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# Do Hydroelectric Reservoirs Emit Greenhouse Gases?

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ABSTRACT / The fluxes of carbon dioxide were determined at the water–air interface using floating chambers connected to an automated nondispersive infrared instrument (NDIR) or a Fourier transform infrared instrument (FTIR). The fluxes were measured in 2002 over 280 sites in Canadian reservoirs, rivers, and natural lakes. Mean measured emissions of $CO_2$ in old Québec reservoirs (> 10 years) were around 1600 ± 1500 mg $CO_2/m^2$/day and around 735 ± 1125 mg $CO_2/m^2$/day in natural lakes. In a young Québec reservoir (5 years old), values were higher, with mean measured emissions around 4400 ± 4000 mg $CO_2/m^2$/day. In Manitoba, old reservoirs had mean values around 3350 ± 2725 mg $CO_2/m^2$/day and natural lakes values around 1365 ± 2375 mg $CO_2/m^2$/day. In British Columbia, mean values were around 250 ± 800 and 500 ± 650 mg $CO_2/m^2$/day, respectively, for old reservoirs and natural lakes. Our data suggest that water quality and the input of carbon from terrestrial systems affect $CO_2$ fluxes from waterbodies. Our results also show that reservoirs older than 10 years are comparable to natural lakes or rivers in terms of the water quality or gross $CO_2$ fluxes and that higher emissions in young reservoir would last 6–8 years.

Following flooding of terrestrial ecosystems by the creation of a reservoir, there is a modification of the chemistry of the flooded soils that leads to the release of labile carbon as well as nutrients to the water column (Schetagne 1994, Chartrand and others 1994). These elements enhance the bacterial activity and primary production, thereby stimulating the overall production of the reservoir ecosystem including plankton and fish communities (Chartrand and others 1994). These modifications occur over a short period of time in a boreal ecosystem, from 5 to 10 years, and the return to values representative of natural conditions is usually within 10–15 years according to the ratio of the land area flooded to annual volume (Chartrand and others 1994, Lucotte and others 1999). In addition, over the first few years, bacteria decompose a fraction of the organic carbon that was stored in the vegetation and in the soils, partly converting it to carbon dioxide ($CO_2$) and methane ($CH_4$). These gases will migrate in the water column, where $CH_4$ will be partly or totally oxidized (Duchemin and others 1995) according to the time of migration in the water column, which will then be released to the atmosphere (Kling and others 1992, St. Louis and others 2000). These greenhouse gases (GHG) may also be emitted from natural ecosystems (lakes, rivers, estuaries, beaver pond, etc.) (Kling and others 1992, Franken and others 1992, Carignan 1998, Makhov and Bazhin 1999).

KEY WORDS: $CO_2$; Greenhouse gases; Reservoirs; Lakes; Boreal

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There is a debate worldwide concerning the role of freshwater reservoirs in GHG emissions and their contribution to the increase of GHG in the atmosphere (Rosa and Scheaffer 1994, 1995, Fearnside 1996, Gagnon and van de Vate 1997, St. Louis and others. 2000). However, it must be pointed out that very few studies base their conclusions on direct systematic flux measurements at the water–air interface of reservoirs. Moreover, for those that have measured direct flux on reservoirs (Rudd and others 1993, Duchemin and others 1995, Duchemin and others 1999a), none have representative emissions of reservoirs and surrounding natural systems. The uncertainties about the gross flux (measured) as well as the net flux (gross flux minus preimpoundment natural emissions) are presently at the heart of the debate concerning methods of energy production.

Increasing the number of measurements will reduce significantly the uncertainties around a representative mean gross flux from natural systems as well as from reservoirs. In this study, we present representative gross flux of $CO_2$ from large waterbodies in Canada. We also propose, for the managers, a watershed life-cycle approach with field GHG flux measurements to compare adequately hydropower plant with other type of power plant.

## Materials and Methods

### Study Areas

Natural lakes and reservoirs were sampled during the ice-free period of 2002. The natural lakes and reservoirs of Manitoba, Ontario, and Québec (Figure 1)

 

