A comparison of methane flux in a boreal landscape between a dry and a wet year

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[1] We used field measurements of methane (CH_4) flux from upland and wetland soils in the Northern Study Area (NSA) of BOREAS (BOReal Ecosystem-Atmosphere Study), near Thompson, Manitoba, during the summers of 1994 and 1996 to estimate the overall CH_4 emission from a 1350 km^2 landscape. June–September 1994 and 1996 were both drier and warmer than normal, but summer 1996 received 68 mm more precipitation than 1994, a 40% increase, and had a mean daily air temperature 0.6°C warmer than 1994. Upland soils consumed CH_4 at rates from 0 to 1.0 mg m^{-2} d^{-1}, with small spatial and temporal variations between years, and a weak dependence on soil temperature. In contrast, wetlands emitted CH_4 at seasonal average rates ranging from 10 to 350 mg CH_4 m^{-2} d^{-1}, with high spatial and temporal variability, and increased an average of 60% during the wetter and warmer 1996. We used Landsat imagery, supervised classification, and ground truthing to scale point CH_4 fluxes (<1 m^2) to the landscape (>1000 km^2). We performed a sensitivity analysis for error terms in both areal coverage and CH_4 flux, showing that the small areas of high CH_4 emission (e.g., small ponds, graminoid fens, and permafrost collapse margins) contribute the largest uncertainty in both flux measurements and mapping. Although wetlands cover less than 30% of the landscape, areally extrapolated CH_4 flux for the NSA increased by 61% from 10 to16 mg CH_4 m^{-2} d^{-1} between years, entirely attributed to the increase in wetland CH_4 emission. We conclude that CH_4 fluxes will tend to be underestimated in areas where much of the landscape is covered by wetlands. This is due to the large spatial and temporal variability encountered in chamber-based measurements of wetland CH_4 fluxes, strong sensitivity of wetland CH_4 emission to small changes in climate, and because most remote sensing images do not adequately identify small areas of high CH_4 flux.

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1. Introduction

[2] The atmospheric concentration of methane (CH_4), although small (~1.7 ppmv), has been increasing at a rate of about 1% per year until the 1990s when the rate of change in the atmosphere became highly variable. The causes of this variability are uncertain but changes in the source-sink dynamics due to volcanic eruptions, cooling wetlands, and the collapse of a major economy have been included in the speculation [cf. *Dlugokencky et al.*, 2003]. An estimated 15% of the global warming potential increase during the 1980s can be attributed to CH_4 [*Intergovernmental Panel on Climate Change*, 1994]. Given this high degree of variability and the important potential climate impacts and feedbacks, there is a need to

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identify, characterize, and develop methodologies to quantify the major global sources and sinks of CH_4. CH_4 is produced under anaerobic conditions in waterlogged soils, and consumed by well-drained, upland soils, so that many landscapes will contain soils as both sources and sinks of CH_4. Ecosystems in the zone of discontinuous permafrost may be particularly prone to increasing methane emission as permafrost degrades [*Turetsky et al.* 2000; *Christensen et al.*, 2004; *Camill*, 2005].

[3] *Bouwman et al.* [1999] examined the problems associated with the scaling of trace gas fluxes from point measurements to the landscape and global scales. They concluded that the largest errors are likely to occur at the local and regional scales, arising from spatial variability of fluxes and inaccuracies in scaling up. Few regional CH_4 flux estimates have been made for complex landscapes. In the BOREAS (BOReal Ecosystem-Atmosphere Study) Northern Study Area (NSA), *Savage et al.* [1997] estimated an areally-weighted average summer consumption of 0.4 mg CH_4 m^{-2} d^{-1} from upland soils. In the Fort Simpson area of the Northwest Territories, Canada, *Liblik et al.* [1997] estimated the overall landscape flux of CH_4 to be $\sim$18 mg m^{-2} d^{-1} during the summer, based on chamber measurements and assignment of the landscape into different upland and wetland units. In the Kuparuk River basin in Alaska, *Reeburgh et al.* [1998] estimated an overall annual CH_4 flux of 0.8 g m^{-2} yr^{-1}, based on measurements during the frost-free season and spatial extrapolations of tundra, wetland, and open water systems. Few studies have estimated landscape fluxes for more than 1 year, but *Heikkinen et al.* [2004] reported that methane and carbon dioxide fluxes in an East European tundra catchment with discontinuous permafrost were significantly different between two summers with different temperature and precipitation patterns.

[4] At a larger regional scale, the Northern Wetlands Study of the Hudson Bay lowland used chamber-based CH_4 flux measurements combined with airborne sensors and remote sensing information (Landsat TM) to estimate an average summer CH_4 flux from this dominantly peat-covered landscape. The chamber-based measurements produced an estimated emission of 20 mg CH_4 m^{-2} d^{-1}, for an estimated overall contribution of 0.5 Tg CH_4 yr^{-1} from the lowland [*Roulet et al.*, 1994]. *Bachand et al.* [1996] produced a map of CH_4 emissions from Canadian wetlands, based on point measurements and extrapolated from maps of peatland type and areal coverage. Annual emissions were estimated to be 3.5 Tg CH_4, or 0.7% of the estimated global emissions of CH_4. At the global scale, both *Matthews and Fung* [1987] and *Aselmann and Crutzen* [1989] made estimates of CH_4 emissions from wetlands, based on flux measurements and estimated spatial coverage of land types. *Bartlett et al.* [1992] estimated global emissions of 11 $\pm$ 3 Tg CH_4 from tundra environments based on wetland and upland ecosystems in the Yukon-Kuskokwim Delta, Alaska. Most of these studies conclude that small wetlands, ponds, and vegetated edges of lakes contribute a substantial portion of the landscape CH_4 flux [cf. *Juutinen et al.*, 2003].

[5] As part of BOREAS [*Sellers et al.*, 1997; *Hall*, 1999, 2001], exchange of CH_4 between the atmosphere and the soil was measured at a series of wetland and upland sites in the Northern Study Area (NSA), near Thompson, Manitoba. BOREAS chamber studies involved measurements in both upland forests [e.g., *Burke et al.*, 1997; *Savage et al.*, 1997] and wetlands [*Bubier et al.*, 1995; *Moosavi and Crill*, 1997; *Bellisario et al.*, 1999]. At a beaver pond, CH_4 flux was measured both by chambers [*Dove et al.*, 1999] and by a micro-meteorological, flux-gradient approach [*Roulet et al.*, 1997].

