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## Greenhouse Gas Emissions from Freshwater Reservoirs: What Does the Atmosphere See?

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### Abstract

Freshwater reservoirs are a known source of greenhouse gas (GHG) to the atmosphere, but their quantitative significance is still only loosely constrained. Although part of this uncertainty can be attributed to the difficulties in measuring highly variable fluxes, it is also the result of a lack of a clear accounting methodology, particularly about what constitutes new emissions and potential new sinks. In this paper, we review the main processes involved in the generation of GHG in reservoir systems and propose a simple approach to quantify the reservoir GHG footprint in terms of the net changes in GHG fluxes to the atmosphere induced by damming, that is, 'what the atmosphere sees.' The approach takes into account the pre-impoundment GHG balance of the landscape, the temporal evolution of reservoir GHG emission profile as well as the natural emissions that are displaced to or away from the reservoir site resulting from hydrological and other changes. It also clarifies the portion of the reservoir carbon burial that can potentially be considered an offset to GHG emissions.

### Keywords

GHG footprint; reservoirs; CO<sub>2</sub> and CH<sub>4</sub> emissions; C burial

## INTRODUCTION

Freshwater impoundments, like all inland aquatic systems, are sites of active carbon processing and transport. They receive carbon (C) in several forms (coarsely aggregated into

dissolved and particulate fractions of both inorganic and organic forms) and convert one species to another, some of which will be emitted to the atmosphere and some of which will be buried more or less permanently in their sediments, and some of which will be transferred downstream. For natural aquatic systems, research of the past decade has demonstrated that their role is disproportionately larger than their surface area would suggest (Cole and others 2007; Tranvik and others 2009; Borges and others 2015). Although they cover only about a few percent of the terrestrial surface of the Earth, the latest estimates (Raymond and others 2013; Holgerson and Raymond 2016) show that their net contribution to the atmosphere is of the same order of magnitude as the net flux between the oceans and the atmosphere (in the opposite direction) and greater than the delivery of terrestrial carbon to the oceans (IPCC 2013). Given the global importance of natural systems, it is to be expected that man-made reservoirs may be questioned as to their own footprints. Following the initial study of Rudd and others (1993), estimates of the greenhouse gas (GHG) emissions associated with reservoirs at the global scale vary by more than one order of magnitude (St-Louis and others 2000; Bastviken and others 2011; Barros and others 2011; Deemer and others 2016; Hertwich 2013) and underscore the considerable uncertainty of these assessments. Although part of this uncertainty is the result of the complexities of the biogeochemical processes involved, it is also due to the continuing lack of a clear methodology to assess their GHG footprint, a situation that has contributed to a steady stream of contentious assertions and heated debates over the past 20 years (Giles 2006; Fearnside 2006; Rosa and others 2006; Fearnside and Pueyo 2012; Fearnside 2016). At one extreme, reservoirs have been considered carbon-neutral or even as carbon sinks with respect to the atmosphere and, for hydro- power reservoirs, this has led to the idea of a completely green source of a renewable energy. At the other extreme, reservoirs are sometimes characterized as akin to coal-fired power plants, that is, intense GHG sources largely in the form of methane ( $\text{CH}_4$ ), a GHG 34 times more powerful than carbon dioxide as warming potential (over 100 years horizon, IPCC 2013). Documented cases approaching either of these two extremes can be found in the literature (Chanudet and others 2011; Delmas and Galy-lacaux 2001; Descloux and others 2017; Kemenes and others 2007).

Assessing the human contribution to reservoir GHGs is not straightforward, independently of the purposes for which they were originally built or are currently used for (flood control, hydroelectricity, irrigation, water storage, navigation, and so on). These human-built ecosystems constitute environmental reactors with new carbon pools (Thérien and Morrison 2005), additional transport pathways (while others are eliminated), altered *C* transformation processes, all of which occurring within a completely different physical setting induced by changes in flow regime, thermal stratification, depth, water-level fluctuations, land uses, and so on (Tremblay and others 2005a,b). Yet, it is the combined interactions between these factors that will ultimately determine the fate of the various carbon pools, the magnitude and direction of the associated *C* fluxes and thereby the GHG emissions attributable to damming. Disentangling which *C* fluxes are new (that is, a result of damming) from those that are natural (that is, would have taken place also in the absence of a dam) constitutes a difficult challenge and one that has remained largely unaddressed in the scientific literature as well as in discussions debating the environmental acceptability of dams with respect to GHG.

From a climate forcing perspective, assessing the anthropogenic impact of reservoir GHG emissions should account only for the net changes (positive or negative) in GHG fluxes to the atmosphere resulting from the landscape transformation of a river into a reservoir. The purpose of this mini-re- view is to draw from our current but scattered knowledge of carbon cycling in aquatic ecosystems to produce a frame of reference consistent with an accounting of GHG emissions that reflects only what is ‘visible to the atmosphere,’ that is, the excess emissions/sequestration due to damming. We review the relevant biogeochemical processes, integrate their temporal evolution following impoundment and identify the terms of the carbon mass balance that need accounting or discounting and show that the resulting GHG footprint assessment can differ markedly from direct flux measurements. Given the number of existing and ongoing damming projects in various stages of completion (Zarfl and others 2014), we suggest that such an approach can contribute to a better understanding of the carbon cycling of these societally important ecosystems while also informing efforts to account for GHG emissions by regulatory agencies.

