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Mineralization rates of peat from eroding peat islands in reservoirs

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Abstract. Reservoirs are sources of greenhouses gases to the atmosphere, primarily due to organic carbon mineralization in flooded plants and soils to carbon dioxide (CO_2) and methane (CH_4). Floating peat islands are common in reservoirs that inundated peatlands. These islands can decompose on mass, or small pieces of peat can erode from islands to decompose in the water column or on the bottom of reservoirs. Here we used large 450 liter sealed enclosures to measure mineralization rates of small peat pieces and larger peat blocks collected from floating peat islands. Mineralization rates were calculated by quantifying dissolved inorganic carbon (DIC), CO_2 and CH_4 accumulation within the water and headspace of the enclosures over time. We found that peat did decompose under water, but rates of mineralization of peat pieces were not different than rates of mineralization of larger peat blocks. Mineralization rates ranged between 59 and l40 μg C g^{-1} d^{-1}. Peat pieces acidified the water, shifting the bicarbonate equilibrium to almost exclusively dissolved CO_2, which was then readily able to flux to the atmosphere. We estimated that 2.4–5.6% of peat carbon was mineralized annually, suggesting that fluxes of CO_2 and CH_4 from reservoirs that flood peatlands could last at minimum 18–42 years from this carbon source alone.

Introduction

Reservoirs are known to be sources of the greenhouse gases (GHGs) carbon dioxide (CO_2) and methane (CH_4) to the atmosphere primarily due to the decomposition (mineralization) of flooded organic matter found in soils and vegetation (e.g., Rudd et al. (1993), Kelly et al. (1993), Duchemin et al. (1995)). The net impact of reservoir creation on GHG emissions to the atmosphere also includes the loss of net primary production following the death of flooded vegetation. It has been estimated that globally there are more than 1.5 million km^2 of reservoir surface area emitting 7.4 GT C yr^{-1}, or 4% of other anthropogenic emissions (St.Louis et al. 2000). Because many of the preferred steep-walled canyon sites have already been developed, vast areas of land are now being flooded to create new reservoirs. In northern reaches of Canada, Scandinavia and Russia, for example, low-lying peatlands are the first part of the landscape to be inundated. Peatlands are thought to be

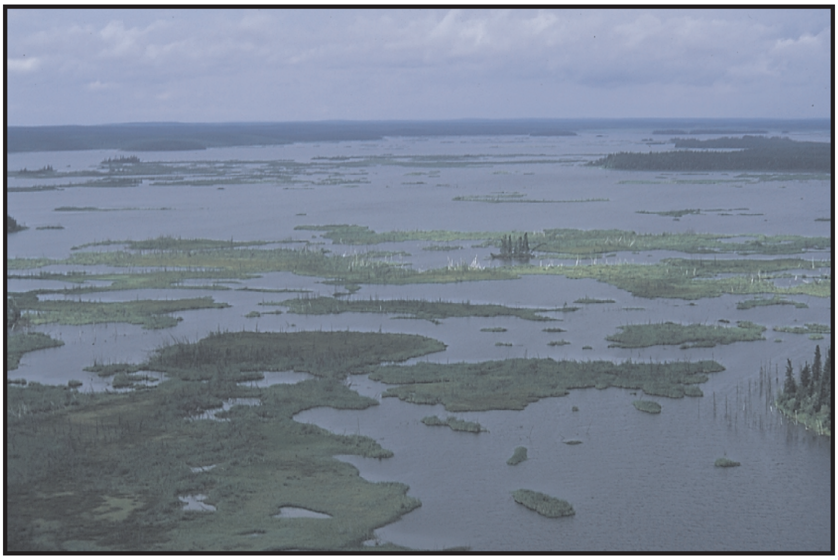

Figure 1. Floating peat islands in a northern Canadian reservoir.

the worst-case scenario for flooding due to the large stores of organic carbon in peat available for mineralization to CO_2 and CH_4, and the elimination of net primary production. Reducing the amount of organic carbon flooded should reduce the amount of GHGs emitted from reservoirs on the long-term (Kelly et al. 1997).

A common feature in flooded peatlands is floating peat islands (Ronka and Uusinoka 1976) (Figure 1). When peatlands are inundated, rates of CH_4 release increase because methanogenic bacteria favour anoxic environments and flooding reduces or removes the zone of oxidation that usually is present in undisturbed peatlands (Scott et al. 1999). Also, elevated temperatures that occur within inundated peat, due to greater heat conductance in water than air, enhances rates of microbial activity (Kelly et al. 1997). The relatively insoluble CH_4 accumulates as bubbles in the peat and eventually causes sections of the peatland to float to the surface forming peat islands (Koskenniemi 1987). Once peat is floating, the surface may become a large emitter of CH_4 because the zone of production is in direct contact with the atmosphere, not allowing for microbial CH_4 oxidation (Scott et al. 1999).

