial organic matter within tropical reservoirs.

A growing body of work highlights the role that nutrient status and associated primary productivity may play in determining overall reservoir GHG dynamics. For example, Li and colleagues (2015) reported a negative correlation between both nutrient enrichment and primary production and CO_2 fluxes, and at least one study has argued that increasing primary production can shift lentic ecosystems from CO_2 sources to sinks (Pacheco et al. 2013). This occurs when additional nutrients promote atmospheric carbon sequestration via enhanced photosynthesis leading to accelerated rates of organic carbon sedimentation and burial. At the same time, eutrophication may promote larger CH_4 emissions, both by reducing O_2 concentrations in reservoir bottom waters and by increasing organic matter quantity (as described below). In wetland ecosystems, NPP has been posited as a "master variable" that integrates several important environmental factors influencing CH_4 emission (Whiting and Chanton 1993). Some of these factors are likely to be more important in wetlands than in reservoirs (i.e., rooted plants as conduits for CH_4 exchange), whereas others are applicable across systems (i.e., increased substrate availability associated with elevated rates of carbon fixation). Regionally, positive correlations between chlorophyll *a* concentrations and both dissolved CH_4 concentrations (Indian reservoirs; Narvenkar et al. 2013) and CH_4 fluxes (north temperate lakes; West et al. 2015a) have been found in lakes and reservoirs. Although less is known about the controls on reservoir N_2O flux, strong positive correlations between NO_3^- concentrations and both N_2O concentration and flux have been observed across aquatic ecosystems (Baulch et al. 2011, McCrackin and Elser 2011).

Overall, better predictive tools are needed for identifying environmental controls on reservoir GHGs. Some progress has been made toward accomplishing these tasks through the modeling of hydroelectric CO_2 and CH_4 emissions (Barros et al. 2011, IEA Hydropower 2012, UNESCO–IHA 2012, Hertwich 2013). Still, we are not aware of any modeling efforts that have explicitly incorporated ebullition; instead, existing efforts have used either diffusive-only emissions or a combination of diffusive-only and ebullitive + diffusive emissions. In the section that follows, we explicitly consider ebullition by categorizing CH_4 fluxes on the basis of collection methods and considering the extent to which environmental controls differed on the basis of CH_4 flux pathway (ebullitive versus diffusive). In particular, we explore the hypothesis that nutrient loading and the resulting increase in primary production stimulates GHG emissions from reservoir water surfaces, primarily via enhanced CH_4 production.

Synthesis findings: Productivity predicts the radiative forcing capacity of reservoir GHG emissions

We collated system characteristics likely to covary with, or control, GHG fluxes. These characteristics included

Table 3. The least squared regression statistics for a subset of the best models relating reservoir CO_2, CH_4, and N_2O fluxes to potential predictor variables. All the significant linear regressions ($p < .05$) with $R^2 > 0.1$ are shown. Sign indicates whether the slope of the regression line was positive (+) or negative (–). Note that reservoir CO_2 fluxes are inverse transformed such that a negative regression correlation indicates a positive relationship between the predictor variable and the CO_2 flux. * Indicates modeled predictor. Complete regression statistics can be found in supplemental tables S4 and S5.

Gas	Predictor	Transformation	df	*p* value	R^2	Sign
Ln (CH_4 + 1) Bubble + Diffusion *n* = 75	[Chlorophyll *a*]	Ln	29	<.001	0.50	+
	[DIP]*	Ln	31	0.02	0.18	+
1/(CO_2 + 1000) *n* = 229	Mean Annual Precipitation	none	31	0.04	0.11	–
Ln (N_2O + 1) *n* = 58	[NO_3^-]	Ln	16	<.001	0.49	+
	Latitude	none	56	<.001	0.46	–
	Mean Annual Air Temperature*	Sqrt	55	<.001	0.33	+
	Mean Annual Precipitation*	Sqrt	54	<.001	0.30	+

morphometric, geographic, and historical properties of study reservoirs (i.e., depth, residence time, volume, surface area, age, and latitude), biologically significant water column solute concentrations (i.e., NO_3^-, total phosphorus, and dissolved organic carbon), and metrics of ecosystem primary productivity (i.e., trophic status and mean or modeled surface water chlorophyll *a* concentrations; see the supplemental materials for a complete list of the tested variables).