### **NET GHG EMISSIONS: DISENTANGLING THE PRE- AND POST-IMPOUNDMENT BALANCES, AND NATURAL VERSUS NEW EMISSIONS**

On a carbon currency, CO<sub>2</sub> is by far the most important GHG species in aquatic ecosystems. In itself, finding that freshwater reservoirs emit large quantities of GHG, in particular CO<sub>2</sub>, is not surprising given that, like natural lakes, reservoirs too receive carbon-rich organic material from the terrestrial landscapes they drain. Nearly all published reports of GHG from reservoirs have largely been interpreted, implicitly or explicitly, as emissions resulting from the creation of the reservoir. Often termed ‘gross emissions’ (see Tremblay and others 2005a,b), they provide a first-order estimate of the post-impoundment conditions at a fixed point in time after flooding. However, when a dam is erected, the resulting flooding provokes multiple changes in the carbon cycling in every compartment of the affected landscape such that emissions at the reservoir surface become a mixture of multiple GHG sources and sinks, some preexisting, some not. Disentangling preexisting and new GH sources is necessary to apportion correctly the part of the GHG emissions that can be legitimately attributed to the creation of a reservoir itself, that is, the new emissions that the atmosphere will see. As we proceed, Table 1 provides a summary of the fluxes that are deemed ‘visible’ or ‘invisible’ to the atmosphere.

### **THE PRE-IMPOUNDMENT GHG BALANCE**

Areas that have been flooded for the purpose of creating a reservoir are rarely homogeneous in land cover but instead contain a mosaic of terrestrial and sometimes aquatic components, each of which are characterized by a specific GHG balance (positive or negative). Consequently, the net carbon balance of the landscape area that will be occupied by a future reservoir is rarely carbon neutral (but see Teodoru and others 2012). Most terrestrial landscapes are net carbon sinks either because the forest biomass is accruing (for example, immature or recently perturbed forests, Coursolle and others 2012) or because the soils are slowly accumulating carbon (Hassink 1997). Ignoring that flooding a rapidly growing forest would suppress a significant carbon sink would naturally lead to an underestimation of the GHG footprint of the reservoir. Conversely, some terrestrial landscape components can also be sources of GHG (for example, CH<sub>4</sub> emission from peatlands and swamps with higher

impact than the simultaneous  $C$  storage in sediment or peat) and flooding such  $CH_4$  emitting land could, at least theoretically, reduce the GHG balance. Ultimately, it is the integrated sum of the respective  $CO_2$  or  $CH_4$  balances of this landscape mosaic that is required to assess the change in GHG fluxes visible to the atmosphere.

Perhaps the best illustration of the importance of accounting for the pre-impoundment GHG balance is for reservoirs that were formerly natural lakes. Although the construction of a regulating structure on a natural lake may increase its water level and therefore surface area, it is clear that any reasonable approximation of the anthropogenic GHG footprint of such a regulated lake must also consider and subtract the pre-impoundment fluxes. This is the so-called net approach (for example, IEA Hydropower 2012). According to the GRanD compilation of 6821 worldwide reservoirs (reservoirs with  $> 0.1 \text{ Mm}^3$  capacity, Lehner and others 2011), about 30% of the global reservoir surface area is occupied by the four largest reservoirs (Onega, Ontario, Baikal and Victoria), all of which are natural lakes that have been dammed for various purposes but for which the surface area has changed very little. Simply estimating the total gross emissions of these systems would clearly overstate the GHG footprint induced by their impoundment. Run-of-the-river reservoirs are analogous in that the river channel (and hence surface area) remains largely unaltered although the biogeochemistry of the system and GHG footprint may be modified as a consequence of increased sediment trapping (Sobek and others 2012) or altered hydrological regime (de Faria and others 2015). Nevertheless, the principle remains the same whether the new reservoir is completely replacing a terrestrial mosaic or slightly expanding a natural lake or river: the pre-impoundment carbon (and GHG) balance prevailing prior to the impoundment needs to be subtracted from post-impoundment (or gross) GHG emissions (Teodoru and others 2012). However, the resulting calculation is still incomplete to assess the GHG footprint of reservoirs because it neglects the critical question of how much of these net emissions would have occurred regardless of the presence of the reservoir, how much may have been displaced elsewhere and what is the temporal evolution of the post-impoundment emissions. The following section examines how the altered biogeochemical processes will affect the timing, quantity, form and location of  $CO_2$  and  $CH_4$  emissions, all of which will ultimately determine their attribution, or not, as an anthropogenic GHG impact 'visible' by the atmosphere.

## THE POST-IMPOUNDMENT GHG BALANCE

**Carbon Dioxide**—In natural systems such as lakes or rivers, there are two main pathways that can lead to the oversaturation and subsequent emission of  $CO_2$  to the atmosphere. First, soil waters can reach very high dissolved  $CO_2$  concentrations (for example, Linn and Doran 1984; Vachon and others 2016a) resulting from the intense but physically confined soil respiration processes. When this  $CO_2$ -rich soil water enters surface waters (lake or stream), the excess gas can be released in the atmosphere. The resulting emissions are essentially soil respiration products escaping through an aquatic portal, but it needs to be recognized that the same mass of would have reached the atmosphere whether it had evaded through a water or soil interface. This pathway is quantitatively most prevalent in streams and rivers of low order (Butman and Raymond 2011; Hotchkiss and others 2015) and is probably not expected to influence the  $CO_2$  balance of most reservoirs. The second main pathway through which

aquatic CO<sub>2</sub> oversaturation arises is from in situ degradation by aquatic microbes and photochemical processes of both dissolved and particulate organic carbon (DOC and POC) of allochthonous origin, either in the water column or in the bottom sediments (Pace and Prairie 2005). The fraction of the organic carbon inputs received by waterbodies that will be mineralized depends on several factors, but water residence time is known as a key modulator of the extent of DOC degradation (Dillon and Molot 1997; Algesten and others 2004; Sobek and others 2007; Vachon and others 2016b), whereas POC degradation is strongly regulated by temperature and oxygen availability (Gudas and others 2010; Sobek and others 2009). An additional source of CO<sub>2</sub> may arise when a significant amount of methane produced under anaerobic conditions is oxidized to CO<sub>2</sub> by methanotrophs prior to evasion to the atmosphere. Although this indirect production is generally small relative to other CO<sub>2</sub> sources, it can be significant in some cases (Guérin and Abril 2007) and is visible by the atmosphere.