[6] Remote sensing has been used to map and estimate coverage of different boreal ecosystems, mainly forests, in this area using AVHRR, AVIRIS, and Landsat TM (Thematic Mapper) images [e.g., *Cihlar et al.*, 1997; *Peddle et al.*, 1997; *Steyaert et al.*, 1997; *Fuentes et al.*, 2001]. Small-scale variations in plant cover may be distinguished by differences in spectral reflectance measurements of vegetation, such as mosses [e.g., *Bubier et al.*, 1997; *Rapalee et al.*, 2001]. CASI [*Gray et al.*, 1997; *Zarco-Tejada and Miller*, 1999] has been used to capture ecosystem variations at finer spectral and spatial scales than other remote sensing instruments. One of the unique aspects of the BOREAS project was the combination of remote sensing and ground-based measurements to understand atmosphere-biosphere interactions in the boreal region in response to climate change [*Gamon et al.*, 2004].

[7] In this paper, we compare chamber CH_4 fluxes across a range of wetland and upland boreal ecosystems in the BOREAS Northern Study Area between 2 years (1994 and 1996) with different precipitation and temperature patterns. We then scale CH_4 fluxes for both growing seasons from point measurements in the field ($<$1 m^2) to the landscape ($\sim$1000 km^2), using CASI and Landsat TM remote sensing data. We focus on the large range in interannual variability depending on moisture status and plant cover type, the difficulty in determining the spatial and temporal variations in CH_4 flux from wetland sites, and the ability of remote sensing images to adequately represent the points of major methane emission in the landscape. We assess the reliability of the landscape CH_4 flux estimate based upon the errors inherent in both CH_4 flux measurements and the uncertainty in remote sensing classifications.

2. Methods

2.1. Study Area

[8] The Northern Study Area (NSA) of BOREAS, near Thompson, Manitoba (55.91°N, 98.42°W), encompasses a wide range of wetland and upland ecosystems, and discontinuous permafrost. The main peatland study area at the fen tower site [*Lafleur et al.*, 1997; *Joiner et al.*, 1999] was chosen for its diverse representation of plant communities, thermal and hydrologic gradients, and its inclusion of permafrost peat plateaus and collapse scars. The underlying substrates supporting the wetlands are Glacial Lake Agassiz sediments overlying regional bedrock of Canadian Shield Pre-Cambrian gneissic granite. Soils are derived primarily from glacial lake sediments and consist mostly of calcium-rich clays and organics. Wetlands are common in the region due to poor drainage across the flat terrain, and include a wide range of types found in northern latitudes from rich fen

Table 1. Climate Data for the BOREAS Northern Study Area in 1994 and 1996 Compared to Climate Normals (1967–1990) From the Atmospheric Environment Service Meteorological Station, Thompson, Manitoba[a]

	Precipitation, mm			Air Temperature, °C		
Period	Normal	1994	1996	Normal	1994	1996
June	71.5	60.5	76.6	12.3	13.7	14.2
July	84.3	68.2	33.4	15.7	15.5	16.8
August	77.7	22.1	59.9	13.8	12.8	15.0
Sept	63.4	19.1	67.8	7.2	12	10.2
Sum	296.9	169.9	237.7			
Mean	74.2	42.5	59.4	12.3	13.5	14.1
Standard deviation	8.9	25.5	18.6	3.6	1.5	2.8

[a]From *Joiner et al.* [1999].

to bog (pore water pH 7.0 to 3.9) [*Zoltai*, 1988], beaver ponds and marshes. Plant associations in fens are diverse, dominated by brown mosses of the Amblystegiaceae family (e.g., *Drepanocladus, Scorpidium* spp.) and sedges (particularly *Carex* spp.). Permafrost underlies many of the peatlands; frozen peat plateaus are dry and wooded with upland plant communities such as black spruce (*Picea mariana*), feathermosses (e.g., *Pleurozium schreberi, Hylocomium splendens*), and ericaceous shrubs (e.g., *Ledum groenlandicum*). Areas of permafrost degradation are found interspersed in the frozen features. These collapse scars become bogs (species poor, acidic, *Sphagnum*-dominated communities) if they collapse completely internal to a peat plateau, isolated from groundwater. They may develop into fens if they collapse on the edge of a peat plateau where groundwater intrudes [*Zoltai*, 1993; *Vitt et al.*, 1994; *Halsey et al.*, 1997]. See *Bubier et al.* [1995] for more complete site descriptions, water chemistry, and plant species composition of the wetlands.

[9] Wetland sites were classified using a modification of the Canadian peatland classification system that uses vegetation physiognomy (tree, shrub, graminoid) and water chemistry (pH, calcium, magnesium) as the primary variables [*Zoltai*, 1988]; for example, open sites had <10% tree cover, treed sites had 10–30% tree cover, and graminoid sites had >50% cover by sedges. The pore water pH of bogs was 3.8–4.7; poor fens 4.5–5.1; intermediate fens 5.1–6.2; rich fens 6.2–7.0. For the purposes of this classification, intermediate and rich fens were combined into one fen class. Open graminoid fens were sites dominated by large sedges (e.g., *Carex rostrata*) with water table close to the peat surface. Open graminoid poor fens were dominated by *Carex limosa* and other smaller sedge species. Open low shrub fens were slightly drier with dominance in the shrub layer of birch and willow (*Betula* and *Salix* spp.) and some sedges, such as *Scirpus hudsonianus*. Treed sites were dominated by larch (*Larix laricina*) and black spruce. Open low shrub bogs had <10% tree cover and were dominated by ericaceous shrubs and *Sphagnum* mosses. Small (<1 km^2) ponds included open water edges of collapse scars and beaver ponds, although a more detailed analysis of beaver ponds was conducted in a separate study (N. Roulet, personal communication, 2004).

[10] Uplands consisted of jack pine (*Pinus banksiana*) communities on well-drained sandy soils, black spruce on poorly-drained, organic-rich soils, and aspen (*Populus tremuloides)* on recently burned sites. Since fire is a major factor shaping the age and species composition of upland ecosystems, we chose sites that represented gradients of fire history as well as drainage. See *Savage et al.* [1997] for more complete site descriptions.