A boreal wetland was experimentally flooded at the Experimental Lakes Area (ELA) in northwestern Ontario in 1993 to quantify *net* GHG emissions from reservoirs (Kelly et al. 1997). Before flooding, the wetland was composed of a 2.4 ha central pond surrounded by a 14.4 ha peatland. The organic peat layer was on average 1.7 m thick. The wetland was flooded to a depth of 1.3 m above the previous pond level by damming the outflow. The flooding increased the surface area of the pond by a factor of three and the water volume by a factor of six. The ELA Reservoir Project (ELARP) showed that the wetland went from being a natural carbon sink of 6.6 g C m^{-2} yr^{-1} to a large source of 130 g C m^{-2} yr^{-1} to the atmosphere

(Kelly et al. 1997). Based on rates of decomposition measured in litterbags, most, but not all, of the gas flux produced in the first two years after flooding resulted from the decomposition of flooded vegetation (Kelly et al. 1997). We therefore hypothesized that some portion of the gas flux was likely derived from microbial activity in the peat matrix. Over 33% of the flooded peatland area was covered by floating peat up to one meter thick by the second year of flooding (Kelly et al. 1997), whereas by the fifth year there was floating peat in almost all areas of the inundated peatland. While we know that these floating peat island surfaces contribute to CH_4 fluxes to the atmosphere (Scott et al. 1999), the contribution of the submerged portions of these islands to buildup of CO_2 and CH_4 in the reservoir water has not been measured. Peat in general is considered to be fairly refractory carbon because on average it is composed of about 50% lignin or phenolics (Williams et al. 1998; Turetsky et al. 2000). It has been shown to decompose under water at the bottom of naturally forming shallow ponds on the Hudson Bay Lowland (Hamilton et al. 1994). The study on natural peat ponds hypothesized that photosynthetic nitrogen-fixing mats may have played a role in the peat decomposition. In the study presented here, we ask whether peat decomposes at a significant rate under water, in the dark, as occurs during reservoir creation and disturbance of peatlands.

When large sections of peat float to surfaces of reservoirs, two things happen that may enhance their rates of decomposition. First, small pieces of peat erode from the main peat body due to wind and wave action and become exposed to an aerobic decomposition environment in the open water, whereas the interiors of peat islands remain largely anoxic (McKenzie et al. 1998). Second, heat conductance into the floating peat is increased because the underside of peat islands are exposed to water (Kelly et al. 1997). Here we report on an experiment to determine if (1) peat decomposes under dark, submerged conditions, and (2) small peat pieces decompose at different rates than intact blocks of peat.

Methods

General experimental design

Organic carbon mineralization was quantified in sealed, dark enclosures containing water and a headspace, and either (1) a large peat block, (2) an equivalent quantity of small pieces of peat, or (3) no peat as a control to quantify net dark respiration in the water column. Each treatment was duplicated. Dissolved inorganic carbon (DIC), CO_2 and CH_4 production were quantified in the water and headspace of each enclosure. The experiment was replicated three times between 14–19 July (Run 1), 31 July–5 August (Run 2) and 17–21 August 1998 (Run 3), using freshly collected peat for each run.

Table 1. Dry weight of peat pieces and blocks added to enclosures, and mean final temperature, pH and O_2 concentrations in the enclosures. Values for replicate enclosures are in parentheses.

	Dry Wt. Peat[1] (g)	Water Temperature (°C)[2]	Final pH	Final O_2 (mg l^{-1})
Run 1 (14–19 July)				
Peat pieces	487 (520, 454)	22.7 ± 0.7	–	5.1 (5.1, 5.1)
Peat blocks	487 (520, 454)	22.7 ± 0.6	–	6.3 (6.5, 6.0)
Water	–	22.8 ± 0.6	–	6.7 (6.7, –)
Run 2 (31 July–5 August)				
Peat pieces	525 (592, 457)	21.8 ± 1.3	5.1 (5.1, 5.2)	4.9 (4.8, 4.9)
Peat blocks	525 (592, 457)	21.7 ± 1.3	6.3 (6.4, 6.3)	6.7 (6.2, 7.1)
Water	–	21.6 ± 1.3	6.7 (6.7, 6.8)	9.4 (9.2, 9.5)
Run 3 (17–21 August)				
Peat pieces	553 (640, 466)	19.0 ± 0.7	5.4 (5.5, 5.3)	5.4 (5.5, 5.3)
Peat blocks	553 (640, 466)	19.3 ± 0.5	6.3 (6.2, 6.4)	6.3 (6.6, 5.9)
Water	–	19.2 ± 0.4	6.7 (6.8, 6.6)	7.7 (7.5, 7.9)

[1] Each run had four treatment enclosures, with groups of two containing comparable weights of pieces and blocks.