Freshwater reservoirs, as new constituents of the hydrological network, are subject to the same processes of DOC loading, in situ degradation and groundwater CO<sub>2</sub> injection as natural systems. In addition, however, the organic carbon contained in soils and biomass that were flooded during the impoundment constitute a direct new and potentially large source of organic matter that can, in principle, be mineralized and liberated as GHG (Courchesne and Turmel 2005). Quantifying the fraction of the CO<sub>2</sub> emissions that can be traced to these new sources of organic material is in itself difficult and is further confounded by several factors such as the decay of emergent vegetation and, more generally, by the altered degradation dynamics of the natural OM sources (Vachon and others 2016b).

**Displaced CO<sub>2</sub> Emissions:** One of the most significant hydrological changes resulting from the impoundment of a river is the increased water residence time ( $\tau_w$ ). The original river segment located within the future reservoir area typically has a  $\tau_w$  of a few hours, while that of reservoirs can be much longer (days, months or, at times, even years). This dramatic increase is important because the fraction of the external load of organic carbon that will be mineralized within the system is known to increase with  $\tau_w$  (Dillon and Molot 1997; Algesten and others 2004). The net result is that proportionately more DOC will be turned into CO<sub>2</sub> at the reservoir site than would have been in the pre-impoundment river, leading to a measurable system-wide increase in CO<sub>2</sub> emissions. In addition to the water residence time effect, mineralization rates can be further modified at the reservoir site by several factors such as enhanced UV exposure (Bastien 2005), warmer temperatures or through a priming effect from elevated nutrients (Guérin and others 2008; Guenet and others 2010, but see Catalaén and others 2015). However, unless the DOC is protected from degradation while flowing in a river but not so within the reservoir itself, the same emissions would have taken place nonetheless, albeit distributed at locations further downstream, including the sea. These emissions are not new but are simply displaced in space; they are therefore largely invisible to the atmosphere as a 'new' GHG flux (Table 1).

We can quantify the importance of this phenomenon in isolation from the confounding influence of soil C degradation using the approach recently developed by Vachon and others (2016b). The model (their Equation 7) explicitly solves how the apparent reactivity of DOC will be affected by changes in  $\tau_w$  even if the nature of the inflowing material remains

unchanged, a scenario that describes well the sudden transformation of a river into a reservoir. Dissolved organic carbon is a complex amalgam of different organic compounds each with their own degradation characteristics such that the most labile molecules will be mineralized first, while the more recalcitrant fraction will remain longer, a phenomenon known as the reactivity continuum (Boudreau and Ruddick 1991; Koehler and others 2012). Vachon and others (2016b) combined the reactivity continuum concept with the age distribution of water within systems to yield an expression linking water residence time ( $\tau_w$  in days) and the apparent DOC decay rate ( $k_{DOC}$ , in  $d^{-1}$ ) for systems receiving inputs of a given reactivity (where  $a$  and  $m$  are the average lifetime of the more reactive compounds and the relative abundance of the most recalcitrant compounds, respectively) as

$$k_{DOC} = \frac{0.85v}{\tau_w} \ln \left[ 1 + \frac{\tau_w}{\alpha} \right] \quad (1)$$

At steady state, the dynamic concentration equilibrium between a reservoir ( $[DOC]_{res}$ ) and its inflowing waters ( $[DOC]_{in}$ ) can be expressed as  $[DOC]_{res} = [DOC]_{in} \left( \frac{1}{1 + \tau_w \times k_{DOC}} \right)$  and the resulting annual C mass lost to the atmosphere (moles  $yr^{-1}$ ) is  $Q \times [DOC]_{in} \times \left( \frac{\tau_w k_{DOC}}{1 + \tau_w \times k_{DOC}} \right)$ , where  $Q$  is the annual water input. In the context of a landscape transformation from a river to a reservoir, the loading of DOC will remain constant and therefore the relative increase in  $CO_2$  emissions following impoundment can be expressed as

$$\frac{(CO_2 \text{ emis})_{post}}{(CO_2 \text{ emis})_{pre}} = \frac{\left( \frac{\tau_w k_{DOC}}{1 + \tau_w \times k_{DOC}} \right)_{post}}{\left( \frac{\tau_w k_{DOC}}{1 + \tau_w \times k_{DOC}} \right)_{pre}} \quad (2)$$

Equations (1) and (2) can be combined to assess the expected increase in  $CO_2$  emissions resulting only from the hydrological changes caused by impoundment. Solutions of Equation (2) for any reactivity of the inflowing riverine DOC (that is, fixed  $a$  and  $m$ ) show that the relative increase in  $CO_2$  emissions is a nearly proportional function of the relative increase in water residence time. As an example, a river segment with  $\tau_w = 0.5$  day receiving DOC of average reactivity ( $a = 25$  days and  $v = 0.11$ , see Vachon and others 2016b) converted to a reservoir with  $\tau_w = 90$  days will see its system-wide  $CO_2$  footprint increase by about 70-fold. The apparent areal gas efflux rate will then depend on the concomitant relative increase in surface area of the system. In the case of reservoirs with anoxic hypolimnia, the increase in  $CO_2$  release may be lower than predicted strictly from the change in the hydrology because of the lower mineralization rates under anoxic conditions (Bastviken and others 2002). Regardless of the exact increase, this inflowing organic carbon would have also been subjected to mineralization in the original riverine situation, except they would have occurred somewhere further downstream and likely at different rates. Thus,

these nominally reservoir emissions can be considered invisible to the atmosphere because they have simply been displaced and would have occurred in the absence of the reservoir (Table 1).