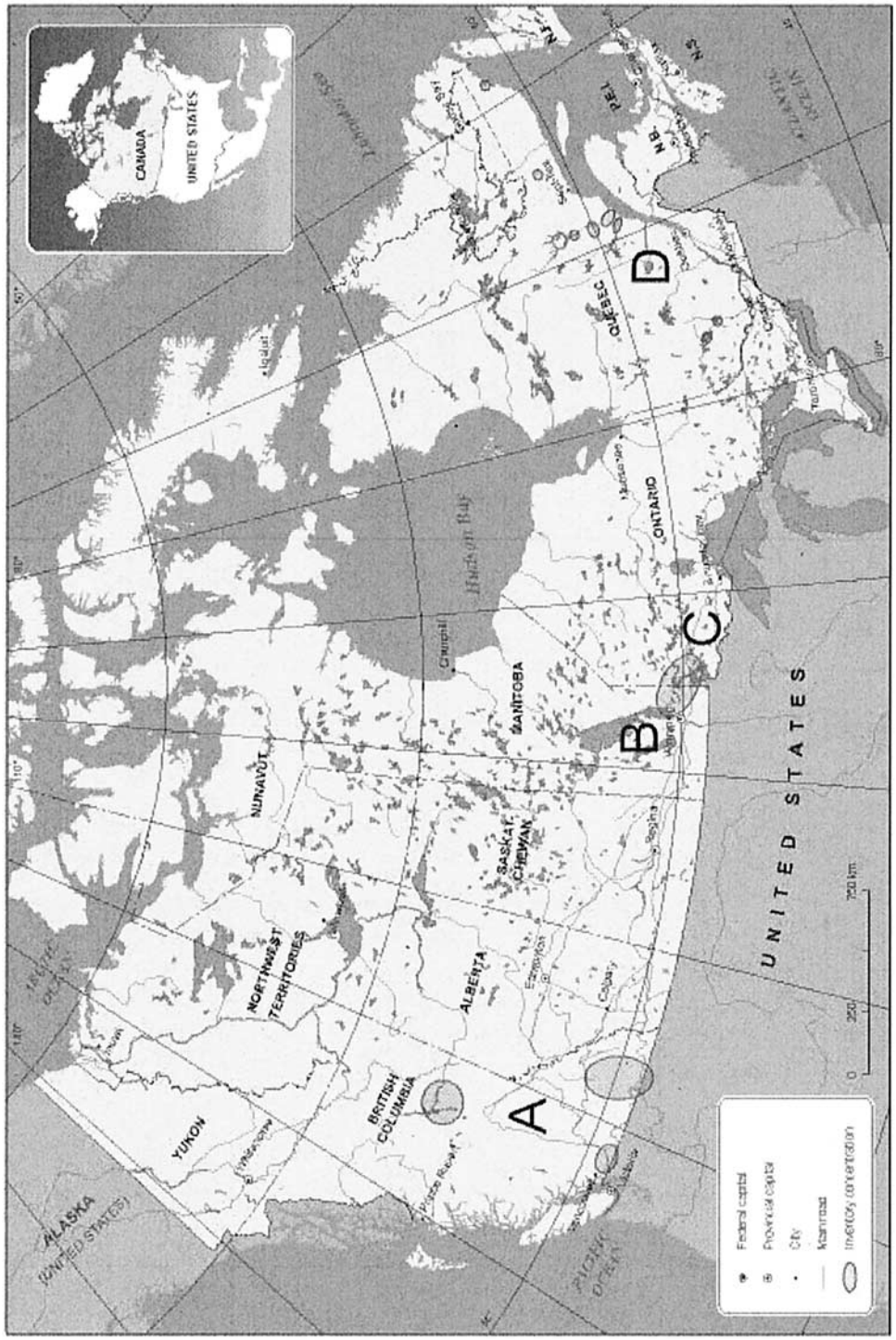


**Figure 1.** Location of the reservoirs and lakes sampled in British Columbia (A), Manitoba (B), Ontario (C), and Québec (D), Canada.

are oligotrophic brown-water systems with smooth, low-gradient slope catchments dominated by coniferous forest, and shallow podzolic and peat soils of the Canadian shield. The natural lakes and reservoirs of British Columbia (Figure 1) are oligotrophic clear-water systems with steep, high-gradient slope catchments dominated by coniferous and organic soils of the Canadian West Coast Mountains.

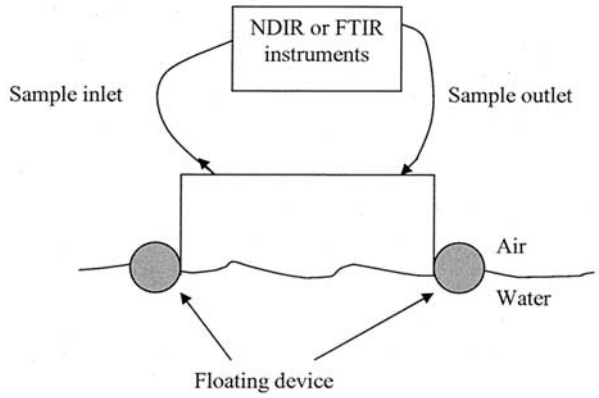


**Figure 2.** Design of the floating chambers for with NDIR and FTIR instruments (not to scale).

### Measurement of $CO_2$ Flux

The design of the floating chamber is very simple, and it is a modification of the Carignan (1998) technique. It is a one-piece Rubbermaid box (polyethylene container) with a surface of 0.2 $m^2$ and a height of 15 cm with about 2 cm below the surface of the water. The volume of air trapped over water is about 18 liters. The air is sampled on top of the floating chamber and returned to the chamber at the opposite end of the chamber. By locating the inlet and outlet at opposite ends of the chamber, the air trapped in the chamber is continuously mixed, which allowed a more representative measurement of the gas concentration (Guertin and others 2002). Figure 2 shows the design of that type of chamber. Air was analyzed with a NDIR non-dispersive infrared (NDIR) instrument (PP-System model Ciras-SC) or a Fourier transform infrared (FTIR) instrument (Temet, Gasmet DX-4010). The accuracy of the instrument in the 350–500 ppm range is ± 0.2–0.5 ppm for the NDIR and ± 3–5 ppm for the FTIR. Air is circulated in a closed loop, the instrument takes a continuous reading, and the data logger (Campbell Scientific, model CR10X) stores a value every 20 sec over a period of 5–10 min. All samples are plotted on a graph and the slope is calculated in parts per million per second (Figure 3). The flux is calculated with equation 1.

