

Greenhouse gas emissions from freshwater reservoirs: what does the atmosphere see?

Short title: GHG emissions from freshwater reservoirs

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Abstract

Freshwater reservoirs are sources of greenhouse gas to the atmosphere. Yet, the quantitative significance of these sources is still only loosely constrained, in part because of the methodological difficulties associated with the measurement of highly variable fluxes in both space and time. However, part of the uncertainty is also the result of a poor definition of the relevant concepts, particularly about what constitutes new emissions and what constitutes potential new sinks. In this paper, we review the main processes involved in the generation of GHG in reservoirs 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, i.e. 'what the atmosphere sees' following the landscape transformation from a river into a reservoir. 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 can potentially be considered an offset to GHG emissions.

Key words: GHG footprint, reservoirs, CO₂ and CH₄ 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), 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; Raymond and others 2013; Borges and others 2015). Although they cover only about a few percent of the terrestrial surface of the Earth, the latest estimates 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. Given the global importance of natural systems, it is to be expected that man-made reservoirs may be questioned as to their own footprint. Following the initial study of Rudd and others (1993), estimates of the 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 et al 2013) and underscore the considerable uncertainty of these assessments. While 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 and others 2006; Rosa and others 2006; Fearnside and others 2012; Fearnside and others 2016). At one extreme, reservoirs have been considered carbon-neutral or even as carbon sinks with respect to the atmosphere and, for hydropower 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, i.e. intense GHG sources largely in the form of methane (CH₄), a GHG 34 times more powerful than carbon dioxide as warming potential (CO₂, IPCC 2013). Documented cases approaching either of these two extremes can

be found in the literature (Chanudet and others 2011; Delmas and others 2001; Descloux and others 2017; Kemenes and others 2007).

Admittedly, assessing the anthropogenic GHG footprint resulting from the creation of a reservoir is not as straight-forward as might appear at first glance and this, independently of the purposes they were originally built or are currently used for (flood control, hydro-electricity, irrigation, water storage, navigation, etc). These human-built ecosystems constitute environmental reactors with new carbon pools, 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, etc,... 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 (i.e. a result of damming) from which are natural (i.e. 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 reflect only 'what the atmosphere sees' following the conversion of a river and its surroundings to a reservoir. In this paper, we consider that reservoir emissions are visible to the atmosphere when they induce a net change (either positive or negative) in GHG fluxes as a result of the landscape transformation. The purpose of this mini-review is to draw from our current but scattered knowledge of carbon cycling in aquatic ecosystems to produce a conceptual framework coherent with an accounting of GHG emissions 'visible to the atmosphere'. We review the main processes involved, their temporal evolution following impoundment, and identify the main terms of the carbon mass balance that need accounting or discounting. 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 and also has important implications to GHG accounting methodologies required by regulatory agencies.

Net GHG emissions: disentangling the pre- and post-impoundment balances, and natural versus new emissions

On a carbon currency, CO₂ is by far the most important GHG species in aquatic ecosystems. In itself, finding that freshwater reservoirs emit large quantities of GHG, in particular CO₂, is not surprising given that, like natural lakes, they too receive carbon-rich organic material from the terrestrial landscapes they drain. To our knowledge, 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”, 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 pre-existing, some not. Disentangling them is necessary to apportion correctly the part of the GHG emissions that can be legitimately attributed to the creation of a reservoir itself, i.e. 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 comprise 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 (e.g. 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 (e.g. CH₄ emission from peatlands and swamps with higher impact than the simultaneous C storage in sediment or peat) and flooding such CH₄ emitting land could, at least theoretically, reduce the GHG balance. Ultimately, it is the integrated sum of the respective CO₂ or CH₄ 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 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” (e.g. IEA 2012). According to the GRanD compilation of 6821 worldwide reservoirs (reservoirs with >0.1 Mm³ capacity, Lehner and others 2011), about 30% of the global reservoir surface area is occupied by the 4 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 their 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 Farias 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. 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 CO₂ and CH₄ 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 CO₂ to the atmosphere. First, soil waters can reach very high dissolved CO₂ concentrations (e.g. Vachon et al. 2016a) resulting from the intense but physically confined soil respiration processes. However, once this CO₂-rich soil water enters surface waters (lake or stream), the excess gas can be released to 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 CO₂ 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 2013; Hotchkiss et al. 2015), and is probably not expected to influence the CO₂ balance of most reservoirs. The second main pathway through which aquatic CO₂ oversaturation arises is from *in situ* degradation by aquatic microbes and photo-chemical 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), while POC degradation is strongly regulated by temperature and oxygen availability (Gudasz and others 2010; Sobek and others 2009). An additional source of CO₂ may arise when a significant amount of methane produced under anaerobic conditions is oxidized to