**Assessing the Additional Anthropogenic CO<sub>2</sub> Emissions from Reservoirs:** Although challenging, the exercise of assigning the fraction of CO<sub>2</sub> emissions that can be legitimately attributed to the creation of the reservoir can be evaluated quantitatively in two different ways: either through the assessment of the potential new OM sources or by subtraction, where the anthropogenic portion emissions are calculated as total emissions minus natural sources (for example, de Faria and others 2015). Both methods have strengths and limitations. In the first approach, the potential new source of carbon is the flooded organic material, namely that contained in the soil and biomass (Courchesne and Turmel 2005). Only considering this source in GHG assessment has the advantage of intrinsically discounting natural sources. However, it remains difficult to estimate the potential degradability of the individual organic compartments of the flooded material. For example, standing biomass such as tree trunks and branches has shown negligible decomposition after decades under water and even under tropical conditions (Guérin and others 2008; Cailleaud 2015), implying that their role as CO<sub>2</sub> sources is largely negligible except over long (centennial) timescales. Standing biomass remaining above the water surface can decompose significantly faster (Abril and others 2013). At the other end of the degradability spectrum, fresh leaf litter and newly released dissolved organic substances can be quite labile in water (for example, Guérin and others 2008). Whether CO<sub>2</sub> produced from the mineralization of flooded leaves should be considered visible to the atmosphere is debatable since most of it would have returned to the atmosphere over short timescales in the terrestrial environment as well. Thus, the main remaining potential new source of carbon is the soil C that would not otherwise have been decomposed in the absence of flooding. It also constitutes the upper bound of the CO<sub>2</sub> emissions that can be attributed to the reservoir.

There are very few studies available on the decay rate of flooded soils, and they have shown contrasting effects, either enhancing (Blodau and Moore 2003; Oelbermann and Schiff 2008) or decreasing (Kim and others 2015) mineralization rates. Although none of these studies are long term, it is reasonable to assume that only the upper portion of the soil profile can be mineralized aerobically, because diffusive transport of oxygen from the overlying waters is slow, a process that, in natural lakes, accounts for most degradation occurring in the early (5–10) years post-sedimentation (Teranes and Bernasconi 2000; Gälman and others 2008). Assuming that the short-term decay rates of flooded forest soils observed experimentally by Kim and others (2015) can operate unabated for 10 years (between 5 and  $10 \times 10^{-5} \text{ d}^{-1}$ ), then between 15 and 30% of the organic carbon mass will have been lost as CO<sub>2</sub> over that time period. Similarly, Guérin and others (2008) estimated that about 22% of the soil carbon flooded under the tropical Petit-Saut reservoir (French Guyana) was decomposed over the same time period. Allowing for a potentially longer (but slower, see Gälman and others 2008) decay period, we estimate that a maximum of about half of the soil C could be lost through mineralization after flooding.

The second approach to quantify the amount of CO<sub>2</sub> emission that is legitimately attributable to the creation of a reservoir is to deduct from total reservoir emissions the

portion that can be considered natural. If the degradability parameters of the inflowing material are known for a given reservoir, then the expected CO<sub>2</sub> resulting only from the change in hydrological regime could theoretically be calculated using Equations (1) and (2). More generally, however, it is useful to rely on known patterns observed in many reservoirs following impoundment. Teodoru and others (2012) and Abril and others (2005) provide compelling examples illustrating how the CO<sub>2</sub> emission surges immediately after impoundment and then quickly recedes in the following years. In the case of the boreal Eastmain-1 reservoir, the rapid exponential decline would suggest negligible emissions after 10 years (Teodoru and others 2012). A meta-analysis of published emission studies (Prairie and others 2017) instead suggests that the rate of decline decreases with time (faster in the early years, slower later on) and that the temporal evolution of CO<sub>2</sub> emissions is better expressed as a general negative power function of the form Flux CO<sub>2</sub> = C x Age<sup>-0:29</sup> where C is a reservoir-specific constant that depends on other factors such as temperature, nutrients or reservoir area. The CO<sub>2</sub> flux that is deemed natural can then be approximated by assuming that the new equilibrium emissions were reached after a given number of years. The literature suggests that 10 years is a realistic period for the return to a quasi-equilibrium (for example, Tremblay and others 2005a,b), reflecting the new balance between primary production and respiration of the reservoir ecosystem. More conservatively, we can assume instead that this new equilibrium is reached only after 100 years (Figure 1), a value often used often to represent the expected lifetime of reservoirs in life-cycle analysis (for example, Gagnon and others 2002). Under this assumption, total emissions attributable to the reservoir can be estimated by integrating the power function as

$$\int_1^{100} C \times (\text{Age}^{-0:29} - 100^{-0:29}) d$$

Age and would still imply that about 75% of the cumulative CO<sub>2</sub> flux is natural, that is, that only 25% can be considered the result of the impoundment process. Thus, current assessments of the CO<sub>2</sub> footprint likely grossly overestimate the visible atmospheric impact of reservoir CO<sub>2</sub> emissions if natural CO<sub>2</sub> emissions are not discounted.