[2] Mean average temperature and mean SD for replicate enclosures.

Peat collection

Saturated peat was cut the day we began each experimental run from a five-yr-old floating peat mat in the ELARP experimental reservoir at the ELA. Live *Sphagnum* spp. was removed from the surface of the peat. For each experimental run, four blocks of peat (approximately 30×30×15 cm) ranging in dry weight from 450 to 640 g (Table 1) were transported waterlogged back to the laboratory. Two intact blocks were wrapped in fine-mesh netting while two blocks were broken into individual peat pieces. Total wet weight of pieces added to enclosures was adjusted to match that of paired blocks.

Experimental apparatus

Six rigid plastic enclosures were placed in a raft anchored in the sheltered southeast basin of Lake 239 at the ELA where they could be filled with oxygenated water (Figure 2). Treatments were randomly distributed amongst enclosures. Pieces of peat were dispersed into the water of enclosures. Wrapped blocks of peat were initially lowered just below the water surface. Samples for internal porewater DIC and dissolved CH_4 analyses were taken by inserting a long needle to various depths in blocks to withdraw porewater into a 5 ml glass syringe before blocks were then anchored to the bottom of enclosure with a small granite rock. Porewater samples were injected into evacuated 13 ml bottles backfilled with 10 mls of UHP nitrogen and 0.1 ml of 10% HCl.

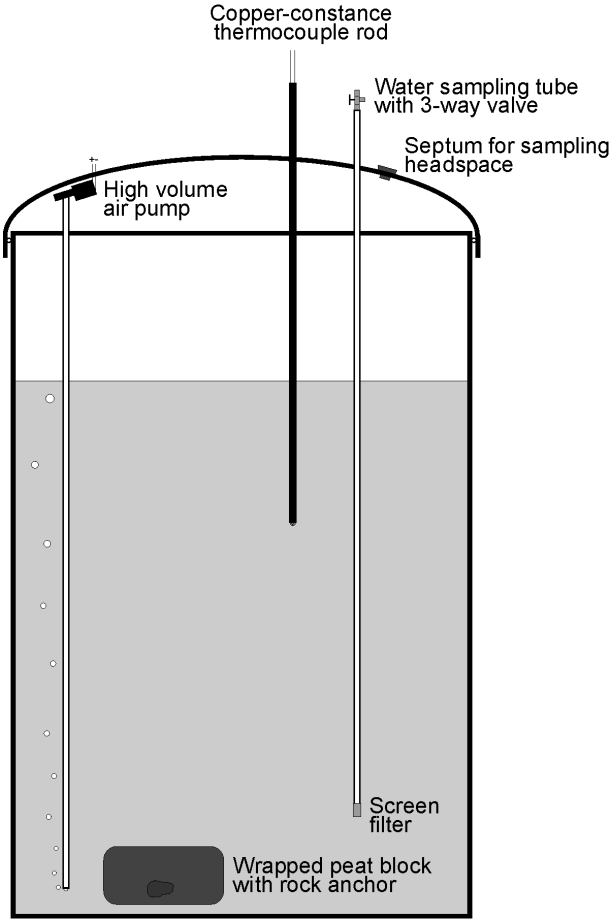

Figure 2. Cross-section of enclosure containing a peat block. Each enclosure was 75 cm wide and 117 cm deep, and impregnated during fabrication with black dye to inhibit light penetration. This prevented CO_2 assimilation through photosynthesis. Each enclosure was filled with 91 cm of water (~ 360 liters), leaving a 26 cm headspace (~ 103 liters). Lids for each enclosure contained a septum port for headspace gas collection, a Tygon tube with a mesh filter on the end through which water was extracted for DIC, dissolved CO_2, CH_4 and O_2, and pH analyses, and a copper-constance thermocouple rod attached to a Campbell Scientific datalogger for monitoring water temperature inside the enclosures. Lids were sealed to enclosures using surgical tubing and/or silicone caulking. To prevent chemical gradients from establishing within enclosures, headspace gas was circulated into the water column for ~ 10 minutes every eight hours using high volume air pumps attached to the underside of the lids (for runs 1 and 2), and a peristaltic pump connected by hoses on the exterior of the enclosures (run 3).