$$\text{Flux} = \frac{\text{slope} \times \text{F1} \times \text{F2} \times \text{volume}}{\text{surface}}$$

where slope is the slope from the graph of concentration versus time in ppm/s; F1 is the conversion factor from ppm to mg/$m^3$; F2 is the conversion factor from second to day (86, 400); volume is the volume of air trapped in the chamber (0.018 $m^3$); surface is the surface of the floating chamber over the water (0.204 $m^2$); and flux is the mg/$m^2$/day.

At each site, a minimum of two fluxes were measured. We also measure GPS position, water temperature, color, transparency, and depth, pH (Beckman 200 and Oakton 300 with a Ross type electrode from Orion), alkalinity, air temperature, and wind velocity (Extech Instruments 407112). The number of sites sampled in each waterbody depended on factors such as accessibility, size of the waterbody, and variability of ecosystems within the waterbody. As a result, small waterbodies (generally lakes) have few sites sampled and bigger waterbodies (generally reservoirs like Robert-Bourassa in Québec or Williston in British-Columbia) with different ecosystems have more sites sampled. Each waterbody was sampled within a two-day period, with the majority sampled within a single day.

The floating-chambers technique is widely used for measurement of GHG fluxes over waterbodies (Duchemin and others 1995, 1999a, St. Louis and others 2000). Some authors have compared the floating chambers with the thin boundary layer method for estimating $CO_2$ fluxes and have found that the floating chambers overestimate the fluxes (Duchemin and others 1999a, Matthews and others 2003). Parameters such as meteorology or floating-chamber design seem to affect the rate of overestimation. Other measurement systems, such as eddy correlation, allow the measurement of integrated fluxes over a larger area and over a longer period of time, but cannot be easily installed in remote areas or transported from site to site.

In our study, the fluxes measured with floating chambers were done in the daytime for logistical reasons. In 2004, we plan to measure fluxes on a 24-hr basis at some sites. This will allow us to evaluate the daily cycle of $CO_2$ fluxes and help us to obtain a more representative mean for a water body.

Because of the relatively short period of time needed to measure a flux and the ease with which our system can be transferred from one site to the other, it is possible to determine the spatial variations between habitats of the same ecosystem over a short period of time.

## Results and Discussion

The mean gross emissions of $CO_2$ measured in Québec in 2002 showed values varying from −308 to 1850 mg $CO_2/m^2$/day for the natural lakes and values of 980–3300 mg $CO_2/m^2$/day for all reservoirs except Sainte-Marguerite (Figure 4, Table 1). Overall mean values of $CO_2$ are 1600 ± 1500 and 725 ± 1125 mg $CO_2/m^2$/day, respectively, for reservoirs and natural lakes. Considering the standard deviation, reservoirs older than 6 years have emissions of $CO_2$ comparable to the surrounding natural lakes or rivers. These emis-