Of the factors examined, CH_4 emissions were best predicted by chlorophyll *a* concentrations (positive correlation, $p < 0.001$, $R^2 = 0.50$, $n = 31$); CO_2 emissions were best predicted by reported mean annual precipitation (positive correlation, $p = 0.04$, $R^2 = 0.11$, $n = 33$); and N_2O emissions were most strongly related to reservoir NO_3^- concentrations (positive correlation, $p < 0.001$, $R^2 = 0.49$, $n = 18$, table 3, supplemental figure S6). Although latitude was also a strong predictor of N_2O flux ($p < 0.001$, $R^2 = 0.47$, $n = 55$), latitude and NO_3^- were weak covariates (–0.29 Pearson correlation), and latitude was not a significant predictor of N_2O ($p = 0.10$) in a multiple linear regression model with NO_3 ($p = 0.01$). CH_4 emissions were only weakly related to latitude ($p = 0.05$, $R^2 = 0.04$), and CO_2 emissions were not significantly related to latitude. Whereas CO_2 emissions were weakly related to reservoir age ($p = 0.003$, $R^2 = 0.04$), CH_4 and N_2O fluxes were not (supplemental table S4). The positive, albeit weak, relationship between CO_2 fluxes and mean annual precipitation is consistent with observations in boreal lakes where precipitation has been observed to flush terrestrial carbon into surface waters and enhance CO_2 concentrations and emissions via organic matter degradation (Rantakari and Kortelainen 2005). The relationship between N_2O fluxes and NO_3^- concentrations is consistent with observations from small streams (Baulch et al. 2011) as well as observed positive relationships between concentrations of N_2O and NO_3^- in reservoirs (Beaulieu et al. 2015) and in lakes receiving atmospheric nitrogen deposition (McCrackin and Elser 2011).

The controls on reservoir CH_4 flux deserve particular attention because our analysis suggests that CH_4 emissions are responsible for 79% of the radiative forcing from reservoirs over the 100 year timespan. Chlorophyll *a* and air temperature were significant predictors of CH_4 emissions from reservoir water surfaces regardless of flux type (i.e., diffusive only, ebullitive only, diffusive + ebullitive; supplemental tables S4 and S5). Mean reservoir depth and chlorophyll *a*, both of which have been reported to control lake and reservoir CH_4 emissions, were weakly correlated in this analysis (Pearson correlation 0.46). Depth was not a significant predictor of CH_4 flux for the whole data set ($p = 0.14$, $R^2 = 0.02$) or for the subset of the data for which chlorophyll *a* concentrations were available ($p = 0.19$, $R^2 = 0.02$), indicating that chlorophyll *a* is a better predictor of system-wide CH_4 emissions than mean depth. Depth has been found to exert an important control on the spatial variability of CH_4 fluxes from lakes, particularly with respect to ebullition (Bastviken et al. 2004, West et al. 2015a). In the marine environment, ebullition-based emissions to the atmosphere are thought to be negligible in waters deeper than 100 meters because of the dissolution of bubbles en route from sediments to the atmosphere (McGinnis et al. 2006), and a recent study of north temperate lakes reported that ebullition rarely occurred at sites deeper than 6 meters (West et al. 2015a). Although both depth and age (discussed above) may be important predictors of carbon emissions in individual reservoir systems,

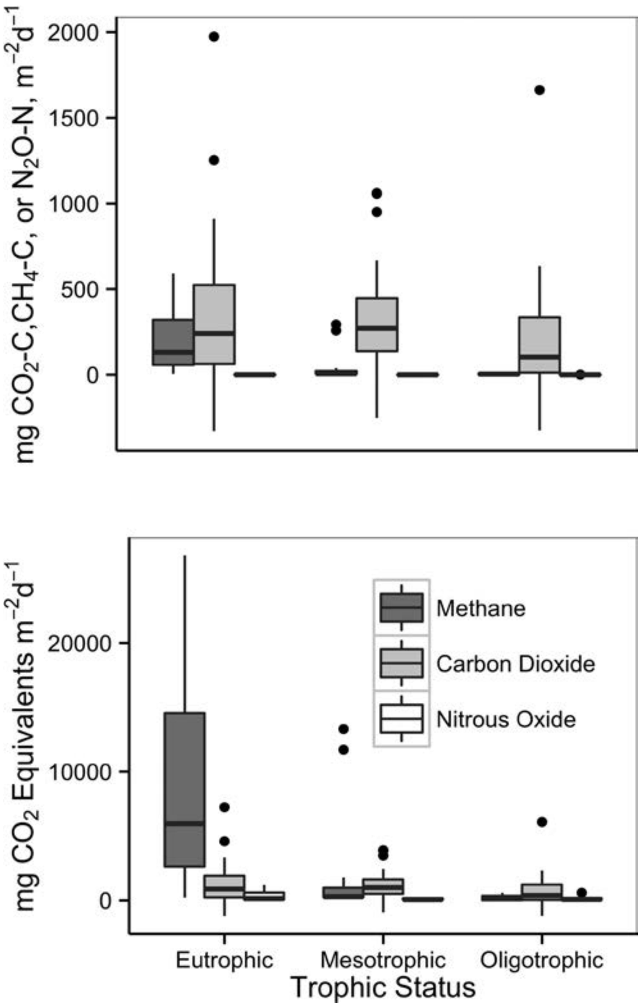

Figure 3. Average reservoir GHG fluxes by trophic status. The top panel shows areal flux rates; the bottom panel shows fluxes converted to CO_2 equivalents. The legend is for both panels. The lines within the boxes indicate median fluxes. The boxes demarcate the twenty-fifth and seventy-fifth percentiles, the whiskers demarcate the 95% confidence intervals, and the dots plot data outside this range. One methane flux from a eutrophic reservoir is removed from the bottom panel (78,000 milligrams CO_2 equivalents per m^2 per day from the Rsezów Reservoir) to improve readability.

these relationships do not appear to scale up in the global model, which only considers mean values for individual reservoirs (e.g., mean reservoir depth or the mean age of the reservoir when carbon emissions were measured).