Sampling protocol and gas measurements

Enclosure headspace samples were collected into evacuated 60 ml Wheaton bottles through a septum in the lid using double-ended needles. Dissolved gas samples were collected into evacuated 160 ml Wheaton bottles containing 8.9 g of KCl and backfilled with 10 ml of UHP nitrogen to maintain a headspace.

Headspace and dissolved concentrations of CO_2 and CH_4 were measured on a Shimadzu Mini 2 gas chromatograph (GC) equipped with a Heyes Sep D column for gas separation and a ruthenium catalyst methanizer (Colket et al. 1974). Samples containing dissolved gases were shaken on a wrist-action shaker for 15 minutes prior to analysis. A 0.2 ml headspace sample was injected into the GC using a pressure-lok syringe. Peak areas were quantified using a SP4270 Spectra-Physics integrator.

To obtain DIC concentrations, two mls of 10% HCl were added to the ~ 145 mls dissolved gas samples to convert dissolved HCO_3 (bicarbonate) to gaseous CO_2. The bottles were re-shaken prior to injecting gas samples into the GC.

Sample peak areas were converted to concentrations using a regression equation derived from peak areas corresponding with known CO_2 and CH_4 standard mixtures (22–19,000 ppm in air). Headspace gas concentrations were then corrected to standard temperature and pressure and expressed in (μmol of CO_2 and CH_4 per liter of headspace. Dissolved gas and DIC concentrations were converted to μmol of CO_2 and CH_4 per liter of water incorporating Henry's Law (Hamilton et al. 1994). Barometric pressure and water temperatures were recorded at the time of sampling and analysis.

Dissolved O_2 samples were obtained on the final day of each run to ensure aerobicity was maintained within the sealed enclosures. Water samples were flushed into BOD bottles for approximately two minutes using a peristaltic sampling pump. Oxygen samples were analyzed using the Winkler titration method (Stainton et al. 1977).

Close-vessel pH samples were collected before peat addition, and on the third and final days of the runs. pH was measured with an Orion model 800 pH meter calibrated with pH 8.0, 6.0 and 4.0 buffers.

Whenever gas and water samples were removed, air and Lake 239 water were injected back into enclosures to replace the volume removed for headspace and water samples to maintain neutral pressure within enclosures.

At the end of each run, peat block porewater was again sampled for DIC content. The peat block was then weighed while it was saturated with water and again after the porewater had been squeezed out of it. The difference between the two weights was assumed equal to the volume of the internal porewater. This volume was used to calculate the change in total DIC and dissolved CH_4 within the porewater of the peat blocks. Peat blocks were then oven-dried at 60 °C to obtain dry weights.

Similarly, volumes of water and headspace within each enclosure were used to calculate the total carbon content within each enclosure. Water volume was kept constant amongst enclosures at approximately 360 liters. Headspace volume varied between enclosures depending on the volume of peat added; headspace volume in each enclosure was adjusted by the volume of water displaced by the peat.

Rates of peat mineralization

For each sampling day, the inorganic carbon pool in each enclosure was calculated by summing: (1) headspace gases with dissolved CO_2 and CH_4, and (2) headspace gases with DIC and dissolved CH_4. Total inorganic carbon in enclosures containing peat blocks was corrected for volume changes in DIC or CH_4 content in peat porewater estimated using a linear regression between initial and final measured porewater content. Linear regressions were then used to calculate best-fit slopes of inorganic carbon accumulation in enclosures over the duration of the runs. Rates of inorganic carbon production (g C hr^{-1}) calculated for enclosures containing only water were subtracted from rates calculated for enclosures containing peat. These inorganic carbon production rates were divided by the dry weights of peat in each enclosure to determine inorganic carbon production per unit mass of dry peat per unit time.

