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The Canadian model for peatlands (CaMP): A peatland carbon model for national greenhouse gas reporting

Kelly Ann Bona[a,*], Cindy Shaw[b], Dan K. Thompson[b], Oleksandra Hararuk[c], Kara Webster[a], Gary Zhang[d], Mihai Voicu[b], Werner A. Kurz[d]

[a] Great Lakes Forestry Centre, Canadian Forest Service, Sault Ste-Marie, ON, Canada.
[b] Northern Forestry Centre, Canadian Forest Service, Edmonton, AB, Canada.
[c] Department of Biology, University of Central Florida, Orlando, FL, USA.
[d] Pacific Forestry Centre, Canadian Forest Service, Victoria, BC, Canada

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ABSTRACT

A model framework for national greenhouse gas emission and removal estimation for Canadian peatlands (CaMP v2.0) was developed and tested. It provides a module that can work alongside the upland forest Generic Carbon Budget Model (GCBM) developed to eventually replace the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) as the core model in Canada's National Forest Carbon Monitoring, Accounting and Reporting System. The CaMP (v2.0) provides a simple model foundation that can be applied nationally for 11 different peatland categories. It tracks the growth, turnover and decay in annual time steps of different vegetation components (foliage, branches, stems, and roots of trees, shrubs, sedges and mosses). It uses a Q_{10} relationship to model peat C pool decomposition as a function of mean annual temperature, and models methane flux response to deviations in annual water table depth. The CaMP takes a simple approach to modeling hydrology for large spatial scales by using the nationally-available Canadian Fire Weather Index Drought Code to predict long-term and annual water table depth. The CaMP (v2.0) provides the framework needed to model disturbances but only includes wildfire in this version. Model behavior and sensitivity were assessed, and evaluated against observed flux data. Results suggest that the CaMP (v2.0) provides an appropriate structure for large spatial- and temporal-scale estimation of emissions, owing to the model behaving as expected relative to shifts in environmental variables, and to reasonably small mean observed to modeled residuals. Methane was overestimated by the model on average by 6 g C $ha^{-1}\ y^{-1}$ (n = 53 years of data across 11 peatland sites), and by 8 g C $ha^{-1}\ y^{-1}$ when weighted by site location (n = 12 sites, ≥ 3 years of data per site). The model overestimated net ecosystem exchange (NEE) by 20 g C $ha^{-1}\ y^{-1}$ (n = 36 years of data across 12 peatland sites), and by 2 g C $ha^{-1}\ y^{-1}$ when weighted by site location (n = 11 sites, ≥ 3 years of data per site), and results demonstrate that inter-site variation is greater than temporal variation across NEE measures. Several aspects were identified as requiring further work to increase explained variation in finer-scale emission estimates. Recommendations include further expanding the existing peatland databases to re-calibrate peat decomposition rates and better parameterize NPP rates by region for certain vegetation layers and peatland types, as well as developing a national annual-scale soil temperature model that could serve to replace the air temperature (Q_{10}) decay relationship currently used in the CaMP (v2.0). Data gaps that were identified include the need for annualized methane flux datasets with appropriate annual-scale meta-data. Future work is required to include permafrost dynamics, as well as additional natural, and anthropogenic disturbances.

1. Introduction

There are growing international requirements for improved estimates of carbon (C) within peatlands for national C budget reporting (IPCC Wetland Supplement 2013). In Canada, peatlands account for 7.3×10^5 km^2 of Canada's landscape (Webster et al., 2018), and store an estimated 150 Gt C (Tarnocai et al., 2000), equal to roughly 5 to 10% of total soil organic C storage globally (1500 to 3000 Gt C; Scharlemann et al., 2014, Köchy et al., 2015). The Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) is the core model used

* Corresponding author.
E-mail address: kellyann.bona@canada.ca (K.A. Bona).