2.2. Climate

[11] The average annual temperature and precipitation for the region are −3.9°C and 585 mm (232 as snow; 353 as rain). June–September average air temperature is 12.3°C, with July as the warmest month on average (Table 1). Average total precipitation for the same period is 297 mm. June through September 1994 and 1996 were both warmer and drier than the 30-year average, but summer 1996 received 68 mm more precipitation than 1994, a 40% increase, and had a mean daily temperature of 0.6°C greater than in 1994. Summer 1994 was the third driest on record, with August and September receiving 99.9 mm less rain than normal. In 1996, only July precipitation was well below normal.

2.3. Environmental Data

[12] In the wetlands, we measured water table position relative to the peat surface, and peat temperature at 10-cm increments from 0–50 cm depth once a week at the same time as CH_4 flux sampling. In order to get a more detailed temporal record at selected sites, water level recorders and thermocouples were installed at the boardwalk sites, placed along moisture and chemical gradients. Data loggers recorded water table elevation and peat temperature at four depths (10, 20, 50, and 100 cm below the peat surface) every 2 hours from June through September. Air temperature was measured at all the upland sites at the same time as CH_4 flux sampling.

2.4. CH_4 Flux Measurements

[13] The majority of the CH_4 flux measurements in uplands and wetlands were made by a static chamber technique, which consisted of placing a chamber over the soil surface and measuring the change in concentration of CH_4 within the chamber headspace [*Bubier et al.*, 1995; *Moosavi and Crill*, 1997; *Savage et al.*, 1997]. The area covered by the chamber was small (0.05 to 0.3 m^2), exposure periods were short (<20 min), and the chamber was opaque to reduce heating and pressure changes within the chamber. Four to nine PVC or aluminum collars were placed in each major vegetation type along moisture, chemistry, and plant community gradients to minimize disturbance and to ensure that the same location was measured on each date. Collars were installed a minimum of 14 days before commencing measurements and remained in place throughout the measurement program. Water was added to the groove in each collar to ensure an air-tight seal with the chamber in place. At the wetland sites, boardwalks were installed to further minimize disturbance. CH_4 flux was measured at each collar location once a week from May through September 1994 and 1996 during the major field seasons of BOREAS.

Table 2. Mean and Standard Deviation of the Reflectance Values for the Training Areas Used to Classify the CASI Fen Tower Image[a]

Class Training Area	Band 8	Band 12
Road	13789 ± 1302	16926 ± 1121
Open water	397 ± 85	914 ± 407
Wet sedge fen (OgF)	5650 ± 513	10078 ± 621
Sedge fen (OgPF)	7478 ± 662	13348 ± 662
Treed fen (TtsF)	3606 ± 391	11674 ± 512
Shrub fen (OlsF)	3152 ± 233	9033 ± 454
Shrub bog (OlsB)	3588 ± 1043	16637 ± 710
Black spruce upland	1683 ± 419	8770 ± 1340

[a]Classifications compare with sites from *Bubier et al.* [1995]. O, open; T, treed; g, graminoid; ts, tall shrub; ls, low shrub; *F*, fen; B, bog; PF, poor fen.

[14] Five air samples were taken from the chamber headspace over 20 min, then returned to a laboratory in Thompson, Manitoba, and analyzed for CH_4 within 4–6 hours of collection. Analyses were conducted with a Shimadzu gas chromatograph with a flame ionization detector (FID-GC) using a Poropak Q column. Nitrogen was used as the carrier gas, and CH_4 standards of 2.35, 211, and 2037 parts per million by volume (ppmv) were used to calibrate. Precision of the analysis (standard deviation as percent of the mean of 10–15 daily repetitions of the standard) was 0.1 to 0.5% of the standards. The minimal detectable flux was 0.1 mg CH_4 m^{-2} d^{-1}. Values between 0.1 and −0.1 were considered to be zero and were included in the final seasonal integration. Fluxes were calculated by linear regression of the change in concentration in the five samples. The correlation coefficient of the regression had to be significant to the 95% confidence limit for $n = 4$ or 5 ($r^2 = 0.95$ or 0.87); otherwise the sample was rejected. Sites with ebullition were kept in the data set even if a large increase was observed between two of the samples as long as the correlation coefficient was still significant at $p < 0.05$.

2.5. Remote Sensing Measurements and Analyses

[15] Scaling CH_4 flux data from point measurements to the landscape was accomplished with a variety of remote sensing techniques. CASI (Compact Airborne Spectrographic Imager) [*Gray et al.*, 1997] data were used for the characterization of vegetation at a fine spatial resolution (3 m) and at narrow bandwidths. Landsat TM data (30 m spatial resolution) were used for a broader spatial and coarser spectral resolution, respectively, for extrapolation purposes. The errors associated with the remote sensing analyses were factored into the final landscape-level CH_4 flux.

2.5.1. Supervised Classification of the 1994 CASI Images

[16] A 2.5 km^2 CASI image of the Tower Fen site area was taken in June 1994 from an altitude of 1639 m, with 15 wavelength bands, spanning the visible and near-infrared portion of the spectrum, ranging in wavelength from 0.409 to 0.905 μm. Surface reflectances were derived from the image by accounting for the attenuation of radiation through the atmosphere, after *Tanré et al.* [1990]. Supervised classification is carried out by feeding the computer a sample of ground reflectance values from an image that is known to be of a particular vegetation type (a training area). The computer uses this as a basis for finding all pixels in the image that correspond to this vegetation type, and proceeds for each class. An initial supervised classification (using maximum-likelihood enhanced neighbor techniques with a class filter) was made of the ability of the CASI image to identify eight vegetation classes in training areas around the Tower Fen. We then verified these initial classes on the ground. Many of the wavelength bands in the CASI image were not effective in distinguishing vegetation types. However, the greatest differentiation among classes was from wavelength band 8 (mean wavelength 0.682 μm with 0.0033 width) and band 12 (mean wavelength 0.768 μm with 0.0051 width). Average percent reflectance values were developed from these channels (Table 2), which were used to define training areas and to conduct a second supervised classification. The same techniques were applied to a CASI image of another area, the Old Black Spruce (OBS) Tower site [*Goulden et al.*, 1998]. Each classification produced statistics on the probability that a particular pixel was classified correctly. This probability is based on how well the training areas were separated into distinct vegetation classes. The scattergrams for the CASI fen and OBS images showed that the training areas were well-separated with the exception of the black spruce upland and the treed fen. The training areas were then checked on the ground with user knowledge, and the classification errors modified. After ground verification, we determined that 84–86% of the CASI images were classified with greater than 90% probability of being classified correctly.