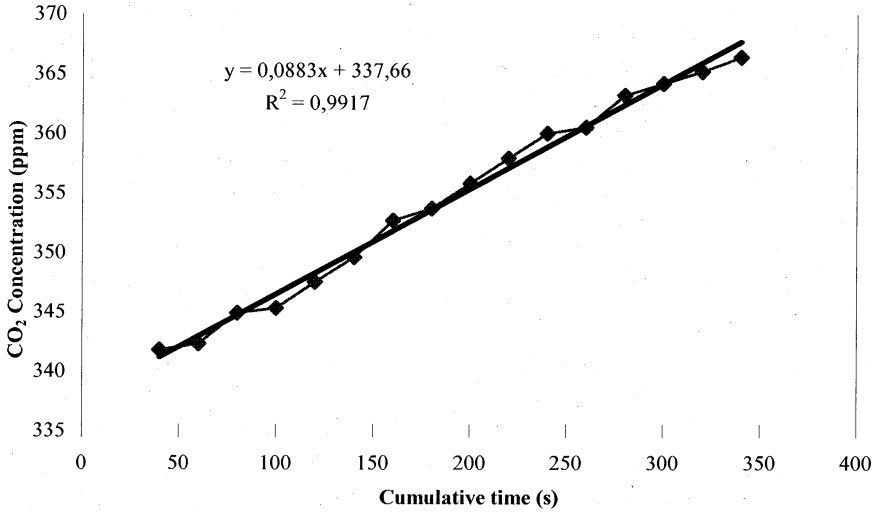


**Figure 3.** Typical graph of results with NDIR or FTIR instruments.

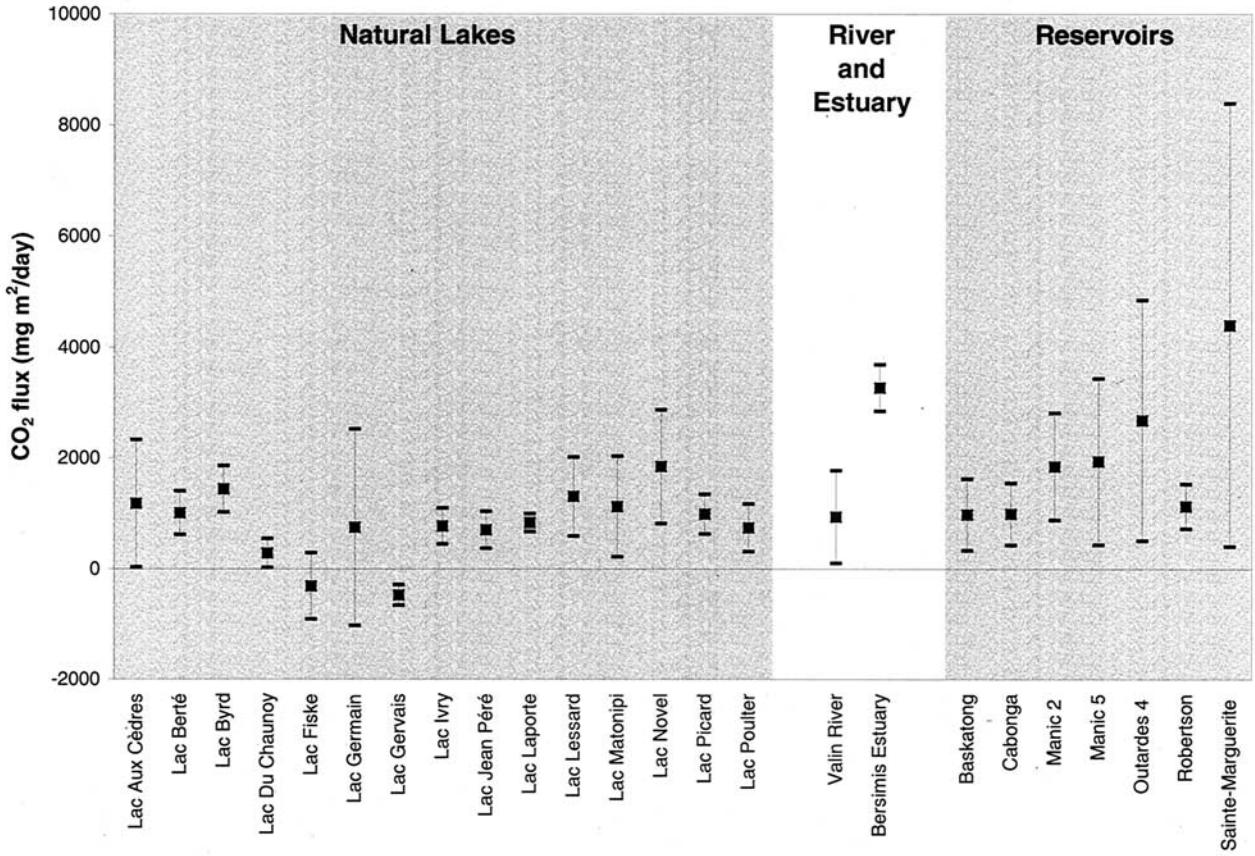


**Figure 4.** Mean value and standard deviation of gross emission of $CO_2$ at the water–air interface with concentration of $CO_2$ in Québec.

sions are similar to those reported by Duchemin and others (1995, 1999b) over boreal reservoirs as well as those reported by Tremblay and others (2001), Lambert and others (2001) and Lambert (2002) from Québec's lakes and reservoirs. Similarly, in Finland, the mean $CO_2$ emissions in Lokka and Porttipahta reser-

Table 1. $CO_2$ flux and pH[a] of reservoirs and natural lakes measured in Québec