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in Canada's National Forest Carbon Monitoring, Accounting and Reporting System (NFCMARS) and for reporting to the United Nations Convention on Climate Change (UNFCCC) on C stocks, stock changes and greenhouse gas (GHG) emissions and removals from Canada's managed forest area (Kurz et al., 1992, 2009; Stinson et al., 2011; Kurz et al., 2018). However, the CBM-CFS3 does not account for peatlands that are known to be a major contributor to the C cycle (Gorham 1991; Yu et al. 2012; Loisel et al., 2014; Nichols and Peteet 2019).

Peatlands accumulate C because their net primary production (NPP) exceeds decomposition rates (Rydin and Jeglum 2006). While peatlands can be a large sink for carbon dioxide (CO_2), they can also be a significant source of methane (CH_4) (Turetsky et al., 2014; Helbig et al., 2017), a 100-year global warming potential of which is 25 to 28 times higher than that of CO_2 (Myhre et al., 2013; IPCC 2014). The ability to calculate the balance between the uptake and emission of these two major greenhouse gases from peatlands is vital to calculating their net radiative forcing (Myhre et al., 2013). Just as important is our capacity to predict the impacts of climate change (e.g., Strack and Waddington 2007; Peltoniemi et al., 2016; Halbig et al. 2017), land-use change (Petrescu et al., 2015), and anthropogenic (Daigle and Gautreau-Daigle 2001; Rochefort and Daigle 2000; Strack et al., 2017) and natural (Wieder et al., 2009; Thompson et al., 2015) disturbances on peatland C emissions.

Northern boreal peatlands have been studied extensively in relation to factors affecting their C balance (e.g., Roulet et al., 2007; Limpens et al., 2008; Yu 2012), and have been modeled widely at many different scales (Farmer et al., 2011), from site-level process based models working at hourly time steps (e.g., PCARS; Frolking et al., 2002; MWM; St-Hilaire et al., 2010), to large-scale regional models of peat accumulation at century scales (eg. PEATbalance; Schuldt et al., 2013; Chaudhary et al., 2017). However, none of these models can be easily applied at the national scale in Canada because of the limited data available for model parameterization. Furthermore, existing peatland C models are generally independent from upland forest C estimation models, therefore there are no, or few, tools available to integrate C emissions and removals across the landscape that provide a consistent and unified approach to modeling both upland and peatland systems.

There are only three existing estimates of national GHG emissions from peatlands in Canada (Roulet et al. 2000; Kurz et al., 1992; Webster et al., 2018). The first two (Roulet et al. 2000; Kurz et al., 1992) use constant rates of C accumulation, CO_2 and CH_4 net emissions for all peatland types and multiply them by the total peatland area in Canada. In order to improve on these methods, Webster et al. (2018) took a more comprehensive approach by developing a peatland map to define the areas covered by different peatland types across Canada and compiling NEE and CH_4 emission rates for each peatland type and region from the literature. All of these methods make the simplifying assumption that one, or a few, value(s) can represent large areas over static climate conditions. These empirical estimates also cannot project emissions from natural disturbances, land-use change, and the potential effects of climate change, thus a framework for a more dynamic approach to peatland C accounting is needed.

The Canadian Model for Peatlands (CaMP) was developed to address the need for national C estimation and reporting in peatlands within Canada. An early version of the model was described by Shaw et al. (2016); here we present a fully implemented working version of the CaMP (v2.0). The CaMP (v2.0) has been developed as a module for the upland forest Generic Carbon Budget Model (GCBM), which is the next generation of the CBM-CFS3. The carbon science in the GCBM is identical to that of the CBM-CFS3 (Kurz et al., 2009) and only some of the implementation details have changed in the GCBM. Therefore, throughout the manuscript we reference the CBM-CFS3 literature for C science, understanding this applies equally to the GCBM. The main differences, and advantage, of using the GCBM framework is that it can use spatially-explicit data layers for input variables and will allow for seamless modeling of upland and peatland C dynamics because of its modular design. The GCBM is built on a collaborative open-source data integration framework, the Full Lands Integration Tool (FLINT) developed by members of the moja global team (http://moja.global).