The strong positive correlation between reservoir CH_4 flux and chlorophyll *a* is also reflected in the significantly different CH_4 emissions found in systems of different trophic statuses (Kruskal Wallis test, $\chi^2 = 16.8$, $p < .001$). Specifically, eutrophic systems emitted approximately an order of magnitude more CH_4 than oligotrophic ones (figure 3). This pattern has been observed regionally in North American, Swedish, and Canadian lakes (Bastviken et al. 2004, Rasilo et al. 2015, West et al. 2015a) as well as Finnish lakes and reservoirs (Huttunen et al. 2003), and is consistent with recent findings from shallow lake mesocosms where CH_4 emissions were best predicted by factors related to primary production (i.e., nutrient concentrations and primary producer abundance; Davidson et al. 2015). This suggests that the low oxygen and high dissolved organic carbon conditions that often develop in eutrophic systems promote elevated CH_4 production relative to lower nutrient systems. In addition to increasing the quantity of organic carbon and reducing the availability of oxygen, eutrophication may also affect the overall quality of organic matter for fueling CH_4-producing archaea. Algae-derived organic matter has been found to fuel higher rates of CH_4 production than land-based “terrestrial” carbon (West et al. 2012), and may even stimulate the enhanced incorporation of recalcitrant terrestrial carbon into bacterial biomass (i.e., priming effect; Guillemette et al. 2015). Thus, increasingly high fractions of algae-derived organic matter will likely support more methane production.

Global surface area of reservoirs

Global-scale estimates of reservoir GHG emissions are dependent on estimates of both areal fluxes (discussed above) and global reservoir surface area. There have been a number of recent efforts to improve global reservoir (and lake) surface-area estimates (Downing and Duarte 2009, Lehner et al. 2011, Verpoorter et al. 2014). Although St. Louis and colleagues (2000) estimated global reservoir surface area by multiplying the surface area of reservoirs in the *World Register of Dams* by a factor of four, more recent reservoir surface-area estimates were made assuming that reservoir surface areas follow a pareto distribution (Downing et al. 2006, Lehner et al. 2011). Downing and colleagues (2006) used data from the International Commission on Large Dams together with pareto-based extrapolations to estimate that reservoirs more than 0.01 square kilometers (km^2) cover 258,570 km^2 of the earth’s surface. Following this, Lehner and colleagues (2011) used the Global Reservoir and Dam Database (GRAND) together with pareto-based extrapolations to estimate that reservoirs more than 0.00001 km^2 cover 507,102 km^2 of earth’s surface. These reservoir surface-area estimates are one-sixth to one-third the value used by St. Louis and colleagues (2000). For our best estimate of global reservoir GHG fluxes, we use 305,723 km^2 of reservoir surface area (table 1). This estimate is based on GRAND and excludes the original surface area of natural lakes that have been modified with water regulation structures (this includes Lakes Victoria, Baikal, and Ontario; Lehner et al. 2011). The 267 reservoirs whose CO_2, CH_4, and/or N_2O emission estimates we synthesize here cover a collective surface area of over 77,287 km^2 (28 reservoirs with unknown surface area), and therefore represent 25% of global reservoir coverage.

In addition, reservoir surface area is likely to increase substantially in coming decades given the 847 large hydropower projects (more than 100 MW) and 2853 smaller projects (more than 1 MW) that are currently planned or under construction (Zarfl et al. 2015). In this synthesis, reservoirs with more than 1MW installed capacity had a median surface area of 226 km^2. Assuming each of the 847 large hydropower projects that are planned or under construction has an equivalent surface area, this would constitute 225,691 km^2 of additional reservoir surface area, nearly doubling current reservoir surface-area estimates. Although there is a net trend toward dam decommissioning in the United States, most of these removals have been small dams, and the global number of removals is more than offset by recent increases in dam construction (O'Connor et al. 2015).

Global magnitude of reservoir GHG emissions

We report global GHG emissions from reservoir water surfaces on the low end of previously published values (table 1), but stress that these emissions still contribute significantly to global budgets of anthropogenic CO_2 equivalent emissions. CH_4 constituted the majority of CO_2 equivalent emissions from reservoirs, and the per area reservoir CH_4 fluxes reported in this synthesis are higher than per area fluxes for any other aquatic ecosystem (table 1). We estimate that reservoirs emit 13.4 Tg CH_4-C per year (5th and 95th confidence interval: 8.9–22.2 Tg CH_4-C per year), 36.8 Tg CO_2-C per year (5th and 95th confidence interval: 31.8–42.8 Tg CO_2-C per year), and 0.03 Tg N_2O-N per year (5th and 95th confidence interval: 0.02–0.07 Tg N_2O-N per year; table 1). The estimate of global reservoir GHG emissions presented here is calculated on the basis of the product of bootstrapped estimates of mean areal GHG fluxes and best estimates of global reservoir surface area (as was done in a recent estimate of global methane emissions from streams and rivers, Stanley et al. 2016). See the supplemental materials for information about the bootstrapping technique used. Given the dominant controls on GHG emissions from reservoir water surfaces identified in this study and given the current availability of relevant predictor variables at the global scale, we do not see an advantage to segmenting our upscaling efforts at this point in time. Still, identifying regional differences in reservoir GHG emissions remains a needed area of future research (see below section on Uncertainties and Future Research Directions).