CO₂ by methanotrophs prior to evasion to the atmosphere. While this indirect production is generally small relative to other CO₂ sources, it can be significant in some cases (Guerin 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 CO₂ 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. Quantifying the fraction of the CO₂ emissions that can be traced to these new sources of organic material is in itself difficult but is further confounded by the fact that the degradation dynamics of the natural OM sources is modified by altered hydrological regime.

Displaced CO₂ emissions: One of the most significant hydrological changes resulting from the impoundment of a river is the increased water residence time (τ_w). The original river segment located within the future reservoir area typically has a τ_w of a few hours while that of reservoirs is in the order of a few months (or days in the case of run-of-river impoundments). 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 τ_w (Dillon and Molot 1997; Algesten and others 2004). The net result is that proportionately more DOC will be turned into CO₂ at the reservoir site than would have in the pre-impoundment river, leading to a measurable system-wide increase in CO₂ emissions. In addition to the water residence time effect, emissions can be further accelerated at the reservoir site due to enhanced UV exposure, warmer temperatures or through a priming effect from elevated nutrients (Guérin and others 2008; Guenet and others 2010, but see Catalan 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 (Fig. 2); they are therefore largely invisible to the atmosphere as a 'new' GHG flux.

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 eq. 7) explicitly solves how the apparent reactivity of DOC will be affected by changes in τ_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 (τ_w in days) and the apparent DOC decay rate (k_{DOC} , in d^{-1}) for systems receiving inputs of a given reactivity (where α and ν are the average lifetime of the more reactive compounds and the relative abundance of the most recalcitrant compounds, respectively) as

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

At steady-state, the dynamic concentration equilibrium between a reservoir ($[DOC]_{res}$) and its inflowing waters ($[DOC]_{in}$) can be expressed as $[DOC]_{res} = [DOC]_{in} \left(\frac{1}{1 + \tau_w \cdot k_{DOC}} \right)$ and the resulting annual C mass lost to the atmosphere (moles yr^{-1}) is $Q \cdot [DOC]_{in} \cdot \left(\frac{\tau_w}{1 + \tau_w \cdot k_{DOC}} \right)$, 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

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

Equations 1 and 2 can be combined to assess the expected increase in CO_2 emissions resulting only from the hydrological changes created by the impoundment. Solutions of Eq. 2 for any reactivity of the inflowing riverine DOC (i.e. fixed α and ν) shows 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 ($\alpha=25$ days and $v=0.11$, see Vachon and others 2016b) converted to a reservoir with $\tau_w=90$ days will see its system-wide CO₂ footprint increase by about 130 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 CO₂ 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 CO₂ emissions from reservoirs: Although challenging, the exercise of assigning the fraction of CO₂ 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 the subtraction from the total emissions of the natural sources (e.g. de Faria and others 2015). Both methods have strengths and limitations. In the first, the potential new source of carbon is the flooded organic material, namely that contained in the soil and biomass. 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 have shown negligible decomposition after decades under water and even under tropical conditions (Guérin and others 2008, Cailleaud 2015), implying that their accounting as CO₂ sources is largely negligible except over long (centennial) time scales. 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 (e.g. Guérin and others 2008). Whether CO₂ 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 time scales in the terrestrial environment as well. Thus, the main remaining potential new source of carbon is the soil C that would not have otherwise been decomposed in the absence of flooding. It also constitutes the upper bound of the CO₂ 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, since 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 x 10⁻⁵ d⁻¹), then between 15 and 30% of the organic carbon mass will have been lost as CO₂ over that time period. Similarly, Guerin 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. While the production of CH₄ from flooded soils (discussed later) may not be important in terms of carbon mass, its higher global warming potential can nevertheless make it a significant GHG source.