*Methane* The case of CH<sub>4</sub> is conceptually easier than CO<sub>2</sub> in its attribution to the creation of the reservoir itself. Methanogenesis generally requires anaerobic conditions (but see Grossart and others 2011; Bogard and others 2014), and when a reservoir is filled, the newly flooded land generally results in the creation of a new anoxic zone just below the soil–water (now sediment–water) interface that can accelerate CH<sub>4</sub> production (for example, Delmas and Galy-lacaux 2001). In addition, depending on the environmental setting, the impoundment’s hypolimnetic zone can also itself become anoxic and become a compartment where new methane can be produced and accumulated, although some will be oxidized during the transport through the water column (mostly at the chemocline, Bastviken and others 2002; Guérin and Abril 2007). Regardless of the rates and location of this new methanogenesis, it is a reasonable assumption to consider that all CH<sub>4</sub> emissions from reservoirs are new and therefore anthropogenic (Table 1).



Although ascribing these emissions to the direct effect of damming is straightforward, the complex biogeochemistry of methane leads to significant challenges in its estimation. The multiple processes and pathways (Figure 2b) through which methane is transformed and emitted from aquatic systems (diffusion, ebullition, macrophyte-mediated and degassing in the case of reservoirs) are all highly temporally and/or spatially variable (Bastviken and others 2004a, b; DelSontro and others 2010, 2011; Guérin and others 2006; Prairie and del Giorgio 2013; McGinnis and others 2015). This variability is tightly linked to the environmental conditions necessary for the production of methane (see Conrad 2005 for review), but also to the management of the reservoir water level (Ostrovsky and others 2008; Harrison and others 2017). In thermally stratified reservoirs, anoxic conditions within the surface mixed layer are essentially confined to the littoral sediments/flooded soils, an area typically comprising just a narrow fringe around the reservoir perimeter. Methane produced there can be emitted through both diffusion and ebullition. Bubbles released from shallow sediments can reach the water surface largely unaltered (McGinnis and others 2006) and therefore bypass microbial methane oxidation. Sediment methane production can also diffuse to the water and, if it escapes oxidation at the sediment water interface, requires transport from horizontal diffusivity and advection throughout the reservoir surface, leading to a substantial horizontal concentration gradient (DelSontro and others 2017). Some reservoir studies have demonstrated strong spatial heterogeneity of CH<sub>4</sub> emission (Musenze and others 2014; Grinham and others 2011) that, in many cases, are linked to areas of high sediment deposition (Sobek and others 2012; DelSontro and others 2011; Maeck and others 2013). Similarly, the degree to which hypolimnetic methane can reach surface waters through diffusion, particularly at the boundaries, remains difficult to assess. Finally, the extreme spatial patchiness, the largely unpredictable timing of bubble emission and CH<sub>4</sub> redissolution during bubble ascent (McGinnis and others 2006) all contribute to make accurate measurements of methane ebullitive fluxes poorly constrained. As an example, the most spatially integrative method to measure CH<sub>4</sub> ebullition is obtained using acoustic technology (Ostrovsky and others 2008). Yet, within a single site, replicate measurements using this method have coefficients of variation typically exceeding 100% (DelSontro and others 2011).

Not surprisingly, data derived from imprecise and/or insufficiently integrative measurements necessarily hinder the identification of the main drivers of CH<sub>4</sub> emissions across different systems. Nevertheless, recent studies suggest that reservoir age (Abril and others 2005; Barros and others 2011), trophic status (Deemer and others 2016; DelSontro and others 2016), temperature (Rasilo and others 2014; or its proxy, latitude, Barros and others 2011) and the extent of the littoral zone (DelSontro and others 2017) are all important modulators of the average CH<sub>4</sub> emission rates in aquatic systems. A feature that is unique to some reservoirs (mostly those constructed for hydroelectricity and flood control) and absent in natural systems is a direct connection between the deeper section of the water column and the downstream river. In the case of hydroelectric reservoirs, these conduits serve to feed the turbines to generate electricity. Depending on the particular configuration of the reservoir, the water intake level can be sufficiently high in the water column to draw mostly warm but well-oxygenated low GHG waters from the reservoir surface layer. In others, the intake level is located well below the thermocline where, in many eutrophic and/or tropical systems,

hypolimnetic CH<sub>4</sub> can accumulate and reach very high concentrations. When such CH<sub>4</sub>-rich water passes through the turbines, CH<sub>4</sub> can be liberated directly to the atmosphere (degassing) or from the river surface below (downstream emission). In some cases (for example, Abril and others 2005; Kemenes and others 2007), this degassing pathway constitutes the main mechanism through which GHG is released to the atmosphere. Although CO<sub>2</sub> is also released through this pathway, its contribution would be already taken into account if the visible CO<sub>2</sub> footprint is estimated from the flooded soil carbon pool. The combined degassing and downstream emissions are difficult to predict because they depend on the exact level of the water intake, the vertical CH<sub>4</sub> concentration profile, heterotrophic respiration, bathymetric shape and diffusion-limited O<sub>2</sub> supply to the oxycline (Deshmukh and others 2016; Guérin and others 2006), as well as the extent of oxidation in the downstream river segment. This is a topic for which much more research is required to adequately assess its potential importance and how it varies depending on the particular environmental setting and construction configuration of each reservoir.