Although the global mapping of reservoir trophic status (and associated upscaling of CH_4 emissions) is beyond the scope of this article, recent progress in the mapping of chlorophyll *a* in medium and large-sized lakes and reservoirs shows that about 60% of systems have more than 10 micrograms per liter chlorophyll *a* (Sayers et al. 2015), and would therefore be considered eutrophic by most classification schemes (Cunha et al. 2013). Similarly, a comparison of large reservoir locations (Lehner et al. 2011) with model-predicted dissolved inorganic phosphorus (DIP) yields (Harrison et al. 2010) indicates that most large reservoirs occur in phosphorus enriched regions (figure 4a) that may promote eutrophication of reservoirs. To illustrate, the average DIP yield (per 0.5 degree grid cell) in grid cells with dams is over threefold higher than the global average DIP yield (45 versus 13 kilograms P per km^2 per year). Given this pattern and the high fraction of nutrient enriched, productive reservoirs in our GHG database (of systems where trophic status data were available, 38% and 24% were eutrophic and mesotrophic respectively), it is likely that a large fraction of reservoirs are highly productive and therefore support high CH_4 emission rates. However, overlaying a map of the hydroelectric projects that are currently planned or under construction (Zarfl et al. 2015) on a map of average DIP yield (Harrison et al. 2010) suggests that newer hydroelectric projects will be more evenly distributed between phosphorus enriched and relatively phosphorus poor regions (Figure 4b). Further research is needed to better understand how much P will be *routed through* current and future reservoirs to support large-scale models of reservoir trophic status and associated CH_4 emissions. Specifically, models of riverine DIP yield would need to be downscaled to quantify how much DIP individual reservoirs are intercepting.

Emissions from alternative flux pathways

There are several emission pathways that are either nonexistent or of marginal importance in natural lakes, but that may contribute significantly to reservoir GHG budgets. These include: drawdown emissions, downstream emissions, emissions from decomposing wood, and emissions from dam spillways and turbines (e.g., "degassing" emissions). Drawdown emissions occur when fluctuating water levels cause large changes in hydrostatic pressure and create sediments that are periodically inundated with water and then exposed to the atmosphere. Although all aquatic systems experience natural fluctuations in water level, the amplitude and/or frequency of these fluctuations is likely more pronounced in reservoir ecosystems (Zohary and Ostrovsky 2011). Drawdown zones (that are periodically dry and then inundated) may contribute disproportionately to systemwide GHG emissions because of the shifting redox conditions they experience (Lu et al. 2011, Yang M et al. 2014). Drawdown may also be a hot moment for systemwide CH_4 release because reductions in hydrostatic pressure can stimulate ebullition events (Maeck et al. 2014). These events may constitute significant components of annual reservoir-wide CH_4 emission budgets and are the subject of ongoing work, but are not included in the analyses presented here. Degassing emissions from turbines and spillways occur when reservoir water undergoes rapid depressurization and/or aeration resulting in rapid emission of dissolved gasses. GHGs that remain in solution after water passes through a dam either diffuse into the atmosphere or are consumed by microbes (e.g., methane oxidation) downstream of the dam. Downstream emissions refer to GHGs that are produced within the reservoir and emitted from the river channel below a dam. The spatial footprint of these emissions is generally defined as the river reach for which GHG emissions are elevated above background (Kemenes et al. 2007). Finally, the decomposition of standing woody material was found to constitute a large fraction