The CaMP was developed to be consistent with modeling of moss C dynamics in MOSS-C (Bona et al., 2016), a module for the CBM-CFS3 to represent C dynamics of thick moss layers in upland forests that can occur in site types transitional between upland forests and peatlands. The CaMP has been built to require a minimal amount of inputs so that it can be applied broadly across different peatland categories in Canada. Ultimately, emission estimates from the CaMP can be used for national GHG emission estimates when used in combination with a national map product already developed (Webster et al., 2018) that defines the land-units classified as each peatland category used within the CaMP. Model outputs are intended to be consistent with the Intergovernmental Panel on Climate Change (IPCC) tables for international reporting purposes (Table 2 in Kurz et al., 2009). This version (v2.0) of the CaMP is built to simulate peatland GHG fluxes to the atmosphere (CO_2, CH_4), and other flux indicators (such as net ecosystem exchange [NEE], and heterotrophic respiration [Rh]) on the basis of a predicted long-term water table position as well as annual fluctuation in the water table, productivity of different vegetation layers, decay, and C transfers between pools. Although the current version of the CaMP does not include permafrost thaw, anthropogenic disturbances, natural disturbances other than wildfire, and some more complex finer-scale climatic and edaphic factors, its framework was developed so that they can be included in future versions, once data limitations are overcome.

In this paper we describe the CaMP (v2.0), beginning with spatial representation of land and peatland categories, modeling of biomass and dead organic matter dynamics, including CH_4 emissions and decay response to temperature and a fluctuating water table, model initialization including use of wildfire disturbances in the spin-up, calibration of decay in the acrotelm and catotelm peat layers, model behavior and sensitivity, model evaluation, and ending with identification of data gaps and recommendations for improvements in a future model version. A prototype version was implemented, tested, and calibrated using the R software (R Core Team, 2017) and then implemented as CaMP (v2.0) module in $C++11$ within the GCBM framework.

2. Spatial representation of the camp peatland categories

The basic modeling unit in the CBM-CFS3 is the “forest stand” (Kurz et al., 2009) that is defined by combinations of classifiers such as forest age, land class, productivity, forest type, and site quality (Kurz et al., 2009). The analogue modeling unit in the CaMP is the “peatland site” that is classified by a unique combination of ecozone, province or territory, and peatland category. Eleven peatland categories are used in the CaMP framework that are defined by combinations of two ecosystem characteristics; wetland type (hydrologic regime), and tree cover type (Table 1). Wetland types modeled in the CaMP are comprised of the dominant peatland types in Canada's forested area; bog (precipitation is the only hydrological input), poor fen (minor groundwater inputs), rich fen (significant groundwater inputs) and swamp (seasonal or riparian water inputs), as defined by the Canadian Wetland Classification System (National Wetlands Working Group 1997). Peat swamps were included in the modules framework, but due to the current lack of data to parameterize or calibrate the swamp class the CaMP (v2.0) was not developed for swamps. Marshes and shallow water types are not included in the CaMP (v2.0) as all wetland categories included had to meet the definition of a peatland (i.e., having greater than 40 cm of peat development; National Wetlands Working Group [1997]). Each wetland class is combined with three tree cover types (open, treed and forested) taken from Canada's definition of forest land that is consistent with the IPCC standards as implemented for Canada's national and international reporting (Stinson et al., 2011).

Table 1
Definition of peatland types and CaMP peatland categories.