| | Age of the reservoir (yr) | pH (range) | Flux (mg/m$^2$/day, mean ± SD) | Measurements (*N*) |
|---|---|---|---|---|
| LAKES | | | | |
| Lac Aux Cèdres | | 6.9–7.0 | 1181 ± 1149 | 5 |
| Lac Berté | | 6.2–6.6 | 1010 ± 392 | 14 |
| Lac Byrd | | 6.3–6.4 | 1441 ± 419 | 10 |
| Lac Du Chaunoy | | 7.0–7.1 | 288 ± 262 | 5 |
| Lac Fiske | | 8.4–8.5 | −308 ± 599 | 7 |
| Lac Germain | | 5.9–6.6 | 750 ± 1773 | 15 |
| Lac Gervais | | 8.2–8.3 | −468 ± 187 | 7 |
| Lac Ivry | | 6.0–6.1 | 772 ± 325 | 4 |
| Lac Jean Péré | | 6.1–6.2 | 705 ± 332 | 15 |
| Lac Laporte | | 6.1–6.3 | 834 ± 161 | 5 |
| Lac Lessard | | 6.5–6.6 | 1307 ± 712 | 5 |
| Lac Matonipi | | 6.2–6.3 | 1127 ± 907 | 7 |
| Lac Novel | | 5.6–6.4 | 1846 ± 1026 | 8 |
| Lac Picard | | 5.4–5.5 | 988 ± 357 | 7 |
| Lac Poulter | | 6.0–6.1 | 745 ± 427 | 7 |
| Rivers and estuaries | | | | |
| Valin River | | 4.9–6.2 | 938 ± 834 | 6 |
| Bersimis Estuary | | 6.6–6.7 | 3270 ± 421 | 3 |
| Reservoirs | | | | |
| Baskatong | 70 | 5.9–6.5 | 980 ± 642 | 22 |
| Cabonga | 75 | 5.8–6.3 | 991 ± 558 | 60 |
| Manic 2 | 31 | 6.3–6.4 | 1846 ± 967 | 14 |
| Manic 5 | 38 | 6.2–6.8 | 1938 ± 1500 | 41 |
| Outardes 4 | 32 | 6.3–6.5 | 2682 ± 2171 | 26 |
| Robertson | 5–7 | 6.2–6.2 | 1131 ± 403 | 22 |
| Sainte-Marguerite | 2–4 | 5.2–6.7 | 4399 ± 3993 | 89 |

[a]or pH, the range represents in situ measurements made at different stations.

voirs, during the open water season, are of the same order of magnitude: 1700 and 3400–4000 mg $CO_2$/m$^2$/day, respectively (Kortelainen 1998, Martikainen and others 1996).

For a young reservoir such as Sainte-Marguerite (5 years old), the emissions were 4400 ± 4000 mg $CO_2$/m$^2$/day (Figure 4, Table 1). In the Laforge-1 reservoir, in northern Québec, Duchemin and others (1995) have measured emissions of $CO_2$ varying from 380 to 6700 mg $CO_2$/m$^2$/day in the first 2 years after flooding. These higher emissions observed in young reservoirs probably originate from the bacterial decomposition of the labile carbon of the flooded soils and the green part of the vegetation (Guertin and others 2002). Since the loss of nutrients and labile carbon in the flooded soils is relatively rapid, less than 5–10 years, the remaining carbon is basically lignin, which is refractory to decomposition, leading, after 10 years of flooding, to emissions comparable to those of natural lakes.

In a small bay formed after the construction of dike 24 of the Robert-Bourassa reservoir, Duchemin and others (1995) observed exceptional emissions of $CO_2$ varying from 200 to 6000 mg $CO_2$/m$^2$/day even 18–20 years after flooding. Although dike 24 was flooded in 1978, the results are comparable to a young reservoir because this bay has very little water exchange with the major waterbody of Robert-Bourassa reservoir, the soils are still mostly intact (little erosion), and the trees are still in place after 20 years. This situation contributes to maintaining high bacterial activity by the recycling of nutrients and carbon year after year and maintaining high emissions of $CO_2$ at the water–air interface. Therefore, the results from dike 24 are not representative of a large reservoir and can not be extrapolated to large reservoir. This raises the issue of using representative GHG fluxes to estimate the contribution of a reservoir.

The mean gross emissions of $CO_2$ measured in Manitoba and Ontario showed values varying from −112 to 4430 mg of $CO_2$/m$^2$/day for the natural lakes and values from 2000 to 5550 mg $CO_2$/m$^2$/day for the reservoirs (Figure 5, Table 2). The reservoirs sampled in Manitoba are located in cascade on the Winnipeg River system. Lake of the Woods and Shoal Lake are two very large lakes, with an immense catchment area–drainage ratio, that feed the Winnipeg River system (Figure 5). These two lakes have mean gross emissions of $CO_2$ very similar to the reservoirs in the Winnipeg River system, with values varying from 2000 to 6500 mg $CO_2$/