Notwithstanding these limitations to representative measurement and prediction of CH<sub>4</sub> emission, the bulk of the methane produced in a reservoir can legitimately be attributed to the creation of the reservoir itself. Only the methane naturally emitted from the pre-impoundment river could be deducted from the post-impoundment emissions, a value that is rarely more than a few percent of the reservoir emissions, except where the reservoir is in fact an impounded lake (see section on pre-impoundment conditions). Thus, the main challenge in assessing the true CH<sub>4</sub> footprint of reservoirs resides in accurately estimating the net balance between the production and consumption processes, through which pathway it is released to the atmosphere, and how it is strongly modulated in reservoirs of differing environmental conditions (Figure 2).

## **CARBON SEDIMENTATION: MIRES AND UNCERTAINTY**

Aquatic ecosystems have a unique dual role in that they can simultaneously act as a carbon source to the atmosphere while accumulating and permanently storing carbon in sediments. This is only possible because, unlike terrestrial ecosystems, lakes and reservoirs receive large inputs of organic carbon from catchments. Reservoirs, particularly in agricultural and other nutrient-rich areas, can accumulate large amounts of organic carbon in their sediments, (for example, Dean and Gorham 1998; Downing and others 2008; Clow and others 2015), a fact that has sometimes been construed as an offset to C emissions (Sikar and others 2009; IEA Hydropower 2012; Bernardo and others 2016; dos Santos and others 2017). In our view, this can be misleading because it depends strongly on the origin of the sedimented C and what the fate of the associated carbon would have been in the absence of a reservoir. The relevance of sedimentation processes in the accounting of the GHG footprint of reservoirs is best illustrated using simple scenarios as to the origin of the sedimented carbon.

As a first scenario, consider the hypothetical case of a reservoir receiving no organic matter from its catchment and in which in situ primary production by either phytoplankton or macrophytes is the only potential source of the sediment carbon. By definition, this is a system that would have to be undersaturated in CO<sub>2</sub> relative to the atmosphere because the CO<sub>2</sub> assimilated via primary production will reduce its partial pressure. At equilibrium, the

net CO<sub>2</sub> influx at the air–water interface would have to match the net C burial in the sediments. This hypothetical scenario clearly illustrates the problem associated with subtracting the net C burial from the net air–water exchange; it would amount to discounting the same carbon twice! More importantly, however, this carbon ‘accounting problem’ remains whole in the more realistic scenario where the reservoir receives significant organic carbon from its watershed, a portion of which is mineralized to CO<sub>2</sub> within the system. In this case, the CO<sub>2</sub> generated by heterotrophic respiration of the allochthonous OC would contribute to the water CO<sub>2</sub> partial pressure, thereby modulating the air–water pCO<sub>2</sub> gradient and its associated flux. This can be described by a simple CO<sub>2</sub> mass balance calculation. Except in very fast flushing lakes or reservoirs, the amounts of CO<sub>2</sub> entering and leaving by the hydrological network are small (< 5%) relative to the other terms of the mass balance and, for simplicity, will be omitted in the following equation:

$$V \frac{d \Delta [\text{CO}_2]_L}{dt} = A \times (R_{\text{autoch}} + R_{\text{alloch}} - \text{GPP}) - k_{600} \times A \times \Delta \text{CO}_{2L} \quad (3)$$

where  $[\text{CO}_2]_L$  is the lake CO<sub>2</sub> concentration (expressed as a differential from atmospheric equilibrium),  $k_{600}$  is a gas exchange coefficient,  $A$  is surface area,  $V$  is volume, and  $\text{GPP}$ ,  $R_{\text{autoch}}$  and  $R_{\text{alloch}}$  are, respectively, the areal rates of gross primary production, respiration of autochthonous OM and respiration of allochthonous OM. (Negative corresponds to removal of CO<sub>2</sub>.) At steady state, the rate of sedimentation of autochthonous OM ( $S_{\text{autoch}}$ ) must equal the difference between  $\text{GPP}$  and  $R_{\text{autoch}}$ , and the equation simplifies to

$$k_{600} \times \Delta \text{CO}_{2L} = R_{\text{autoch}} + R_{\text{alloch}} - \text{GPP} = R_{\text{alloch}} - S_{\text{autoch}} \quad (4)$$

showing that the sedimentation rate of autochthonous OM is necessarily reflected in the CO<sub>2</sub> exchange with the atmosphere (term on the left hand-side). Depending on the balance between autochthonous sedimentation and allochthonous respiration (largely derived from the heterotrophic respiration of allochthonous DOC), there will be an influx or efflux of CO<sub>2</sub> at the air–water interface. Thus, measuring the net exchange at the air–water interface already accounts for the internal CO<sub>2</sub> fluxes and DOC transformation within the system, including the organic carbon buried via sedimentation of autochthonous material (algal or macrophyte production) (Table 1).

Lastly, consider another scenario in which the sediment carbon originated instead from the direct deposition of allochthonous particulate organic carbon (POC) within the reservoir, a phenomenon plausible because of the longer water residence time of the reservoir relative to the original system. Reservoirs are very effective in trapping the riverine suspended sediment load (Vörsmarty and others 2003; Maavara and others 2017). Dissolved organic carbon directly flocculating and sedimenting in the reservoir would also behave similarly. In this scenario, the reservoir basically acts as a local trap. If it bypasses completely any GHG phase, it is therefore invisible to the atmosphere although high sedimentation rates have been shown to fuel intense methanogenesis and bubbling (Sobek

and others 2012; Maeck and others 2013; Wilkinson and others 2015). However, from the perspective of a net change at the larger regional scale, only the fraction of this  $C$  burial that would not have been sedimented elsewhere in the original hydrological network (including the coastal ocean) could potentially be considered an offset to emissions and therefore be discounted. Such a scenario is possible if the carbon burial efficiency is sufficiently enhanced by the high sedimentation rates induced by the dam (Sobek and others 2009). Thus, while sediment carbon accumulation can now be reliably estimated using sub-bottom profilers (Mendonça and others 2012, 2014; Ferland and others 2012, 2014), it is generally incorrect to simply ascribe the measured bulk rates of  $C$  burial as an offset to surface GHG emissions. In many cases, it would constitute a case of displaced  $C$  burial, akin to the displaced emissions described above and thus invisible to the atmosphere. Nevertheless, estimating the putative fraction that constitutes new burial, i.e., in excess to what would have occurred in a pre-impoundment situation, is a difficult task that merits further research. It will require tracing the POC burial efficiencies dynamics all the way to its ultimate fate, including the oceans. It will also require taking into account any sediment management practices (for example, sediment sluicing) at the dam site.