2.5.2. Supervised Classification of the 1994 Landsat TM Image

[17] Landscape-level methane emissions were estimated from a supervised classification of a Landsat TM image of the NSA by first determining training areas. Because the CASI and Landsat TM images have different band wavelengths, a direct transfer of the CASI classification could not

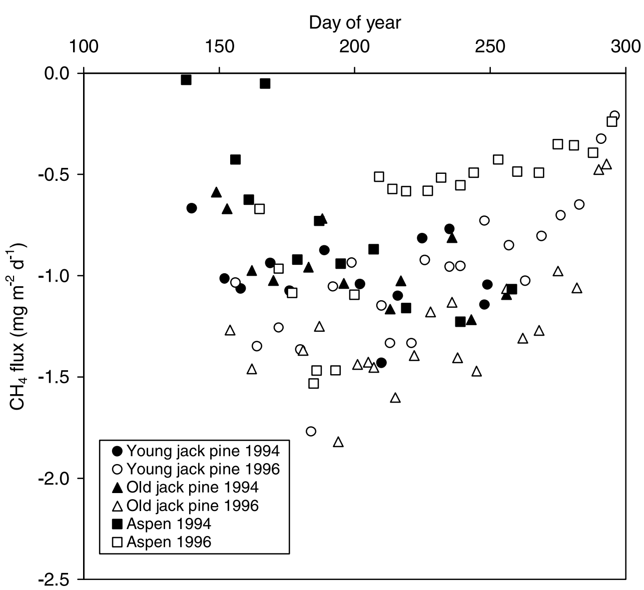

Figure 1. Seasonal variation in 1994 and 1996 CH_4 flux at three upland sites.

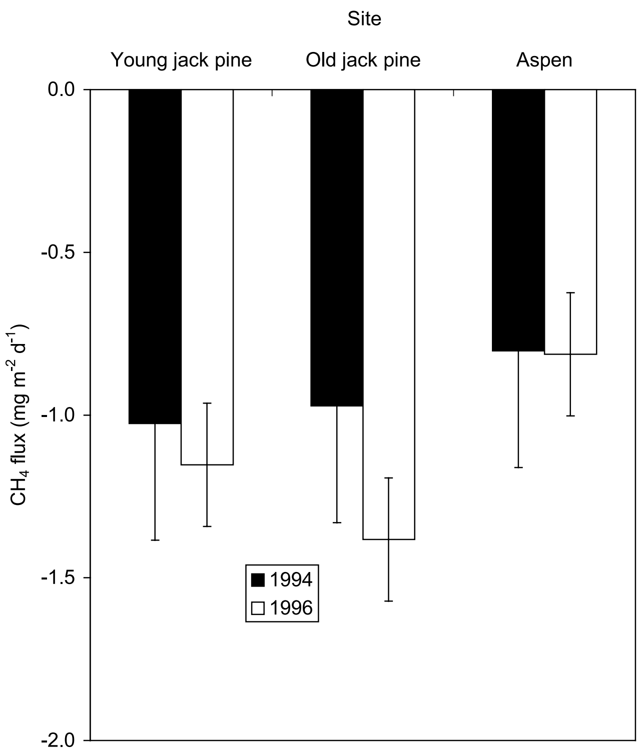

Figure 2. Mean (±standard deviation) CH_4 flux for the period day of year 150 to 250, 1994 and 1996, at the three upland sites. A t-Test revealed no significant difference between the means for the 2 years at the Young Jack Pine (p = 0.16) and Aspen (p = 0.94) sites, but there was a significant difference at the Old Jack Pine site ($p < 0.01$).

be applied to Landsat. Therefore the training areas were defined by selecting a 1349 km^2 area from the Landsat image, overlaying the CASI Fen image, and using the CASI vegetation classifications. We then determined the probability of a pixel being classified correctly (>80%) in the Landsat area, which showed that 70% of the total Landsat area was classified with >80% probability. Classification errors were then corrected with ground verification using the CASI images and user knowledge of the area as the basis for comparison. This resulted in a 10–12% change in the area assigned to each vegetation class.

3. Results and Discussion

3.1. CH_4 Flux Measurements and Variability

[18] In three upland soil sites used in 1994 and 1996, the average daily flux of CH_4 from June to September ranged from 0.0 to -1.8 mg CH_4 m^{-2} d^{-1} (Figure 1). Rates of CH_4 consumption generally increased from spring to summer and then decreased in the fall. The slopes of the regression between flux and air temperature for the combined years are similar for each site (-0.027 to -0.037 mg CH_4 m^{-2} d^{-1} per °C), showing a rather weak dependence of CH_4 uptake on temperature. When a comparison was made of CH_4 fluxes at the three sites during the period that overlapped in 1994 and 1996 (day of year 150 to 250), there was no significant difference at the Young Jack Pine and Aspen sites (t-Test p = 0.16 and 0.94, respectively; Figure 2). There was, however, a significantly greater rate of CH_4 consumption at the Old Jack Pine site in 1996 than in 1994 (t-Test $p < 0.01$), though the increase in absolute terms was modest (from -0.97 to -1.38 mg CH_4 m^{-2} d^{-1}). Two of these sites (Young and Old Jack Pine) have well-drained, sandy soils, and the Aspen is underlain by clay, though on a slope, and so is moderately well drained. In these soils, most of the CH_4 consumption activity was located at the base of the soil organic horizons and appears related to soil pH, nitrogen cycling, and the presence of soluble organics [*Amaral and Knowles*, 1997a, 1997b].

[19] The combination of CH_4 flux data from upland soils collected in 1994 [*Burke et al.*, 1997; *Savage et al.*, 1997] and in 1996 allows a compilation by upland site in a matrix based on moisture class and age since fire disturbance (Table 3). The consumption of CH_4 was largest in well-drained soils on both sand and clay substrates under jack pine and aspen, and generally decreased as the soils become wetter and overlain by a thick layer of organic matter and *Sphagnum* mosses. CH_4 flux appeared to be slightly increased by fire and vegetation regeneration [*Burke et al.*, 1997].