The second approach to quantify the amount of CO₂ 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 CO₂ resulting only from the change in hydrological regime could theoretically be calculated using eqs. 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 CO₂ emissions surge immediately after impoundment and then quickly recede 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, in prep.) instead suggests that the exponential decline rate actually decreases through time (faster in the early years, slower later on) and that the temporal evolution of CO₂ emissions is better expressed as a general negative power function of the form $Flux\ CO_2 = C \cdot Age^{-0.29}$ where C is a reservoir-specific constant that depends on other factors such as temperature, nutrients, or reservoir area. The CO₂ 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 (e.g. Tremblay and others 2005), reflecting the new balance between primary production and respiration of the reservoir ecosystem. More conservatively, assuming instead that this new equilibrium is reached only after 100 years (Fig. 3)- a value often used often to represent the expected lifetime of reservoirs in life-cycle analysis (e.g. Gagnon and others 2002)- it would still imply that about 75% of the cumulative CO₂ flux is natural, i.e. that only 25% can be considered the result of the impoundment process¹. Thus, current assessments of the CO₂ footprint likely grossly overestimate the visible atmospheric impact of reservoir CO₂ emissions if natural CO₂ emissions are not discounted.

Methane

The case of CH₄ is conceptually easier than CO₂ in its attribution to the creation of the reservoir itself. Because the bulk of methanogenesis requires anaerobic conditions (but see Grossart and others 2011; Bogard and others 2014), the entire extent of the flooded area constitutes a new anoxic zone just below the soil-water interface that can be highly conducive to the production of CH₄ (e.g. Delmas and others 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

¹ Calculated as $\int_1^{100} C \cdot (Age^{-0.29} - 100^{-0.29}) dAge$

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 CH₄ emissions are new and therefore visible to the atmosphere.

While 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 (Fig. 1b) 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 2004; DelSontro and others 2010, 2011; Guerin and others 2006; Guerin and others 2016; 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 2016). 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, submitted). Some reservoir studies have demonstrated strong spatial heterogeneity of CH₄ 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 and largely unpredictable timing of bubble emission all contribute to make precise measurements of methane ebullitive fluxes poorly constrained. As an example, the most precise and spatially integrative method to measure CH₄ 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 CH₄ 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, submitted) are all important modulators of the average CH₄ emission rates in aquatic systems.

A feature that is unique to some reservoirs (mostly those constructed for hydro-electricity 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 CH₄ can accumulate and reach very high concentrations. When such CH₄-rich water passes through the turbines, CH₄ can be liberated directly to the atmosphere (degassing) or from the river surface below (downstream emission). In some cases (e.g. Abril and others 2005; Kemenes and others 2007), this degassing pathway constitutes the main mechanism through which GHG is released to the atmosphere. Although CO₂ is also released through this pathway, its contribution would be already taken into account if the visible CO₂ 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 CH₄ concentration profile, heterotrophic respiration, bathymetric shape and diffusion-limited O₂ 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 CH₄ 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 CH₄ 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.

Carbon sedimentation: mires and uncertainty

Aquatic ecosystems are unique 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 their catchments. Reservoirs, particularly in agricultural and other nutrient-rich areas, can accumulate large amounts of organic carbon in their sediments, (e.g. 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 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 either by 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 CO₂ relative to the atmosphere because the CO₂ assimilated via primary production will reduce its partial pressure. At equilibrium, the net CO₂ 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 CO₂ within the system. In this case, the CO₂ generated by heterotrophic respiration of the allochthonous OC would contribute to the water CO₂ partial pressure, thereby modulating the air-water pCO₂ gradient and its associated flux. This can be described by a simple CO₂ mass balance calculation. Except in very fast flushing lakes or reservoirs, the amounts of CO₂ 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 mass balance:

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

where $\Delta[\text{CO}_2]_L$ is the lake CO₂ concentration (expressed as a differential from atmospheric equilibrium), k_{600} is a gas exchange coefficient, A is surface area, V is volume, and GPP , R_{autoch} and R_{alloch} are, respectively, the areal rates of gross primary production, respiration of autochthonous OM, and respiration of allochthonous OM (negative corresponds to removal of CO₂). At steady-state, the rate of sedimentation of autochthonous OM (S_{autoch}) must equal the difference between GPP and R_{autoch} , and the equation simplifies to

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

showing that the sedimentation rate of autochthonous OM is necessarily reflected in the CO₂ 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 CO₂ at the air-water interface. Thus, measuring the net exchange at the air-water interface already accounts for the internal CO₂ fluxes and DOC transformation within the system, including the organic carbon buried via sedimentation of autochthonous material (algal or macrophyte production).

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 (Vörösmarty 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 (Sobek and others 2012; Maeck and others 2013). 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 hydrologic 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 (e.g. sediment sluicing) at the dam site.