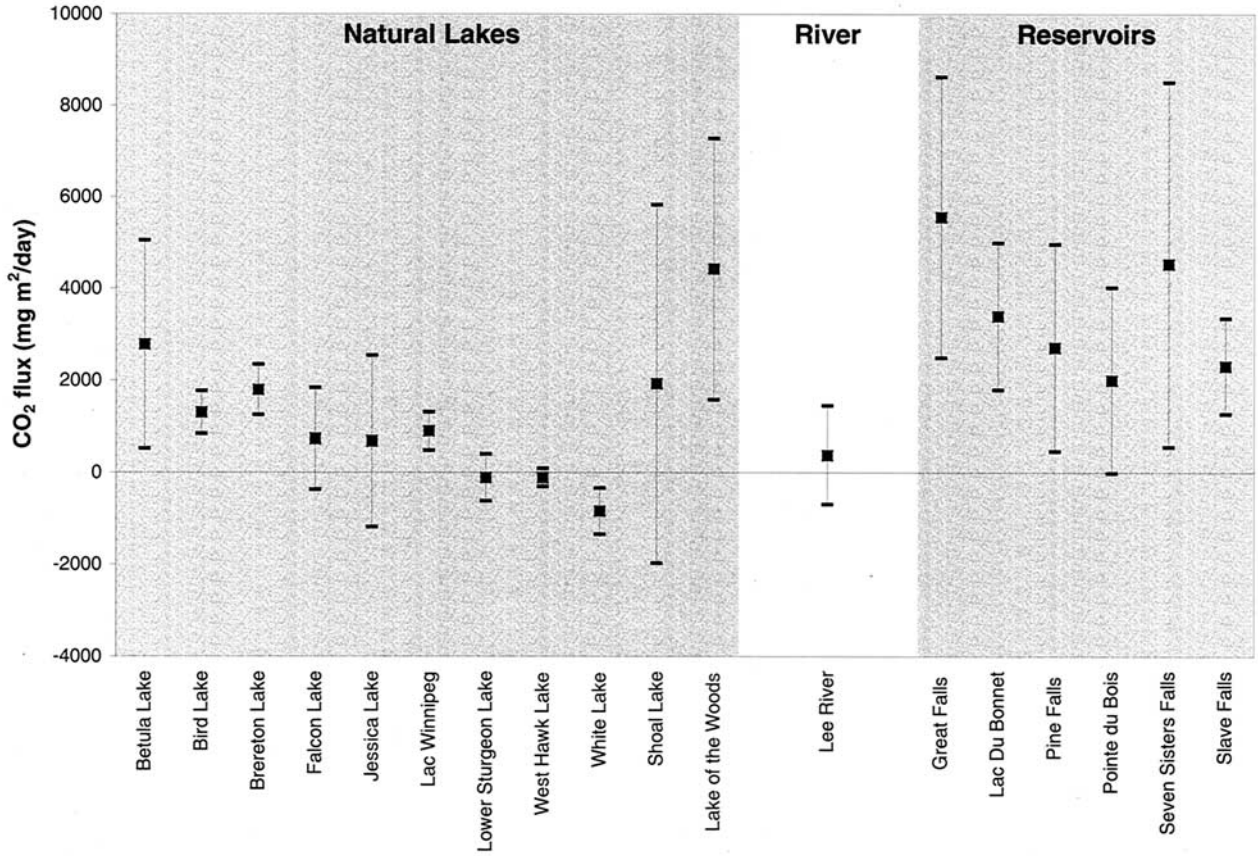


**Figure 5.** Mean value and standard deviation of gross emission of $CO_2$ at the water–air interface with concentration of $CO_2$ in Manitoba and Ontario.

Table 2. $CO_2$ flux and pH[a] of reservoirs and natural lakes measured in Manitoba and Ontario

| | Age of the reservoir (yr) | pH (range) | Flux (mg/ $m^2$/day, mean ± SD) | Measurements (*N*) |
|---|---|---|---|---|
| Lakes | | | | |
| Betula Lake | | 7.3–7.8 | 2783 ± 2262 | 7 |
| Bird Lake | | 7.3–7.4 | 1300 ± 468 | 8 |
| Brereton Lake | | 7.1–7.2 | 1795 ± 549 | 8 |
| Falcon Lake | | 8.1–8.2 | 732 ± 1104 | 8 |
| Jessica Lake | | 7.9–8.3 | 675 ± 1866 | 11 |
| Lac Winnipeg | | 8.3–8.5 | 894 ± 412 | 12 |
| Lower Sturgeon Lake | | 7.5–7.8 | −112 ± 512 | 5 |
| West Hawk Lake | | 7.9–8.1 | −111 ± 200 | 8 |
| White Lake | | 8.1–8.3 | −840 ± 503 | 8 |
| Shoal Lake | | 8.0–8.4 | 1929 ± 3902 | 12 |
| Lake of the Woods | | 7.5–7.7 | 4430 ± 2855 | 15 |
| River | | | | |
| Lee River | | 7.8–8.7 | 374 ± 1072 | 9 |
| Reservoirs | | | | |
| Great Falls | 73 | 7.3–7.7 | 5559 ± 3068 | 10 |
| Lac Du Bonnet | 47 | 7.4–7.8 | 3395 ± 1601 | 20 |
| Pine Falls | 50 | 7.7–8.2 | 2712 ± 2252 | 14 |
| Pointe du Bois | > 35 | 7.5–8.3 | 2000 ± 2019 | 10 |
| Seven Sisters Falls | 71 | 7.1–7.6 | 4533 ± 3979 | 16 |
| Slave Falls | > 35 | 7.6–7.8 | 2309 ± 1042 | 8 |

[a]For the pH, the range represent in situ measurements made at different stations.