Type	Definition	CaMP category	Potential tree growth at maturity Cover (%)	Height (m)
Bog	Bogs are raised above or level with surrounding areas and are dominated by *Sphagnum* moss. They are characterized as being ombrogenous; therefore, they receive water solely from precipitation, fog, or snow melt and are not influenced by groundwater or run-off from the surrounding terrain. Water is low in dissolved minerals and generally acidic (ranging from pH of 4.0 to 4.8). Peat accumulated in bogs is > 40 cm in depth and is mainly from *Sphagnum* moss mixed with woody debris from ericaceous shrubs, and, if trees are present, they are black spruce.	Forested Bog Treed Bog Open Bog	> 25 10–25 < 10	≥ 5 < 5 < 5
Fen	Fens are characterized by the flow of geogenous water from groundwater and/or various surface water sources such as lakes, streams, run-off, or spring melt. Differences in water sources and mode of water transport (e.g., via channels or open pools) create different fen surface characteristics and nutrient statuses. Peat accumulated in fens is > 40 cm in depth and is mainly derived from sedges and brown moss, as they are dominated by graminoids, dominated by bryophytes, or contain a mixture of both.			
Poor Fen	Water sources are low in base-cations, with little to no alkalinity and a high concentration of hydrogen ions leading to a poor nutrient status. Generally these fens have a pH < 5.5. Poor fens can be seen as intermediates between bogs and rich fens, and they share elements of both. They are dominated by graminoids, with some *Sphagnum* moss cover (usually > 20% cover).	Forested Poor Fen Treed Poor Fen Open Poor Fen	> 25 10–25 < 10	≥ 5 < 5 < 5
Rich Fen	Rich fens are fed by water sources that tend to be alkaline, with a pH generally > 5.5, leading to a richer nutrient status. Rich fens are dominated by sedges and brown mosses, and in contrast to poor fens, they tend to contain no or very little *Sphagnum* moss (usually < 20%) or ericaceous shrubs.	Forested Rich Fen Treed Rich Fen Open Rich Fen	> 25 10–25 < 10	≥ 5 < 5 < 5
Swamp	Swamps are minerogenous wetlands dominated by trees and/or shrubs that generally cover > 30% of the area. Peat (formed in situ) is therefore mainly derived from wood but there is often organic matter accumulated from lateral transfers, setting swamps apart from forested or treed bogs and fens. Swamps can be on organic or mineral soils. Swamps on organic soils that have > 40 cm of organic layer (or peat) are categorized as peatlands. In contrast, mineral wetland swamps (with < 40 cm organic layer depth) can be on a variety of soil parent materials ranging from sand to clay, but they are frequently on Gleysols. Swamps develop peat through basin filling where the original system was a fen or a marsh or through paludification where the original ecosystem was an upland forest. Swamps can be dominated by conifers, hardwoods, shrubs, or mixed wood/shrub. Swamps include a wide range of nutrient regimes.	Forested Swamp Treed Swamp	> 25 10–25	≥ 5 < 5

Forested peatland categories only include sites with large trees (potential height ≥ 5 m and > 25% cover at maturity), such as the CaMP forested bog category that includes sites that are potentially transitional to upland black spruce sites with shallow peat layers. Broadly, treed peatland CaMP categories are dominated by small trees (potential height < 5 m, 10–25% cover), and the open peatland CaMP categories are dominated by a shrub or sedge layer that can contain tree species and sparse small trees (potential height < 5 m, < 10% cover) (Table 1).

Minimum inputs required for each site are area (ha), peatland category and location, with polygon boundaries required if results are to be mapped. Default model parameters are provided for each peatland category that captures large-scale variation in growth, turnover, and decomposition. These default parameters were estimated based on literature-cited values from 186 peatland study sites compiled into the "Peatland Decomposition and Productivity Parameter Database" (Bona et al., 2018). The CaMP is structured so that users can specify their own model parameters for combinations of peatland category and jurisdiction (province or territory) and ecozone (ESWG; Ecological Stratification Working Group, 1995) if they have the appropriate data to do so. This structure will also allow CaMP developers to include finer resolution default model parameters when more data become available in the future. For climate parameters the finest resolution data available are used depending on the variable in question (10 km resolution for mean annual temperature (Hopkinson et al., 201; McKenney et al., 2011; https://open.canada.ca/data/en/dataset/d432cb3d-8266-http://4487-b894-06224a4dfd5), and 9 × 9 km resolution for the fire weather index drought code (NARR; North American Regional Reanalysis; Jain et al., 2017; https://www.publish.csiro.au/wf/WF17008).