Results

Total inorganic carbon buildup in enclosures (DIC + dissolved CH_4 + headspace CO_2 + headspace CH_4) generally increased linearly during incubation (Figure 3). Linear regressions used to determine the rate of inorganic carbon production per hour gave R^2 values between 0.63 and 0.98 with an average value of 0.94. The R^2 values for enclosures with only water ranged between 0.05 and 0.94 with an average value of 0.63. The control enclosures consistently had the lowest total inorganic carbon pools. These control enclosures quantified respiration rates in the water that were on average 36 ± 14%, 26 ± 3% and −12 ± 6% of those measured in enclosures with peat for runs 1 through 3, respectively. When DIC (i.e., dissolved CO_2 and HCO_3) and dissolved CH_4 were used to calculate total inorganic carbon in the enclosures instead of just dissolved CO_2 and CH_4, concentrations of total inorganic carbon were somewhat variable between enclosures containing pieces and blocks of peat (Figure 3). Inorganic carbon stored within peat block porewater over the duration of each run accounted for on average only 0.8 to 2.2 % of the total inorganic carbon pool in the enclosures (Table 2).

For any given run, mean mineralization rates of peat pieces were on average similar to mineralization rates of peat blocks (Table 3). However, measured rates of mineralization were quite variable between duplicates, and these rates were more variable for peat blocks than for peat pieces (Table 3). The overall pattern of mineralization, ranging from on average 67 ug C g^{-1} d^{-1} in mid July, to 141 ug C g^{-1} d^{-1} in late July/early August, and 123 ug C g^{-1} d^{-1} in mid-late August (Table 3) could not be explained by the overall pattern in water temperatures because mean temperatures differed only slightly from one run to another (Table 1). There was little variation in temperatures amongst enclosures throughout each run, except for a few dates during Run 2 (Figure 4).

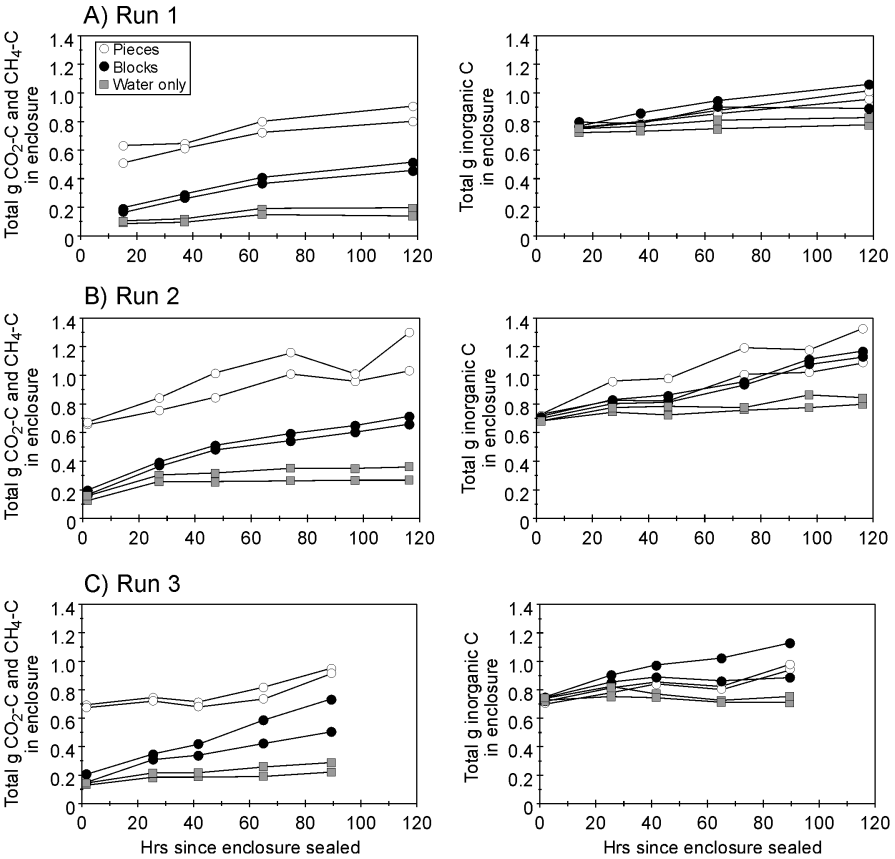

Figure 3. Gas carbon (CO_2-C and CH_4-C) buildup in enclosures with time (left column of graphs) and total inorganic carbon buildup in enclosures with time (right column of graphs).

Enclosures did not become anoxic during the experiments, but there was net O_2 consumption in enclosures containing both peat pieces and peat blocks. Dissolved O_2 concentrations in enclosures containing no peat averaged (± SE) 7.9 ± 0.8 mg l^{-1} at the end of runs versus 6.4 ± 0.1 mg l^{-1} in enclosures containing peat blocks and 5.1 ± 0.2 mg l^{-1} in enclosures with peat pieces (Table 1). We did not measure O_2 concentrations in enclosure headspaces, but extrapolating from dissolved O_2 measurements, it appeared that much more O_2 was consumed than CO_2 produced, probably due to chemical O_2 demand on the surface of small peat pieces.