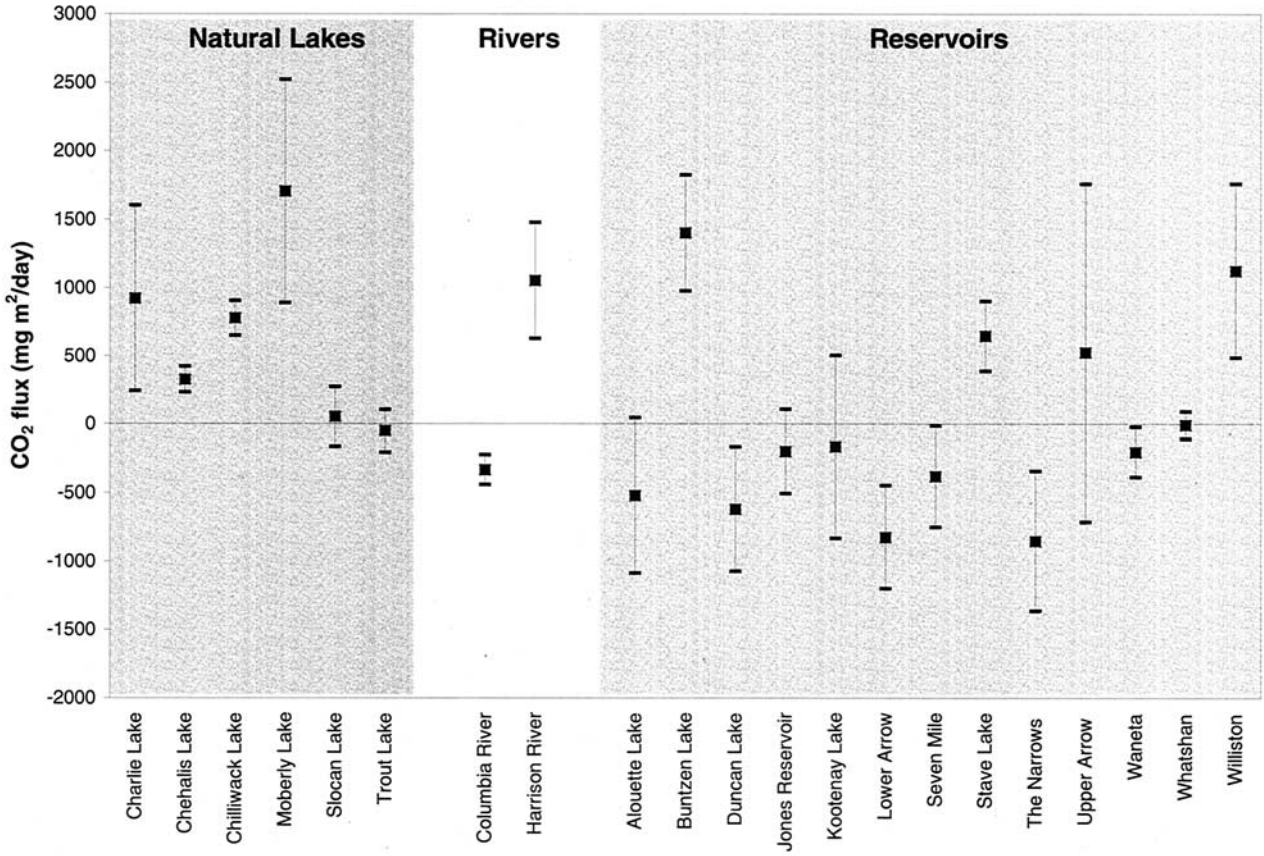


**Figure 6.** Mean value and standard deviation of gross emission of $CO_2$ at the water–air interface with concentration of $CO_2$ in British Columbia.

$m^2$/day. This indicates that the reservoirs may be affected by the input of terrestrial carbon since they are emitting as much as the lakes feeding them. Similarly, Aronsen and others (2002) have observed a positive relationship ($R^2$ of 0.88) between the lake drainage ratio and the loss of carbon from the lakes, indicating a higher loss of carbon from the lakes with bigger catchment area or higher export of terrestrial carbon to the lakes.

The mean gross emissions of $CO_2$ measured in British Columbia showed values varying from −50 to 1706 mg $CO_2/m^2$/day for the natural lakes, from −333 to 1053 mg $CO_2/m^2$/day in rivers, and from −852 to 1402 mg $CO_2/m^2$/day for the reservoirs (Figure 6, Table 3). It is quite clear that both natural lakes and reservoirs in this area of Canada can either emit or absorb $CO_2$. The waterbodies in British Columbia are much deeper (> 50 m), are composed of clear water, and are much less productive than those from Manitoba or Québec, where the mean depth is around 5–10 m with brown water. The pH is also generally higher in British Columbia waterbodies, favoring the formation of bicarbonate instead of the $CO_2$ dissolved in the water (Clark and Fritz 1997). These results indicate that the water quality and the productivity of the systems may influence the flux of $CO_2$ either in or out of the waterbody. Aronsen and others (2002) have observed lower emissions of $CO_2$ in lakes with low total organic carbon (TOC) than lakes with high TOC.

Similarly, Kortelainen (1998) has measured $CO_2$ emissions over 175 natural Finnish lakes; all the lakes studied were supersaturated with respect to $CO_2$. The highest supersaturation was found in small eutrophic lakes and lower supersaturation in the largest and deepest lakes. Moreover, in a literature review Therrien (2003) observed that gross $CO_2$ emissions generally increase from oligotrophic lakes to eutrophic lakes.