[20] In contrast to the conservative nature of CH_4 flux from upland soils, emissions from wetlands showed much greater variability, both seasonally and spatially. Average summer daily CH_4 flux ranged from 10 to 350 mg CH_4 m^{-2} d^{-1}, with the highest fluxes at graminoid fens [*Bubier et al.*, 1995]. Standard deviations of CH_4 flux within each category are absolutely large (5 to 167 mg CH_4 m^{-2} d^{-1}), but the coefficients of variation are similar to that observed in the upland soils (0.4 to 1.8, average 1.0). A strong seasonal pattern existed at most sites, related to temperature, water table position, vascular plant cover, and net primary productivity [*Bubier et al.*, 1995; *Moosavi and Crill*, 1997; *Bellisario et al.*, 1999]. Methane emissions were greatest from open graminoid sites because of the high water table and the presence of highly productive sedge species, particularly *Carex rostrata*, which has been shown to enhance CH_4 flux by transporting CH_4 to the atmosphere

Table 3. Compilation of CH_4 Flux Data (mg CH_4 m^{-2} d^{-1}) From Upland Forest Soils in the Northern Study Area[a]

Moisture Class	Mature Forest (>60 Years Since Burn)	Intermediate Forest (15–30 Years Since Burn)	Young (<5 Years Since Burn)
Dry	pine: −1.0, **−1.4**, *−1.2*	pine: −1.0, **−1.1**	pine: −1.8, *−1.9*
Mesic	aspen: −0.7, **−0.8**	aspen: −1.3	spruce: −0.4
Wet	spruce: 0.0, *−0.8*	spruce: 0.0, *−1.2*	spruce: −0.8, *−0.4*, *−0.9*

[a]On the basis of summer season averages of 1994 data from *Savage et al.* [1997] and *Burke et al.* [1997] (italic), and 1996 data from this study (boldface). The ecological matrix is divided by age since disturbance (fire) and moisture class.

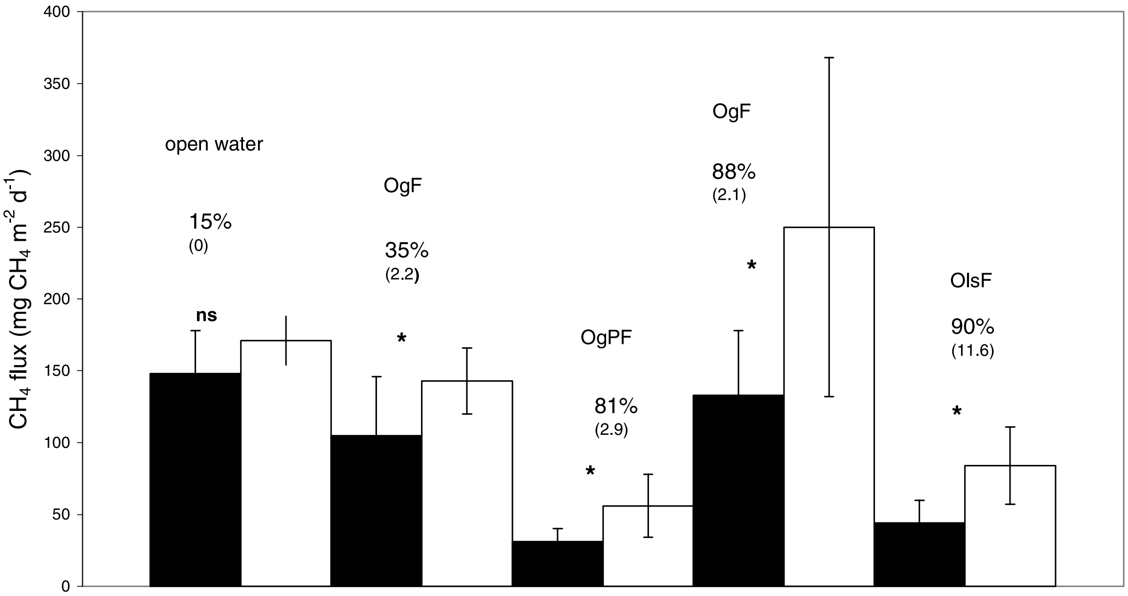

Figure 3. The 1994–1996 seasonal mean (±standard deviation) CH_4 flux for representative wetland types described in section 2 and Table 4. Open water sites were on the edges of permafrost collapse scars. Stars indicate significant differences between years at $p < 0.01$ or $p < 0.05$. Percent increase in CH_4 flux is shown between 1994 and 1996, and increase in mean water table position below the peat surface (cm) is shown in parentheses.

through air spaces in the stem (aerenchyma), bypassing the zone of CH_4 oxidation in the aerated part of the peat profile [*Whiting and Chanton*, 1992; *Waddington et al.*, 1996; *King et al.*, 1998; *Joabsson et al.*, 1999; *Christensen et al.*, 2003a, 2003b]. Isotopic analyses also showed that the CH_4 produced in *Carex*-dominated areas was derived from recent plant compounds rather than the breakdown of organic matter [*Chanton et al.*, 1992; *Bellisario et al.*, 1999; *Dove et al.*, 1999]. These sites of high methane emission tended to be located along the margins of peat plateaus and collapse scars and were small in their areal coverage. Other studies have also found high CH_4 emission from edges of ponds or wetlands, thermokarst features, and permafrost collapse scars [*Bartlett et al.*, 1992; *Reeburgh et al.*, 1998; *Juutinen et al.*, 2003; *Christensen et al.*, 2004; *Heikkinen et al.*, 2004]. Methane flux from the beaver pond by continuous measurements on a micro-meteorological tower was estimated to average 112 mg CH_4 m^{-2} d^{-1} over the summer [*Roulet et al.*, 1997]. Using an enclosure method on the same beaver pond, *Dove et al.* [1999] reported fluxes of 155 and 320 mg m^{-2} d^{-1} from vegetated and open portions of the pond. Treed and shrub-dominated sites were lower in methane emission because they indicated lower water tables, drier conditions, and lacked the type of vascular plants that enhance methane emission (e.g., sedges).