## SUMMARY AND AREAS OF FUTURE RESEARCH

We have outlined in this mini-review the key components of the carbon cycle of reservoirs necessary to develop a coherent basis to assess the anthropogenic GHG footprint of reservoirs. While the science of reservoir GHG has matured considerably in recent years, there are still fluxes that are only loosely constrained and for which active research is required. In many systems, the water-level fluctuations inherent to the management operations periodically expose large surface areas known as drawdown zones (for example, Three-Gorges and Nam Theun 2 reservoirs). The few reported measurements have shown considerable variation, ranging from small to significant  $CH_4$  sources (Sercia and others 2016; Chen and others 2009; Yang and others 2012) and, in some cases, even  $CH_4$  sinks (Li and others 2016). The degree and generality to which these rates represent higher GHG emissions relative to the surface emissions when reservoirs are at full capacity are largely unknown and will require more research on different soil types. Similarly, empirical rules to estimate the magnitude of downstream emissions from stratified reservoirs are currently lacking. Given the dominance of this pathway in certain systems, it represents a significant knowledge gap. Lastly, the issue of carbon burial is probably one of the most complex to resolve, in particular, how to determine the fraction that could be potentially considered an offset to GHG emissions.

Although uncertainties remain, our proposal emphasizes the need to quantify the GHG footprint of reservoirs on the basis of whether they have generated new fluxes visible to the atmosphere. We provide here some of the methodological elements necessary to help distinguish between natural fluxes and those induced by the reservoir impoundment, thereby forming a basis for the accounting of the anthropogenic GHG footprint of reservoirs, such as those required by international regulatory agencies.