3. Biomass and decay dynamics

The CaMP simulates growth, biomass turnover (mortality), transfer and decomposition sequentially in annual time steps. The following text describes the peatland C pools, transfers between those pools, and C emitted to the atmosphere, over the course of peatland development in the absence of a disturbance.

3.1. Pools

The CaMP tracks a total of 18 pools; seven live biomass and 11 dead organic matter pools (Fig. 1). Biomass pools represent three different vegetation categories: woody vegetation (trees and shrubs), sedges, and mosses. Biomass C pools transition to appropriate litter pools (e.g., live sedge foliage C is transferred to dead sedge foliage C) at pool-specific turnover rates. After some decomposition, litter pools are deemed buried as peat and transferred into the acrotelm pools, eventually transferring into the catotelm pools.

The CaMP shrub vegetation includes small evergreen shrubs (such as *Ledum groenlandicum* (Oeder) Hulten.), small deciduous shrubs (such as *Salix pedicellaris* Pursh.), and tall deciduous shrubs (such as *Alnus crispa* (Aiton) Pursh.). The shrub vegetation category is split between different plant components to form three pools: foliage, stems and branches, and roots. Sedges (such as *Carex rostrata* Stokes.) are split between foliage, which includes all aboveground parts, and roots. Mosses in the CaMP are split between two different groups: *Sphagnum* spp. mosses (such as *Sphagnum fuscum* (Schump) Klinggr.), and feather mosses (such as *Pleurozium schreberi* (Brid) Mitt). The feather moss category conceptually includes all peat forming mosses with the exception of *Sphagnum* mosses that are modelled separately. Feather moss was kept separate in order to facilitate the transition between the CBM-CFS3 MOSS-C module that also keeps track of both *Sphagnum* and feather moss (Bona et al., 2016) as two C functional groups (Bona et al., 2013) due to the differences in litter quality, which affects peat physical and chemical properties (Turetsky et al. 2003; Turetsky et al., 2008), decomposition rates (Fenton et al., 2010), and fire consumption rates (Shetler et al., 2008).

The GCBM is used to simulate small (< 5 m height potential) and large (≥ 5 m height potential) live tree biomass growth for treed and forested peatlands, respectively (Table 1). The annual litter fall and mortality transfers from the live tree pools generated within GCBM are calculated and the C is transferred into the appropriate CaMP dead woody pool where decomposition is simulated within the CaMP module.