The pH of enclosure water was measured for Runs 2 and 3 only (Table 1). Prior to addition of any peat, the mean pH (± SD) of water in all enclosures was 7.2 ± 0.1 for run 2 and 7.3 ± 0.1 for run 3 versus 6.7 ± 0.1 and 6.8 ± 0.1 at the end of runs in controls. The enclosures with peat blocks had slightly lower pHs, averaging 6.4 ± 0.1 for Run 2 and 6.3 ± 0.1 for Run 3. Peat pieces, however, acidified the water column to on average 5.1 ± 0.1 and 5.4 ± 0.2 for Runs 1 and 2 (Table 1). Because of the low pHs in the enclosures containing peat pieces, the majority of

Table 2. Mean net change in inorganic carbon content of peat block porewater over duration of run in each replicate enclosure (net change in carbon content in individual replicate chunks are in parentheses). A positive change indicates a gain of carbon within the porewater of the peat chunk, while a negative change indicates loss the to surrounding water. Please see Figure 3 for mean net change in total inorganic carbon in enclosures containing peat blocks.

	Run 1 (14–19 July) *net* Δ porewater C (mg)	Run 2 (31 July-5 August) *net* Δ porewater C (mg)	Run 3 (17–21 August) *net* Δ porewater C (mg)
CO_2-C	7 (35, –21)	36 (–45, 120)	40 (55, 25)
CH_4-C	10 (10, 10)	12 (1, 22)	16 (13, 20)
Total-C	*17 (45, −11)*	*48 (−44, 140)*	*56 (68, 45)*
Mean % total-C in porewater as CH_4-C	41%	25%	29%
Mean % total-C in enclosures in porewater	0.8%	1.3%	2.2%

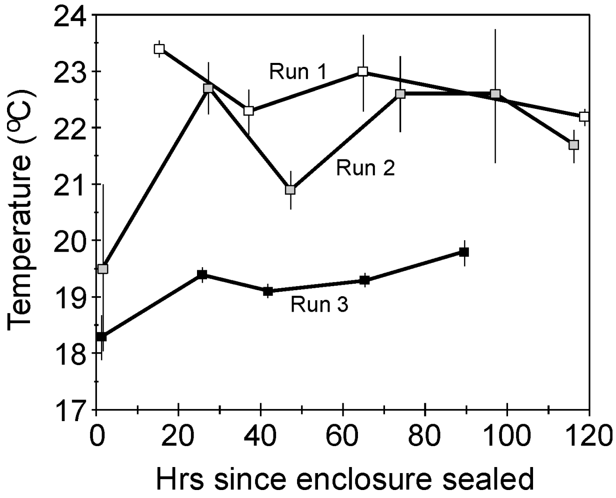

Figure 4. Mean (± SD bars) temperatures amongst enclosures for each run.

the inorganic carbon in these enclosures appeared as CO_2-C instead of bicarbonate ions (Figure 3). This was not found to be the case for enclosures containing peat blocks or water only.

CH_4-C generally constituted less than 0.2 % of the *total* inorganic carbon pool in enclosures. Only in enclosures containing larger peat blocks was there an increase in CH_4 during each 5 to 6-day run period, but this usually remained below 2.5% of the total inorganic carbon pool. In porewater within peat blocks, however, the percentage of total inorganic carbon that was CH_4-C ranged between 25–41% (Table 2). The production of CH_4 within peat blocks indicated that viable methanogenic bacterial populations were maintained within blocks and that some portions of peat blocks were decomposing anaerobically.

Table 3. Mean mineralization rates of pieces and blocks of peat (decomposition rates in individual replicate enclosures are in parentheses).