[21] CH_4 emission increased significantly from 1994 to 1996 at all the wetland sites (Figure 3). The increase ranged from 15% to 90% between years, depending on the change in water table position and the biomass of sedges. The seasonal pattern of CH_4 flux showed that emissions were generally higher throughout the 1996 summer than 1994, but with a sharp rise in CH_4 flux around day 190 (Figures 4a and 4b). This peak may be due to a combination of higher water table, peat temperature, and plant productivity during that time period in 1996. Individual points represent mean ±1 s.d. for four to nine collars on that date, and the large

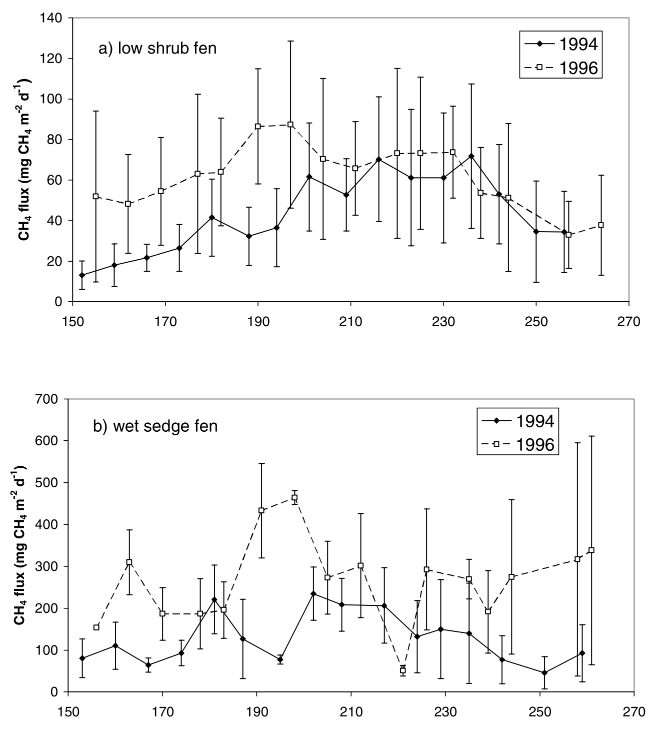

Figure 4. Seasonal pattern of CH_4 flux comparing 1994 and 1996 in (a) low shrub fen and (b) wet sedge fen.

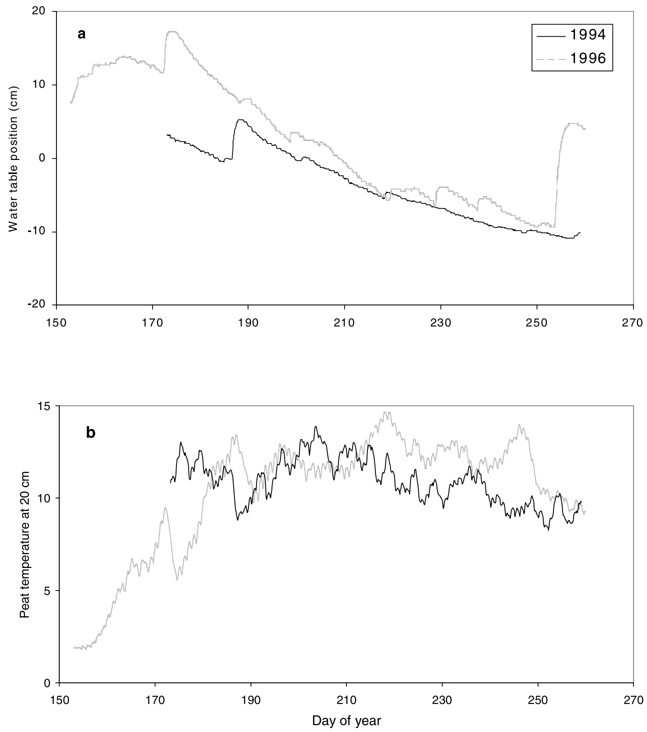

Figure 5. The 1994 and 1996 seasonal patterns of (a) water table position and (b) peat temperature at 20 cm at a sedge fen.

spatial variability within each vegetation class is evident. The seasonal pattern of the water table position and peat temperature showed that water table was higher in 1996, especially at the beginning and end of the season (Figure 5a). The peat temperatures were similar between years, except for a generally higher temperature toward the second half of the summer in 1996 (Figure 5b). This resulted in a seasonal average difference of approximately 0.5°C in peat temperature at 10–20 cm below the peat surface, similar to difference in summer average air temperature between years (Table 1). The net ecosystem CO_2 exchange (NEE) data for the same 1994–1996 period reported by *Joiner et al.* [1999] shows that in the drier 1994 season, the Tower Fen was a net source of CO_2 (30.8 g CO_2-C m^{-2}) to the atmosphere, while the wetter 1996 season was a net sink (−91.6 g CO_2-C m^{-2}).

[22] Wetland CH_4 flux was strongly correlated with water table position in both years (Figure 6), and the linear regressions had very similar slopes and intercepts. Because of the log linear relationship between mean seasonal CH_4 flux and water level, very small increases in water table position result in a much larger increase in CH_4 flux. This was evident in the 2 years of our study where the average water table change between years was 2–5 cm, yet CH_4 emission increased on average 60%. The slope of the linear regression was slightly higher in 1996, reflecting this sensitivity to warmer and wetter conditions. However, the slightly weaker correlation coefficient in 1996 may be due to the smaller range in water table position among sites compared with 1994. Temperature at the average position of the water table is an even stronger predictor of CH_4 flux [*Bubier et al.*, 1995] because it combines two of the most important environmental controls on CH_4 production and oxidation. The slightly warmer peat temperatures in 1996 also contributed to the substantial increase in CH_4 emission from the wetlands. Sedge-dominated wetlands had the highest increase in CH_4 between years (Table 4; Figure 3). In spite of the similar increase in water level among sites in 1996, the sedge sites probably had greater plant production enhancing CH_4 emission to the atmosphere. *Bubier et al.* [2003] reported that sedges produce more biomass in wetter years, which could contribute to higher CH_4 fluxes by enhancing root exudates [*Bellisario et al.*, 1999; *King et al.*, 2002; *Ström et al.*, 2003] and by increasing the coverage of aerenchyma in the stems. This could also contribute to the weaker water table: CH_4 relationship in 1996 in that shrub-dominated sites with similar water table position did not have the same increase in CH_4 flux as sedge sites.