Fresh litter from each of the live pools is transferred to a

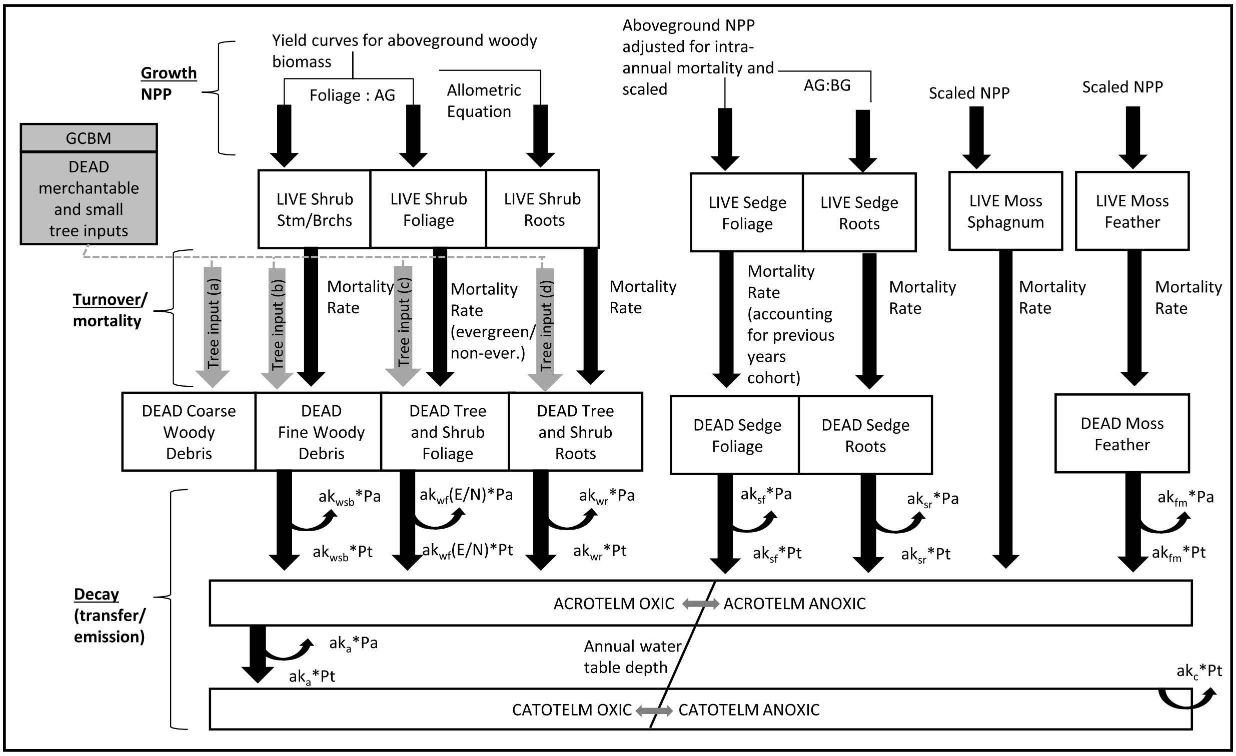

Fig. 1. The Canadian Model for Peatlands version 2.0 (CaMP v2.0) model structure. Carbon (C) from live biomass production is calculated based on the annual net primary productivity (NPP) of different vegetation types and components: LIVE Shrub Stm/Brchs, Shrub stems and branches; LIVE Shrub Foliage, shrub foliage; LIVE Shrub Roots, shrub roots; LIVE Sedge Foliage, aboveground parts of sedges; LIVE Sedge Roots, belowground parts of sedges; LIVE Moss *Sphagnum, Sphagnum* moss; and LIVE MOSS Feather, non-*Sphagnum* mosses. Annual mortality of live biomass pools results in turnover of C transferred from live to dead pools (fresh litter): DEAD Coarse Woody Debris; fallen snags from merchantable sized trees; DEAD Fine Woody Debris, stemwood and branches of small trees and shrubs; DEAD Tree and Shrub Foliage, shrub and tree foliage; DEAD Tree and Shrub Roots, shrub and tree root; DEAD Sedge Foliage, aboveground sedge; DEAD Sedge Roots, belowground sedge; DEAD Moss Feather, non-*Sphagnum* moss. Live and standing dead biomass C from large merchantable sized and small non-merchantable trees is simulated within the Generic Carbon Budget Model (GCBM) and turned over directly into the CaMP Dead pools when reaching the peat surface (grey arrows): Tree input (a) standing large tree snags, (b) small tree stemwood and branches, and merchantable tree branches and other pools, (c) small and merchantable tree foliage, (d) small and merchantable coarse and fine roots. Carbon from modelled decomposition is either emitted to the atmosphere as CO_2 or CH_4 (at a proportion, Pa) or transferred to peat pools (at a proportion Pt): Separation of the acrotelm (oxic and anoxic) from the catotelm (oxic and anoxic) is based on long-term water table depth. The anoxic acrotelm and oxic catotelm pools are used to simulate changes in decomposition resulting from annual water table movement; the anoxic acrotelm stores C only when the annual water table is higher than the long-term water table depth, and the oxic catotelm stores C only when the annual water table depth is below the long-term water table depth.

corresponding dead pool: dead tree and shrub foliage, dead fine woody debris, dead coarse woody debris, dead tree and shrub roots, dead sedge foliage, dead sedge roots, and dead feather moss. These dead pools act as transition pools that decay relatively quickly in comparison to the less labile peat in the acrotelm and catotelm pools. Conceptually the C in these pools is ready to transfer to the acrotelm once it is deemed buried by mosses and sufficiently incorporated into the peat, determined by the pool's turnover time. The *Sphagnum* moss category is the only live pool which is transferred directly to the acrotelm, because *Sphagnum* mosses bury themselves as they grow, making it difficult to clearly define a dead *Sphagnum* moss pool.