Treatment	Run 1 (14–19 July) μg C g^{-1} d^{-1}	Run 2 (31 July-5 August) μg C g^{-1} d^{-1}	Run 3 (17–21 August) μg C g^{-1} d^{-1}
Peat pieces	75 (58, 93)	140 (160, 130)	120 (100, 150)
Peat blocks	59 (95, 24)	140 (110, 170)	130 (180, 75)
Average	*67*	*140*	*123*

Discussion

Our experiments confirmed the hypothesis that a portion of the high fluxes of CO_2 and CH_4 to the atmosphere following the experimental flooding of the ELARP wetland could have been derived from peat mineralization in addition to the decomposition of flooded labile vegetation. All enclosures containing peat showed increases in CO_2-C and CH_4-C over time. However, we found no difference in total inorganic carbon produced by mineralization of peat pieces and blocks, and so estimating loss of peat carbon over time is simplified. This is helpful because initial mass of carbon in flooded peat is not too difficult to quantify, but it is very hard to predict physical breakup of peat after flooding. However, physical configuration of the decomposing peat could be important for predicting the ratio of CO_2 to CH_4 production. The ratio of CO_2 to CH_4 emitted to the atmosphere is important because CH_4 is approximately 21 times more effective a GHG over a 100 year period (Houghton et al. 1995).

Anoxia is required for CH_4 production. McKenzie et al. (1998) measured anaerobic mineralization rates for peat collected from floating islands on the ELARP reservoir in 1995. They incubated small 10-g (wet weight) saturated chunks of peat in Wheaton bottles evacuated and backfilled with UHP nitrogen, and so their conditions were more anoxic than ours. At the in-situ postflood temperature of 20–25 °C within floating peat islands, peat mineralization rates ranged between 142–497 μg C g^{-1} d^{-1}, with a mean of 286 μg C g^{-1} d^{-1}. Approximately 40% of the inorganic carbon produced in the McKenzie et al. (1998) incubations came from CH_4 production. This percentage of total inorganic carbon as CH_4-C was comparable to the 25–41% we observed within the porewater of our peat blocks (Table 2), but much higher than the maximum of 2.5% CH_4-C we measure in the enclosures overall. Because a portion of the peat in our blocks decomposed anaerobically, the small quantity of CH_4 measured within enclosures as a whole was likely due to the large volume of O_2 dissolved in water at the outset of our incubations and CH_4 oxidation in the aerobic water column in the enclosures. In fact, the δ ^{13}C of CH_4 in the water of the ELARP reservoir in the two years after flooding was approximately −38‰ compared to −53 to −70‰ at sites of production in the flooded peat, showing that some of the CH_4 had been oxidized in the water column of the experimental reservoir (Kelly et al. 1997). Fluxes of CH_4-C from the ELARP experimental reservoir pond surface averaged 7% of the total inorganic carbon flux in the first two years of flooding (Kelly et al. 1997). This percentage of total inorganic carbon as CH_4-C was higher than the maximum 2.5% CH_4-C in our study, but dissolved O_2 concentrations were often lower in the ELARP reservoir than in our enclosures (C.A. Kelly, unpublished data). Therefore, CH_4 diffusing into the water column from submerged portions of peat islands may be oxidized to CO_2 prior to diffusion to the atmosphere (Kelly et al. 1997), whereas the peat island surface may emit high proportions of CH_4 because the zone of production is in direct contact with the atmosphere (Scott et al. 1999).

Anaerobic mineralization rates measured by McKenzie et al. (1998) were over 2.5 times higher than rates we measured using peat blocks at similar temperatures (mean of 109 μg C g^{-1} d^{-1}, ranging between 24–180 μg C g^{-1} d^{-1}). In fact, our highest decomposition rates were similar only to the lowest values measured by McKenzie et al. (1998). Because our study took place in 1998, whereas the McKenzie et al. (1998) study was done in 1995, it is possible that our slower mineralization rates were related to the peat being more aged. Spatial differences between collection sites may have also influenced the peat decomposition rates.

We also observed a seasonal pattern to the rates of mineralization, with greater rates in late July/early August than in mid August and mid July, suggesting that microbial communities in the peat when it was sampled were more abundant and active during these times even though mean incubation temperatures declined from approximately 23 °C to 19 °C throughout the experimental runs. Because temperature is an important driver of peat mineralization over larger temperature ranges than those measured in our enclosures (McKenzie et al. 1998), the floating of peat islands is expected to enhance rates of peat mineralization in reservoirs. In the first two summers following creation of the ELARP reservoir, the inundated peat was on average 1–4 °C warmer than the peat in its natural state (Kelly et al. 1997) because water was a better conductor of heat than air. More importantly, though, the surface of the inundated peat was on average 19 °C, whereas two meters down in the peat profile the average temperature was only 6 °C. The floating islands allow for warm surface waters to circulate beneath the islands, regulating temperatures in the peat islands up to 70 cm from their bases (McKenzie et al. 1998). The remainder of the inundated peat is often exposed to much lower temperatures at the bottom of the reservoir, especially if the reservoir is deep and thermally stratified as is often the case (Rudd et al. 1993).