Summary and areas of future research

We have outlined in this mini-review key components of the carbon cycle of reservoirs in order to develop a coherent framework 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 (e.g. Three-Gorges and Nam Theun 2 reservoirs). The few reported measurements have shown considerable variation, ranging from small to significant CH₄ sources (Serça and others 2016; Chen and others 2009; Yang and others 2012) and, in some cases, even CH₄ 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.

While uncertainties remain, our proposal emphasizes the need to quantify the GHG footprint of reservoirs on the basis of whether they 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 landscape transformation, thereby forming a basis for the accounting of the anthropogenic GHG footprint of reservoirs, such as those required by international regulatory agencies.

Figure legends

Figure 1. The 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 CO₂ and CH₄ emissions.

Figure 2. The concept of displaced emissions whereby mineralization of organic carbon are enhanced at the reservoir site due to the changes in water residence time following impoundment. These emissions would have occurred further downstream in the pre-impoundment conditions.

Figure 3. The decline in CO₂ emission over 100 years post-impoundment. Assuming conservatively that CO₂ emissions at 100 years corresponds to a new natural equilibrium, the hatched area is represents the integrated additional CO₂ efflux resulting from the creation of the reservoir.

Table 1.

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

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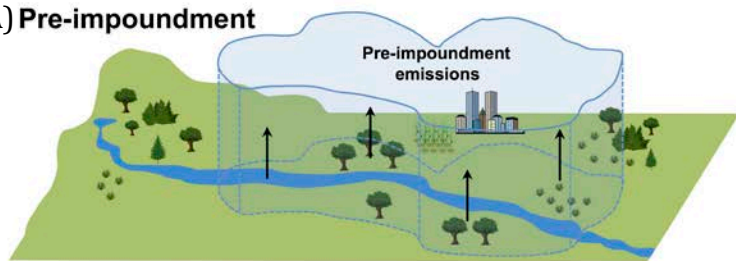
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Figure 1.

A) Pre-impoundment



B) Post-impoundment

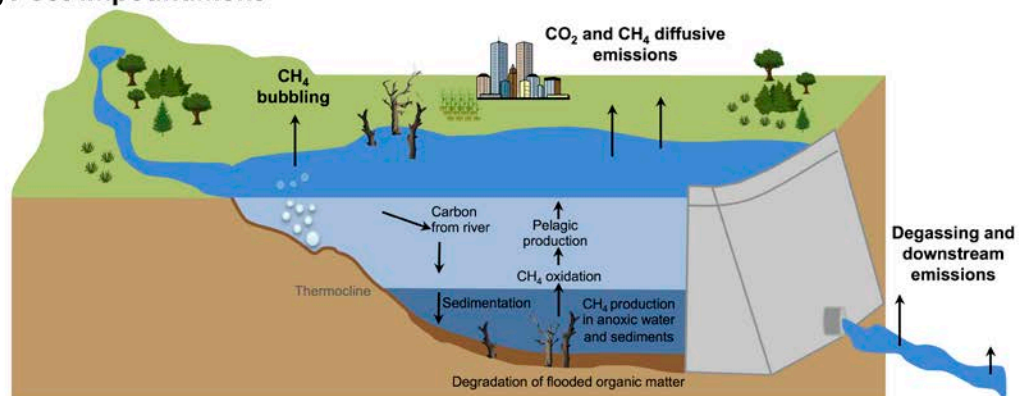


Figure 2.

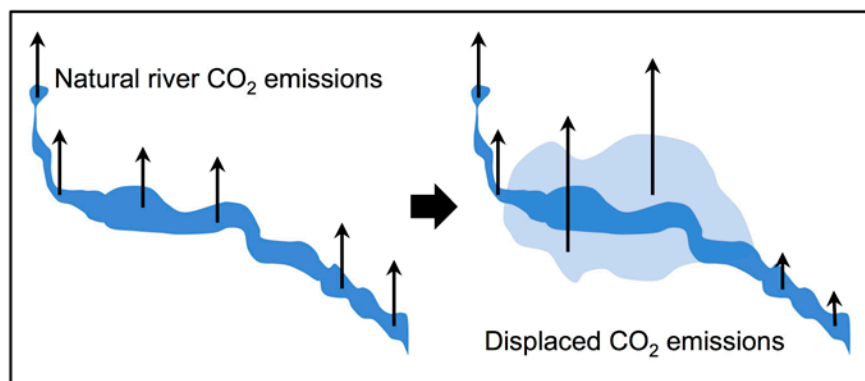


Figure 3.