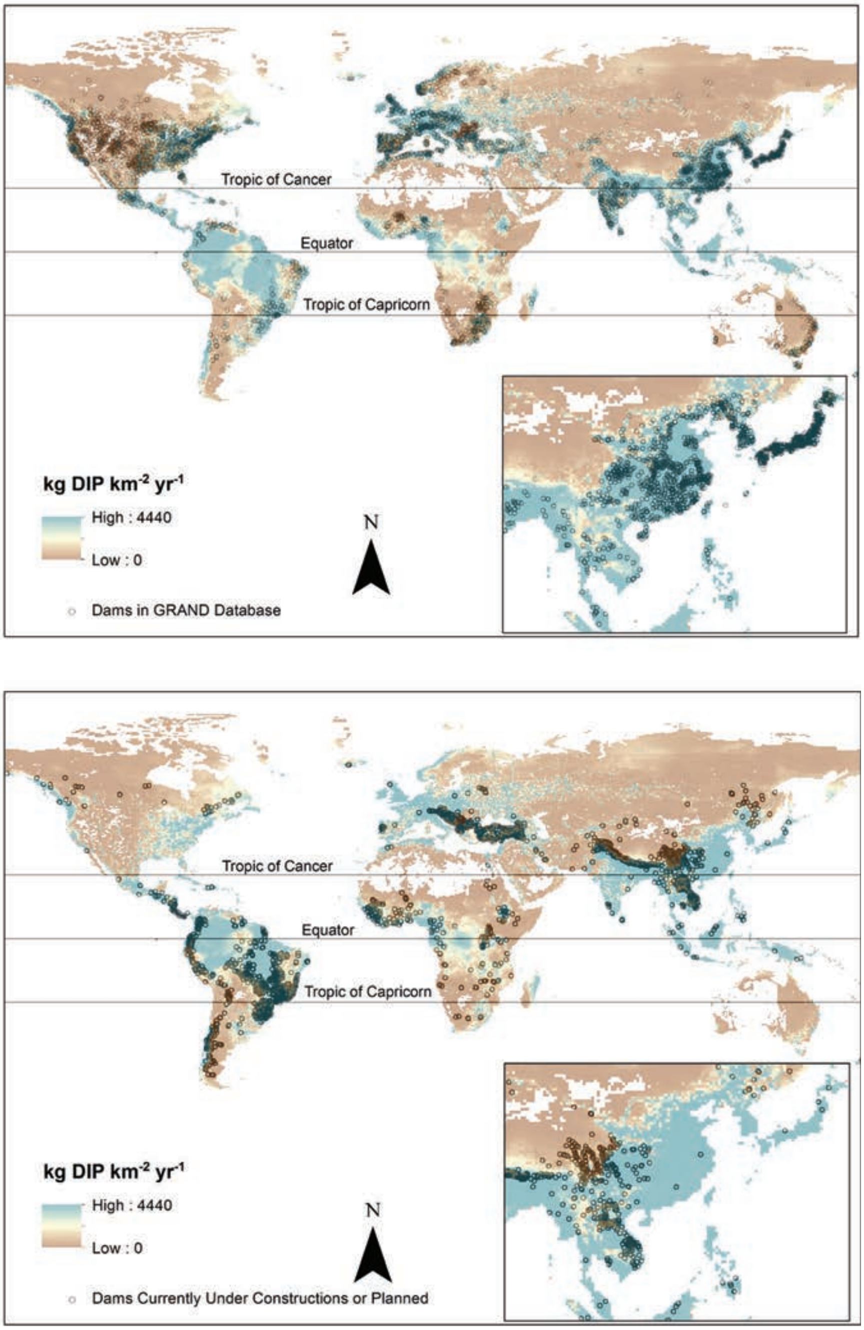

Figure 4. Global NEWS half-degree dissolved inorganic phosphorus (DIP) yield (Harrison et al. 2010) overlaid on existing reservoirs from the GRAND database (Lehner et al. 2011) and hydropower reservoirs currently either under construction or planned (bottom panel; Zarfl et al. 2015). The inset maps show Southeast Asia, a region of rapid dam construction. China is projected to remain the global leader in hydropower dam construction, producing approximately one-fourth of global hydropower (Zarfl et al. 2015).

of total GHG emissions in a tropical reservoir (26%–45% of CO_2 equivalents over a 100-year time frame; Abril et al. 2013), but this GHG source remains to be studied in reservoirs from other regions.

Both downstream and degassing emissions are likely highly dependent on reservoir GHG concentrations, dam engineering, spill practices and downstream biogeochemistry. Larger degassing and downstream emissions are expected when the spilled reservoir water is high in GHGs (Guérin et al. 2006). This generally occurs in systems in which the water is withdrawn from the lower portion of the reservoir (hypolimnetic release), because this water is typically highly pressurized and is also enriched in GHGs relative to surface waters (Kemenes et al. 2007). These emissions may also depend on dam-specific engineering. For example, an aerating weir at Petit Saut reservoir was installed to optimize CH_4 degassing immediately below the dam to avoid problems associated with methane-oxidation-induced hypoxia (Abril et al. 2005). Finally, the environmental controls on methane consumption (e.g., methane oxidation; Abril et al. 2005, Kemenes et al. 2007) and air–water gas exchange rates downstream of a dam may also play an important role in determining the magnitude of downstream emissions.

Measurements of GHG emissions from drawdown zones, downstream river reaches, wood decomposition, as well as spillways and turbines are currently too limited and/or too poorly constrained to meaningfully include in analyses of the controls and magnitude of reservoir GHGs. Still, these pathways may contribute significantly to overall ecosystem fluxes, particularly in the case of CH_4 (figure 1). For a more detailed summary of reservoir GHG fluxes via alternative flux pathways, see the supplemental discussion and table S1.

Uncertainties and future research directions

In developing this synthesis, we identified a number of areas that are beyond the scope of our analysis but that certainly deserve additional attention and research. Although the spatial coverage of GHG flux measurements has improved in recent years, there are still few measurements from many regions, including Africa, Australia, and Russia (table 2, figure 2). With respect to the forms of GHGs measured, there are currently threefold and fourfold more reservoirs with CO_2

emission estimates than for CH_4 (ebullitive + diffusive) and N_2O emissions, respectively (table 2). In addition, there is a crucial need to better constrain GHG emissions from boreal reservoirs, especially the relative role of diffusive versus ebullitive CH_4 emission pathways. The roles of reservoir typology, spatiotemporal variability, and ecosystem productivity in determining GHG emissions all deserve further analysis. In the sections that follow, we highlight some significant research needs that will improve our ability to model and potentially manage reservoir GHG emissions.