3.2. Landscape Extrapolation of CH_4 Flux

[23] We used the supervised classification of the Landsat image to determine the areal coverage of the major upland and wetland vegetation classes (Table 4). Uplands cover most of the boreal landscape, with black spruce being the most common ecosystem. Different age classes of jack pine and aspen at regenerating burn sites occupy most of the other upland areas. Wetlands cover approximately 30% of the landscape, primarily treed, shrub, and sedge fens. Although uplands cover almost 70% of the landscape, they contribute a very small amount to the landscape CH_4 emission and interannual variability. The areally weighted flux for uplands was −0.31 Mg (−314 kg) and −0.35 Mg (−354 kg) CH_4 d^{-1} for 1994 and 1996, respectively, which represented a 13% increase in CH_4 consumption between years. However, when 95% confidence limits are factored into these estimates, the difference between years was not significant, except for mature jack pine sites (Figure 2). In

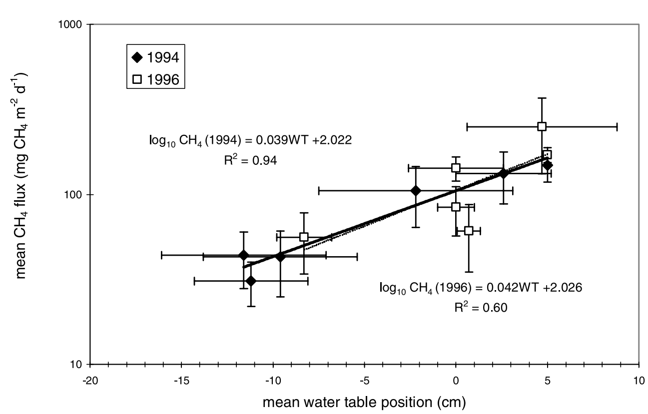

Figure 6. Linear regression between mean seasonal $\log_{10}$ CH_4 flux and mean water table (WT) position for wetlands (May–September, 1994 and 1996). Values for WT indicate centimeters below (negative) or above (positive) the peat surface. Error bars are standard deviations, which incorporate both seasonal and spatial variability in fluxes.

Table 4. Landscape Estimates of CH_4 Flux for 1994 and 1996 Using Landsat Supervised Classification of a Portion of the NSA[a]

Landscape Component	Area, km^2	CH_4 Flux ± Standard Deviation, mg m^{-2} d^{-1} 1994	1996	Change 1996–1994, %	Areal Flux, kg CH_4 $area^{-1}$ d^{-1} 1994	1996	Change 1996–1994, %
			Uplands				
Black spruce[b]	623.9	−0.1 ± 0.1	−0.1 ± 0.1	0	−62	−62	
Jack pine	135.7	−1.0 ± 0.2	−1.3 ± 0.3	30	−136	−176	
Regenerating burns (aspen)	145.0	−0.8 ± 0.4	−0.8 ± 0.4	0	−116	−116	
Disturbed (includes roads)	21.1	0 ± 0	0 ± 0	0	0	0	
Total	925.7				−314	−355	−13
			Wetlands				
Treed fen (TtsF)[b]	117.7	12 ± 10	12 ± 10	0	1412	1412	
Shrub fen (OlsF)	88.3	44 ± 18	70 ± 15	59	3885	6181	
Wet sedge fen (OgF)	6.9	133 ± 45	250 ± 118	88	918	1725	
Sedge fen (OgPF)	103.0	70 ± 25	120 ± 22	71	7210	12360	
Shrub bog (OlsB)[b]	2.0	45 ± 28	45 ± 28	0	90	90	
Small ponds[b]	1.8	148 ± 30	171 ± 17	16	266	308	
Rivers and lakes	90.2	0	0	0	0	0	
Total	409.9				13782	22076	60
			Combination				
Areal extrapolation	1335.6	10.1 ± 4.5	16.3 ± 4.3		13468	21721	61

[a]Flux estimates are seasonal means (±1 standard deviation).
[b]CH_4 fluxes at Treed fen, Shrub bog, and Black spruce sites were not measured in 1996; therefore the 1996 CH_4 flux estimates are probably an underestimate because 1994 data were used for both years for these classes. The small pond estimate is also very conservative since beaver ponds cover a larger percent of the landscape than estimated here (N. Roulet, personal communication, 2004). On the basis of the ecological matrix in Table 3, upland fluxes were weighted by their areal coverage according to *Savage et al.* [1997] and *Rapalee et al.* [2001].

contrast, the 2-year wetland CH_4 emissions were 13.8 and 22.1 Mg d^{-1}, which resulted in a 60% increase between 1994 and the wetter and warmer 1996 (Table 4). The total landscape flux estimate, which accounts for the relative contribution of different wetland and upland ecosystems, was 13.5 and 21.7 Mg CH_4 d^{-1} for the 1350 km^2 area in the 2 years, respectively, which was a 61% increase in CH_4 emission. This is equivalent to an average daily flux of 10.1 (±4.5) and 16.3 (±4.3) mg CH_4 m^{-2} d^{-1} for the two summers. The large increase was entirely attributable to wetland CH_4 emission because the uplands had insignificant change between the 2 years. Although wetlands are small in spatial coverage, they control the landscape-level fluxes and changes between years. It is also noteworthy that the increase in wetland CH_4 emission corresponded to only a small change in water level (2–5 cm on average) and peat temperature (0.5°C) between years.

3.2.1. CASI Versus Landsat

[24] The areal count for classes showed that the Landsat image underestimated the total wetland coverage and CH_4 flux compared to the CASI images. For example, total areas classified as wetland from the Fen and OBS areas using Landsat were 39.9% and 25.4%, respectively, compared with 46.9% and 37.6% using the CASI images (Table 5). A similar comparison of CASI and Landsat data was used in mapping mountain heath vegetation in the sub-arctic region of northeastern Norway [*Tommervik et al.*, 1997]. That study found that CASI data separated the area into more vegetation classes and that sparsely vegetated areas were mapped in more detail with the CASI sensor than with the Landsat data. In a study of the Southern Study Area (SSA) of BOREAS, *Zarco-Tejada and Miller* [1999] showed that the "red edge" portion of the reflectance spectrum (near 700 nm) in CASI can lead to improved vegetation mapping. In that study, the fen in particular, which was not easily resolved by other sensors [*Gamon et al.*, 2004], was easily distinguished with these red-edge parameters, which are responsive to variations in leaf pigment content, leaf area (LAI), and understory vegetation. *Bubier et al.* [1997] found that mosses exhibit distinctly different spectral characteristics than vascular plants, which may in part account for these red-edge distinctions in the fen.