Freshwater lakes, rivers, and reservoirs have been suggested to play an important role in the transfer of terrestrial-fixed carbon to the atmosphere, although they account for less than 0.4% of the earth's surface (Wetzel 1975, Hope and others 1996). Moreover, according to Aronsen and others (2002), $CO_2$ fluxes from natural freshwater lakes of a boreal watershed are the major factor in removing 30%–70% of terrestrial carbon exported from the watershed. The present study and oth-

Table 3. $CO_2$ flux and pH[a] of the reservoirs and natural lakes measured in British Columbia

| | Age of the reservoir (yr) | pH (range) | Flux (mg/m$^2$/day, mean ± SD) | Measurements (*N*) |
|---|---|---|---|---|
| Lakes | | | | |
| Charlie Lake | | 8.1–8.2 | 922 ± 681 | 8 |
| Chehalis Lake | | 7.0–7.2 | 326 ± 95 | 6 |
| Chilliwack Lake | | 7.2–7.4 | 777 ± 128 | 10 |
| Moberly Lake | | 8.0–8.2 | 1706 ± 816 | 10 |
| Slocan Lake | | 7.8–8.0 | 53,9 ± 217 | 10 |
| Trout Lake | | 7.9–8.0 | −50,5 ± 156 | 6 |
| Rivers | | | | |
| Columbia River | | 8.3–8.4 | −333 ± 109 | 2 |
| Harrison River | | 7.3–7.5 | 1053 ± 425 | 6 |
| Reservoirs | | | | |
| Alouette Lake | 74 | 6.9–7.2 | −520 ± 566 | 10 |
| Buntzen Lake | 88 | 6.3–6.4 | 1402 ± 424 | 8 |
| Duncan Lake | 37 | 8.3–8.5 | −619 ± 454 | 8 |
| Jones Reservoir | 50 | 7.7–7.8 | −199 ± 307 | 8 |
| Kootenay Lake | > 20 | 7.9–8.6 | −163 ± 668 | 25 |
| Lower Arrow | 33 | 9.0–9.1 | −823 ± 376 | 6 |
| Seven Mile | 23 | 8.4–8.5 | −381 ± 368 | 10 |
| Stave Lake | 91 | 6.7–6.9 | 645 ± 257 | 18 |
| The Narrows | 33 | 8.5–9.0 | −852 ± 511 | 7 |
| Upper Arrow | 33 | 7.2–8.7 | 524 ± 1236 | 21 |
| Waneta | > 20 | 8.3–8.4 | −204 ± 182 | 9 |
| Whatshan | 51 | 7.7–7.9 | −8 ± 99 | 7 |
| Williston | 41 | 8.0–8.4 | 1125 ± 637 | 49 |

[a]For pH, the ranges represent in situ measurements made at different stations.

ers (Cole and Caraco 2001) confirm that natural lakes, rivers, and estuaries are naturally significant emitters of $CO_2$ and $CH_4$. Therefore, to correctly estimate GHG emissions from reservoirs, it is essential to determine the emissions from the various ecosystems in the watershed before and after creation of the reservoir. This will make it possible to calculate the net emissions for which the reservoir is responsible. Emissions measured directly on reservoirs must be regarded as "gross" emissions.

In this watershed approach, total gross emissions can be estimated by extrapolating emissions per square meter to the whole reservoir. This estimate should assign a specific emission factor (per square meter) for each zone of a reservoir, depending on the age of reservoir, the type of ecosystem flooded, depth, currents, etc. The area of each zone should be quantified and multiplied by the appropriate emission factor.

Preimpoundment emissions must also be determined by multiplying the area of each flooded ecosystem by appropriate natural emission factors. Obvious ecosystems to consider are wetlands and peatlands, which are well-known sources of methane. The assessment should also include the estuary of the concerned river, because it may have conditions similar to those of reservoirs and favor a higher level of emissions (reduced current speed, higher aquatic productivity than the river, etc). At the international level, very few measurements of GHG emissions on estuaries seem to have been conducted. A few measurements have been made by Hydro-Québec on the Bersimis estuary (see Table 1); they show levels of GHG emissions higher (per square meter) than those of reservoirs. The assessment of this portion of a watershed is important, because creating a reservoir may produce the following changes: increase of GHG emissions at the site of the reservoir, reduction of carbon flowing downstream, and reduction of emissions at the estuary.

Finally, to assess the performance of a hydroelectric power plant, in terms of GHG emissions per kWh, it is necessary to determine net reservoir emissions over the life of the plant. Examining similar reservoirs of different ages can be the method to estimate long-term emissions that should then be divided by expected generation (kWh) over the period (most life-cycle assessments use 100 years as the normal life of a hydroelectric power plant).

In conclusion, the approach proposed here, the definition of representative net GHG flux from reservoirs based on a whole watershed approach, would allow a fair comparison of hydropower emissions with those of thermal generation.

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