Reservoir typology. Currently, there are relatively few GHG flux estimates from nonhydroelectric systems. Although hydroelectric dams are estimated to constitute 30%–62% of global impoundments (Lehner et al. 2011, Varis et al. 2012), 82% of reservoirs with known uses in our GHG database had the capacity to generate hydroelectricity (supplemental figure S7). Although we did not detect any significant difference between the areal emission of CH_4, CO_2, or N_2O from hydroelectric versus nonhydroelectric systems (Mann Whitney test, $p = .83$, .27, and .87 respectively; figure S3), we also did not consider degassing, downstream, or drawdown zone emissions, all of which are likely to vary on the basis of reservoir typology. Better characterization of reservoir outlet structure (e.g., proportion of surface versus bottom water withdrawals by reservoir type) and associated turbine and downstream GHG emissions would aid our understanding of how different types of reservoirs (hydroelectric, flood control, irrigation etc.) contribute to overall GHG emissions. In addition, small farm impoundments were not included in this data set because of lack of data, but these systems clearly deserve more attention because they are often located in eutrophied areas and are disproportionately active with respect to carbon cycling (Downing et al. 2008). In fact, natural ponds less than 0.001 km^2 are estimated to make up less than 10% of global lake and pond surface area but constitute more than 15% of CO_2 emissions and more than 40% of diffusive CH_4 emissions (Holgerson and Raymond 2016).

Hot spots and hot moments. Lake and reservoir GHG emissions are often highly variable in both space and time. The flux estimates presented in previous sections use available estimates from every reservoir where GHG emissions have been reported (and mean estimates from reservoirs where multiple studies or years of data have been collected), but it is important to note that the spatial and temporal coverage of these emission estimates are highly variable. Reservoir GHG emission estimates are often made at temporal scales ranging from minutes to hours even though lake and reservoir GHG emissions can vary over single day–night cycles (Morales-Pineda et al. 2014, Podgrajsek et al. 2014, 2015), seasonally with changes in productivity and/or river inflow (Knoll et al. 2013, Morales-Pineda et al. 2014, Pacheco et al. 2015), and episodically because of water-level fluctuations (Maeck et al. 2014) or water-column mixing dynamics (Jammet et al. 2015). The spatial coverage of reservoir GHG emission measurements is also often limited; many studies measure emissions at fewer than 5 sites and very few studies have more than 10 sites. Recent spatial analyses of reservoir CH_4 dynamics highlight the disproportionate importance of inlets and other depositional zones toward overall flux (DelSontro et al. 2011, Maeck et al. 2013) as well as from seasonally flooded and downstream zones (see the Emissions From Alternative Flux Pathways section).

Despite the considerable uncertainty associated with the reservoir-specific GHG emission estimates synthesized here, we argue that these data provide a low-end estimate of global emissions. A recent study quantified the effects of spatial and temporal sampling resolution on diffusive and ebullitive CH_4 emission estimates from 3 shallow boreal lakes and found that low sampling coverage is more likely to lead to underestimates of flux than overestimates (72% chance of flux underestimation when bubble trap sampling is limited to 1–3 days; Wik et al. 2016). The authors estimate that diffusive and ebullitive CH_4 fluxes should be measured from a minimum of 3 and 11 depth stratified sites on at least 11 and 39 days (respectively) to achieve ±20% of the emissions estimated from sampling more intensively (Wik et al. 2016). More work is needed to characterize sampling bias in other types of systems, and to understand how sampling bias scales up. In this analysis, we treated system-specific estimates of GHG flux equally despite a large range in the degree of sampling effort represented by each study.

The development of methods and protocols that effectively capture spatial and temporal variation in GHG fluxes is crucial for improving our ability to compare “apples to apples” between different reservoir systems. Efforts are already being made in this direction (UNESCO–IHA 2010, Bastviken 2015).

Seasonality and ice cover. The seasonality of reservoir GHGs is a major frontier. Future research should aim to quantify both seasonal patterns in emission and the extent to which water-column mixing and other short-term events contribute to annual-scale GHG emissions. Although warmer temperatures have been correlated with higher rates of CH_4 production across a range of ecosystems (Yvon-Durocher et al. 2014), annual-scale reservoir GHG data are currently too limited to make inferences on how seasonal biases may either under or overestimate annual-scale fluxes. Spring (ice melt) and fall (destratification) turnover events can result in pulse emissions wherein gasses that have accumulated under the ice or thermocline are suddenly mixed upward and vented to the atmosphere as a lake circulates. Although turnover data from reservoir systems is extremely sparse (but see Bastien et al. 2011, Demarty et al. 2011, Beaulieu et al. 2014), in lakes, turnover flux may account for an average of 35% (and a range of less than 1% to 70%) of annual CH_4 emissions, with the highest contribution from small systems (Michmerhuizen et al. 1996, Bastviken et al. 2004, Jammet et al. 2015).