[25] In our study of the NSA, smaller wetlands (<30 × 30 m^2) were detectable with CASI, but not with Landsat. Often these smaller areas were beaver ponds or permafrost collapse features located in the margins of peat plateaus. These areas were usually the sites of highest CH_4 emission or "hot spots" because they had either shallow open water

Table 5. Comparison of the Net CH_4 Flux in 1994 From the CASI and Landsat TM Classifications With Ground Truthing

Location	Classification	Wetland, %	Upland, %	Flux (Range), mg CH_4 m^{-2} d^{-1}
Fen	CASI	46.9	50.6	17.3 (11.7–31.8)
	Landsat	39.9	51.1	13.6 (9.2–24.2)
OBS	CASI	37.6	57.8	9.3 (5.0–19.8)
	Landsat	25.4	69.7	4.9 (2.7–10.3)

underlain by organic soils or wet sedge mats. The corresponding 1994 CH_4 fluxes for Fen and OBS were 17.3 and 9.3 mg CH_4 m^{-2} d^{-1} using the CASI data, and 13.6 and 4.9 mg CH_4 m^{-2} d^{-1} for the same two sites using Landsat (Table 5). This represents a 30–90% underestimate of CH_4 emission by using Landsat instead of CASI spatial resolution because the CASI data were better able to detect small wetlands. However, even more detailed analyses of beaver ponds showed that CASI underestimated their areal coverage, which may be as much as 3–5% of the landscape (N. Roulet, personal communication, 2004). Since beaver ponds are hot spots for methane emission [*Naiman et al.*, 1991; *Roulet et al.*, 1997; *Dove et al.*, 1999], our estimates for wetlands are conservatively low.

3.2.2. Error Estimates for Areal Extrapolation

[26] Although the Landsat classification underestimated landscape CH_4 emission compared to CASI, the associated errors due to misclassification of ecosystem types and variability of CH_4 fluxes add another layer of uncertainty to the extrapolated flux estimate. Most landscape-level studies calculate the flux uncertainty based on spatial and temporal variability of chamber measurements, but do not include the classification errors in the final landscape flux estimate because of the difficulty in combining these two types of error. *Reeburgh et al.* [1998] estimated that their landscape CH_4 emission for the Kuparuk Basin in Alaska had 50–70% uncertainty based on CH_4 flux variability and remote sensing errors in classification. *Roulet et al.* [1994] estimated 90% uncertainty in the extrapolated flux for the Hudson Bay Lowland based on a combination of flux measurement uncertainty, classification errors, and range of extreme cases of CH_4 emission. Our flux estimates for 1994 and 1996 have 45% uncertainty based on the standard deviations of the summer mean CH_4 flux (which incorporates temporal and spatial variability of the chamber measurements) (Table 4). We calculated the potential classification error in the Landsat data by estimating the range of extreme cases. In other words, based on the assumption that the Landsat image had an 80% probability of being classified correctly, we increased the area of the highest CH_4 emitting classes (small (<1 km^2) ponds, wet sedge fen, sedge fen, sedge bog, and shrub bog) by 20% and reduced the lowest emitting wetland class (treed fen) by an equivalent area. Our areal estimate for small ponds includes open water margins of collapse features, but does not include the full range of beaver ponds, and is therefore a very low estimate. We did not change the areal estimate for lakes or uplands, since the fluxes were very small in both years and did not contribute significantly to the landscape estimate. Also, the probability of mistaking a wetland class for an upland class was lower than the possibility of misclassifying wetlands among the various types of bogs and fens. The result was a 14% increase in CH_4 flux for 1994 and a 17% increase for 1996. Adding these errors to the uncertainties based on standard deviations of the flux estimates, the total uncertainty is 59% and 62% for 1994 and 1996, respectively, for the Landsat extrapolation. Assuming an additional 30–90% uncertainty based on CASI versus Landsat data, the total error for landscape CH_4 flux could be as high as 90–150%.

4. Conclusions

[27] Reliable estimation of seasonal CH_4 flux from complex landscapes, such as this boreal region of discontinuous permafrost, requires accurate estimates of the spatial and temporal variability in flux and a method of extrapolating these fluxes to the landscape using remote sensing. Chamber techniques measure fluxes for short periods, often infrequently, and thus there can be large temporal variations leading to uncertainty in the seasonal estimate. In addition, within ecological classes, there can be large variations among chamber locations. In relative terms, the spatial variability (e.g., coefficient of variation) appears to be similar for classes with small and large CH_4 fluxes (such as uplands and wetlands), but the absolute variability in flux is much greater (from <1 to >100 mg CH_4 m^{-2} d^{-1}). CH_4 flux varies little among a variety of upland classes. Thus the greatest uncertainty in CH_4 flux is associated with the wetland sites, which may occupy a relatively small proportion of the total landscape.

[28] Remote sensing techniques have difficulty in identifying and classifying small wetland areas, which may be smaller than the 30 × 30 m pixels of Landsat images. The CASI images, with pixels of 3 × 3 m, can be used to identify and map these small wetland areas, although even CASI may miss some of the small ponds. For part of the BOREAS Northern Study Area, use of the Landsat image calibrated against two CASI-image sites produced an estimate of an areally-weighted average daily CH_4 flux of 10.1 (±4.5) m^{-2} d^{-1} for 1994 and 16.3 (±4.3) m^{-2} d^{-1} for the wetter and warmer 1996 growing season, an increase of 61%. This increase was entirely attributable to wetlands, which occupy less that 30% of the landscape. This large increase was also associated with a small increase in water table position (2–5 cm) and peat temperature (0.5°C), showing how responsive CH_4 emissions can be to small changes in climate. Detailed analyses are particularly important for the zone of discontinuous permafrost, where collapse scar wetlands are very small in area, are changing rapidly due to melting permafrost, and have some of the largest CH_4 emissions because of their surface hydrology, thermal regime, and plant cover. Error estimates combining CH_4 flux variability and classification errors could be as high as 62% using Landsat data, and as high as 150%, considering the underestimate of wetland coverage in the Landsat image as compared with higher spatial and spectral resolution CASI data.

[29] Our results suggest that to improve regional estimates of CH_4 flux, there needs to be better identification of small wetlands with high CH_4 emission in the landscape, more detailed spatial and temporal measurement of fluxes within these areas, and mapping of them using fine-scale remote sensing techniques. Failure to account for these "hot spots" is likely to result in a significant underestimate of fluxes to the atmosphere.

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