We found large differences between treatments in the proportion of the DIC pool that was dissolved CO_2. Enclosures containing peat pieces consistently had higher absolute values of CO_2, followed by enclosures with peat blocks and just water. Naturally occurring organic acids associated with the peat pieces acidified the water in the enclosures, shifting the bicarbonate equilibrium in favour of dissolved CO_2. The implications of this bicarbonate shift may be large in real reservoirs, depending on the ratio of reservoir volume to volume of peat flooded. For example, the pH in the pond of the experimental ELARP reservoir dropped 0.5 units from 6.7 in the two years prior to flooding to 6.2 in the three years after (unpublished data), a drop similar to that seen in our enclosure with peat blocks. pH induced shifts of bicarbonate to dissolved CO_2 may result in higher fluxes of CO_2 to the atmosphere than if the pH remained unchanged (Table 1).

How do decomposition rates of flooded peat compare with decomposition rates of peat found in its natural state? To make this comparison, we compiled from the literature a list of mineralization rates of peat, quantified using aerobic and anaerobic laboratory incubations, from northern bogs and fens (Table 4). There were two general differences in mineralization rates between the flooded peat in our study and unflooded peat studied in other locations. The first was that decomposition rates of flooded peat tended to be lower than aerobic rates measured in undisturbed bogs and fens, possibly because five-yr-old flooded peat was more recalcitrant than undisturbed peat. Second, mineralization rates of flooded peat were higher than some, but not all, anaerobic rates quantified in many of the undisturbed peat studies. Generally, anaerobic decomposition is slower than aerobic decomposition in these studies (Table 4). In the end, it is important to understand that peat does decompose under water, outside the natural habitat that it is normally found. It is also important to note that although peat does decompose in its natural state, peatlands generally tend to be carbon sinks because photosynthetic CO_2 uptake exceeds total carbon mineralization (Gorham 1991). Following inundation of peatlands, this carbon sink is lost (e.g., Kelly et al. (1997)).

Assuming that peat decomposes only during the ice-free season (200 days), a % carbon content in peat of $\sim 50\%$ (Turetsky et al. 2000), and a range of mineralization rates of 60–140 $\mu g\ C\ g^{-1}\ d^{-1}$, we estimate that only 2.4–5.6% of peat carbon is mineralized annually. This estimate suggests that fluxes of CO_2 and CH_4 from reservoirs that flood peatlands are not likely to diminish in the short term, potentially lasting 18–42 years. This duration of mineralization is probably a conservative estimate, because our rates were measured at the warmest time of the year using peat from warm floating peat islands. Furthermore, we expect that rates of mineralization will slow with time as the more labile carbon stores in peat are first depleted, leaving the recalcitrant carbon to decompose more slowly. Certainly as time proceeds following the onset of inundation of peatlands, the mineralization of peat will be relatively more important to the flux of CO_2 and CH_4 to the atmosphere than the decomposition of labile plant tissues in reservoirs.

Table 4. Mean mineralization rates (ranges in parentheses when available) of subsurface peat (−5 to −15 cm) from northern bogs and fens. Mineralization rates were quantified using aerobic and anaerobic laboratory incubations of intact peat.

Location	Incubation Temperature	Aerobic	Anaerobic		Reference
		CO_2-C (μg C g^{-1} d^{-1})	CO_2-C (μg C g^{-1} d^{-1})	CH_4-C (μg C g^{-1} d^{-1})	
Experimental reservoir (floating peat)					
ELA	20–25 °C	–	166 (80–280)	120 (61–220)	McKenzie et al. (1998)
Bogs and fens					
ELA	20–25 °C	–	142 (56–250)	6.4 (0.5–10)	McKenzie et al. (1998)
ELA	22 °C	176	145	0.4	Yavitt et al. (2000)
Northern Canada	15–20 °C	278	141	2.3	Moore and Dalva (1997)
Minnesota, USA	22 °C	166	238	0.1	Yavitt et al. (2000)
Minnesota, USA	30 °C	182	26[1]		Bridgham et al. (1998)
Minnesota, USA	30 °C	137	24[1]		Bridgham et al. (1998)
Sweden	15 °C	172	23	1.0	Bergman et al. (1999)
Sweden	16 °C	137 (103–170)	13 (13–14)	0.2 (0–0.6)	Magnusson (1993)
Sweden	18 °C	180	–	–	Hogg (1993)
Sweden	25 °C	172	66	10	Bergman et al. (1999)

[1] Total CO_2-C plus CH_4-C production quantified only

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