Currently, the role of CH_4 oxidation (a microbial process that consumes methane) in mediating atmospheric CH_4

fluxes during lake turnover events is also not well understood. Commonly employed methods for estimating turnover flux use hypolimnion storage (i.e., the gasses that have accumulated under the ice or thermocline) to estimate emissions and assume that there is no significant CH_4 oxidation during turnover (Michmerhuizen et al. 1996). Research in boreal and temperate lakes has found that anywhere between 60 and 94% of the CH_4 stored in the water column can be oxidized during turnover (Rudd and Hamilton 1978, Utsumi et al. 1998) but the environmental controls on turnover-related methane oxidation rates are not well known. Given current uncertainties, our global-scale estimate of reservoir GHG flux does not account for ice cover, but see the supplemental materials for an estimate of the extent to which ice cover could reduce annual-scale emissions (assuming no turnover emissions).

The role of boreal systems. Results from this synthesis suggest that biases in the application of different measurement techniques have led to spurious assignment of age as a significant control on reservoir CH_4 fluxes. In addition, this sampling bias may have overemphasized the significance of latitude as a predictor of CH_4 fluxes. The majority of measurements from old systems and high latitude systems have been diffusive only (supplemental figures S4 and S5), which may underestimate true CH_4 fluxes. It is possible, however, that ebullition is limited in boreal systems. Large-scale monitoring efforts in Canadian hydroelectric reservoirs suggests that CH_4 bubbling constitutes less than 5% of total emissions in many boreal systems (Tremblay pers. comm.). Still, we are aware of only a handful of published studies that report both diffusive and ebullitive emissions from boreal systems, and the fraction of bubbling in these systems covers a broad range (0%–20% in Eastmain reservoir, 18% in Porttipahta reservoir, 61% to 75% in Canadian experimental reservoirs, and 87% in Lokka reservoir [Huttunen et al. 2002, Matthews et al. 2005, Teodoru et al. 2012]). Unfortunately, CH_4 flux measurements from permafrost reservoirs and nonhydroelectric boreal reservoirs are currently lacking. Future study of boreal reservoir GHG fluxes should target these underrepresented systems and incorporate more comprehensive ebullition rate measurements.

The role of reservoir productivity. Recent work has suggested that eutrophication might “reverse” the carbon budget of lakes and reservoirs (i.e., shifting the ecosystem from net heterotrophy to net autotrophy) by converting large amounts of CO_2 to organic matter via elevated primary production (Pacheco et al. 2013). Our analysis does not support this idea. A comparison of CO_2 and CH_4 fluxes from eutrophic reservoirs suggests that eutrophication does little to change the net carbon balance of reservoirs, but greatly increases the atmospheric radiative forcing caused by these systems through the stimulation of CH_4 production (figure 3). This suggests a potential positive feedback loop wherein a warming climate supports larger algal populations, larger algal populations provide more organic matter to support more methane production, and a portion of the methane produced escapes to the atmosphere, where it functions to further warm climate. The relationship between organic matter quality and methane production is an active area of research that may reduce the strength, or possibly even negate, the feedback loop proposed above. A recent laboratory study revealed that algal biomass quality, in terms of lipid content, enhanced rates of methane production (West et al. 2015b). Because algae grown under nutrient rich conditions tend to be relatively lipid poor, the authors posit that this resource quality feedback reduces the strength of the positive feedback between eutrophication and methane production (West et al. 2015b). Developing our understanding of these feedbacks should help inform quantitative modeling efforts.

The larger context. In this study, we have discussed only gross carbon emissions from existing reservoirs, ignoring other stages or factors of a reservoir’s carbon cycle that are important to consider. For example, it will be necessary to eventually place gross fluxes in context by comparing them with (a) the GHG balance of the land prior to flooding, (b) the rates of reservoir carbon fixation and storage, (c) the GHGs associated with reservoir creation and decommissioning (e.g., life-cycle-analysis perspective), and (d) the long-term fate of carbon buried in reservoirs that are decommissioned. Few studies have placed reservoir GHG emissions into such a context, but those that have find that reservoirs result in a net carbon footprint that exceeds that of the preflooded landscape and that they are net emitters of CO_2 equivalents (Jacinthe et al. 2012, Teodoru et al. 2012, Faria et al. 2015). A recent analysis of CH_4 fluxes from hydroelectric reservoirs showed that 10% of reservoirs have emission factors (gCO_2e per kilowatt hour) larger than the CO_2 emissions from natural gas combined cycle plants (Hertwich 2013), although the authors did not consider carbon burial offsets. Although dams are responsible for high rates of carbon burial (Clow et al. 2015), it has been argued that at least a portion of this burial would still be occurring farther downstream, perhaps even in coastal waters, in the absence of dams (Mendonça et al. 2012). The role of dams in re-locating sediment carbon pools may be significant in determining total carbon burial (Mendonça et al. 2012) as well as the fraction of carbon that is emitted as CH_4. For example, faster-moving, more oxygenated “lotic” waters typically support more rapid decomposition and CO_2 production but less CH_4 production. Similarly, at the coast, high concentrations of SO_4^{2-} generally prohibit high CH_4 emissions. Accounting for the short and long-term fate of carbon in reservoir sediments is an important next step in global carbon budgeting exercises.