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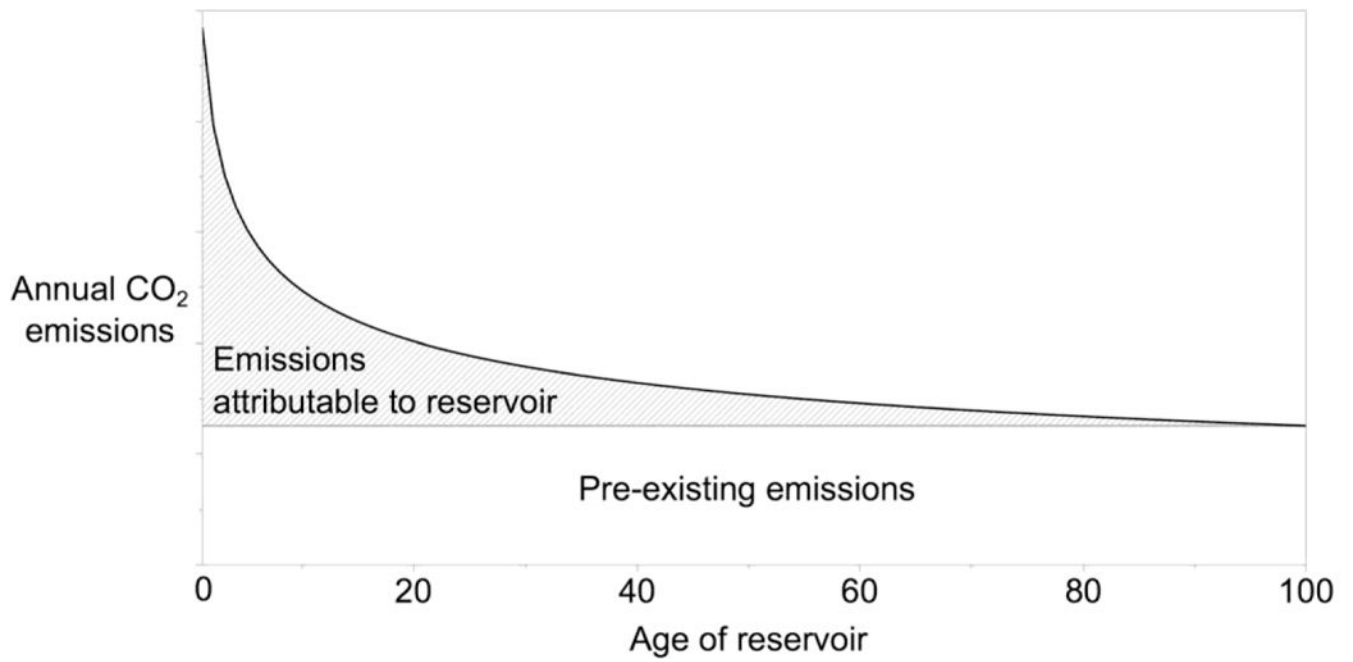
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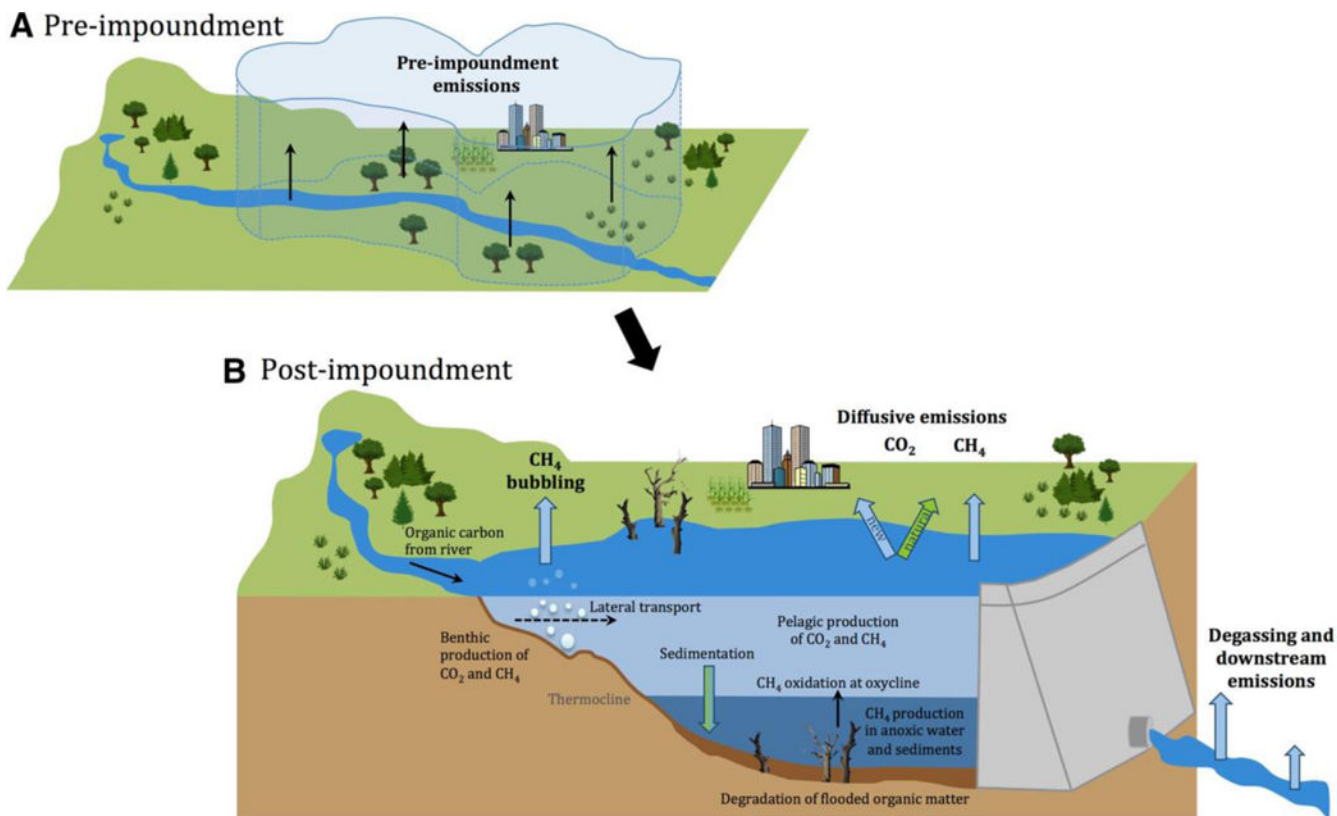
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**Figure 1.** Decline in CO<sub>2</sub> emission over 100 years post-impoundment. Assuming conservatively that CO<sub>2</sub> emissions at 100 years correspond to a new natural equilibrium, the hatched area represents the integrated additional CO<sub>2</sub> efflux resulting from the creation of the reservoir.



**Figure 2.** Landscape transformation from a river to a reservoir. A Pre-impoundment conditions: the dashed line corresponds to the area of the future reservoir where the GHG balance needs to be accounted for. B Post-impoundment conditions and the multiple processes and pathways for CO<sub>2</sub> and CH<sub>4</sub> emissions. Although a large fraction of the diffusive CO<sub>2</sub> emissions occurring at the reservoir surface can be considered natural or ‘invisible to the atmosphere’ (green arrows), the fraction emanating from the mineralization of flooded organic material is new and therefore ‘visible to the atmosphere’ (blue arrows) and induced by the impoundment. In contrast, essentially all CH<sub>4</sub> emissions can be considered attributable to the reservoir, except those present in the pre-impoundment conditions. The main CH<sub>4</sub> emission pathways are diffusive at the reservoir surface, bubbling emissions and degassing/ downstream emissions, all of which are modulated to different extent by methane oxidation. Depending on the source of the sedimenting material and the increase in C burial efficiency, a variable but unknown portion of the carbon burial can be potentially considered an offset to GHG emission (Color figure online).

**Table 1.**

Components of the Reservoir Carbon Cycle for Which the Landscape Transformation to a Reservoir Constitutes New Net Fluxes to the Atmosphere

C cycle component	Process or pathway	Visible	Invisible or displaced
CO <sub>2</sub> emission	Soil–water CO <sub>2</sub> input		X
	OC loading and in situ degradation		X
	Flooded soil C degradation	X	
CH <sub>4</sub> emission	CH <sub>4</sub> production from anoxic sediment and ebullition	X	
	CH <sub>4</sub> produced from autochthonous OC	X (as GWP)	X (as C)
	Drawdown zones	X	
C burial	C Sedimentation of in situ primary production		Accounted in CO <sub>2</sub> air-water exchange
	C sedimentation of allochthonous OC	Change in burial efficiency	

In all cases, the pre-impoundment GHG balance needs to be accounted for

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