Policy implications

When CH_4, CO_2, and N_2O emissions are combined, our synthesis suggests that reservoir water surfaces contribute 0.8 Pg CO_2 equivalents per year over a 100-year time span (fifth and ninety-fifth confidence interval: 0.5–1.2 Pg CO_2 equivalents per year), or approximately 1.5% of the global anthropogenic CO_2-equivalent emissions from CO_2, CH_4, and N_2O reported by the IPCC (table 1; Ciais et al. 2013) and 1.3% of global anthropogenic CO_2-equivalent emissions from well mixed GHGs overall (Myhre et al. 2013). Therefore, we argue for inclusion of GHG fluxes from reservoir surfaces in future IPCC budgets and other inventories of anthropogenic GHG emissions. The reservoir-based CH_4 emissions reported here (8.9–22.2 Tg CH_4-C per year) are similar in magnitude to estimates of CH_4 emissions from rice paddies and to those from biomass burning (which includes biofuel emissions) by the IPCC (21–30 and 18–29 Tg CH_4-C per year respectively; Ciais et al. 2013). Reservoir CO_2 and N_2O fluxes, however, are lower than other anthropogenic or natural sources as reported by the IPCC (Ciais et al. 2013).

Although global-warming potentials for CO_2-equivalent calculations are often reported for a 100-year time span, the selection of time span is somewhat arbitrary (Myhre et al. 2013). CH_4 is relatively short-lived in the atmosphere (atmospheric lifetime on the order of a decade) relative to CO_2 (atmospheric lifetime on the order of centuries) and therefore has a higher global warming potential over the shorter 20-year time horizon (86 versus 34; Myhre et al. 2013). Policymakers should carefully consider the timescales that are relevant to GHG mitigation efforts, especially given the recent international push to maintain average global temperatures within 1.5–2°C of the pre-industrial mean (Fearnside 2015). Over shorter timescales (decades), and given the exclusion of several important alternative emission pathways (i.e., degassing, downstream and drawdown zone emissions; see section above), reservoirs are almost certainly contributing more than the 0.8 Pg CO_2 equivalents per year calculated here. In fact, when looking over the 20-year time horizon, CO_2 equivalent emissions from reservoir surface waters are estimated at double the flux presented here (1.7 Pg CO_2 equivalents per year , 5th and 95th confidence interval: 1.1 to 2.7 Pg CO_2 equivalents per year). With the current boom in global dam construction (Zarfl et al. 2015), reservoirs will represent an even larger fraction of anthropogenic CO_2 equivalent emissions in the coming years. Therefore, policymakers and water managers that are siting new dams or decommissioning old ones should weigh the multiple services that reservoirs provide against their GHG-related costs in planning to either construct or decommission a dam. A number of papers compare reservoir GHG emissions to those of the natural gas combined cycle (see the Larger Context section above), but many reservoirs do not produce energy at all.

Conclusions

Sixteen years ago, the first global review of reservoir GHG emissions highlighted the potential significance of reservoir surfaces as GHG sources and postulated that factors such as age, water temperature, and organic carbon inputs could regulate fluxes (St. Louis et al. 2000). At that time, there were GHG flux estimates from only 22 reservoir systems and potential controlling factors could not be quantitatively assessed. Here, we discuss a more comprehensive set of reservoir GHG flux estimates than has previously been analyzed, and use that data set to develop new insight into the rates and controls of reservoir GHG fluxes. Specifically, this work highlights the dominant contribution of CH_4 emissions to total reservoir carbon emissions, and the importance of including ebullitive CH_4 emissions in modeling efforts. Furthermore, it appears that reservoir nutrient loading and associated eutrophication leads to increased radiative forcing by reservoirs because of increased CH_4 emissions. The relationship between reservoir eutrophication and GHG emissions presented here provides a crucial first step in identifying potential management opportunities for the reduction of reservoir GHGs. Specifically, watershed nutrient reduction strategies aimed at preventing reservoir eutrophication may also mitigate both CH_4 and N_2O emissions (specifically via reduction of P and NO_3^- loading). In addition, when possible new reservoirs could be strategically sited upstream from anthropogenic nutrient sources. With the need for better global water management and the push for expanded global hydropower capacity, careful siting of new reservoirs, and revising management of existing ones may help balance the positive ecosystem services that reservoirs provide against the GHG emission costs.

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Supplemental material

The supplemental material is available online at *http://bioscience.oxfordjournals.org/lookup/suppl/doi:10.1093/biosci/biw117/-/DC1.*

There is also an accompanying supplementary spreadsheet available via Dryad (doi:10.5061/dryad.d2kv0) that contains the complete data set used in this synthesis and accompanying references.

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