In a peatland the physical split between the acrotelm and the catotelm layer is generally defined as the split between oxic and anoxic conditions. For consistency and for modeling purposes (see Section 3.2 Water table depth) we used the well-regarded definition of the acrotelm from Ingram (1978, 1982), where the acrotelm depth is expressed as the maximum water table depth, i.e., the water table at the lowest position under drought conditions, such that the acrotelm is the oxic layer (or at least periodically oxic) and the catotelm is the consistently water saturated anoxic layer. In the CaMP, we use an operational definition of drought being the driest summer conditions found every 5 years (80th percentile of water table depth), corresponding to "moderate" drought in the North American Drought Monitor framework (Svoboda et al., 2002). This definition is supported by Morris et al., 2011, who suggested annual maximum, not extreme (i.e., 100-years) maxima water depths, as being appropriate for the acrotelm definition, and suggested a time frame of circa 5 years is most fitting.

The degree of water saturation has a large impact on the decomposition rates of these pools (Beer and Blodau, 2007). The CaMP models two conceptual water tables: the long-term climatic water table depth (100–8000 years) as well as the estimated annual water table depth when the model is run annually for contemporary C estimation. The long-term water table is used to determine the lasting boundary between the acrotelm and catotelm, which ultimately impacts the litter quality between these layers and their resulting decomposition rates. The annual water table depth is used to track the fluctuation in inter-annual water table and the resulting level of water saturation that also significantly impacts decay. To accomplish this, the model first predicts the long-term water table depth that is used to determine the oxic acrotelm and anoxic catotelm pools. The module then tracks the movement of the annual water table across these pools by using temporary holding pools such that C that is under water within the acrotelm for a particular year is placed in the anoxic acrotelm holding pool, and similarly, C that is exposed to oxygen within the catotelm is placed in the oxic catotelm holding pool for that year. We are aware that because the anoxic catotelm pool is defined by the 80th percentile of the long-term water table depth it will have an oxic portion present only once in every five years, corresponding to a moderate drought under the North

American Drought Monitor framework (Lawrimore et al., 2002). Regardless, an oxic portion of the catotelm was included in order to account for drought years outside the upper range of long-term weather conditions, which may be of particular importance in a changing climate or peatland drainage scenario.

3.2. Water table depth

Water table depth in the CaMP is calculated as a function of the Canadian Fire Weather Index Drought Code (DC), which has been shown to approximate water table decline in peatlands, as it follows a simple one-dimensional water balance approach (precipitation and evaporation) that mirrors the water balance of peatlands (Waddington et al., 2012). The relationship used for the CaMP was based not on the small sample of peatland water table time series from Waddington et al. (2012) but rather on single water table measurements for many ($n = 296$) sites from Zoltai et al. (2000) where both records of precise peatland type and Drought Code from nearby weather stations are available. These single measurements of peatland water table per site were then related back to long-term DC values for the sampling date to yield a simple linear model (Eq. (1)) describing the relationship between water table (WT) depth (cm) and DC in different peatland categories,

$$WT = -0.045DC + b_{w,c} \tag{1}$$

where b (cm) varies by the peatland sites wetland class (bog, poor fen, rich fen) (w) and canopy class (c) as follows: −12.5 for open bogs, −25.9 for treed bogs, −29.2 for forested bogs, 0.7 for open poor fen, −12.7 for treed poor fen, −16 for forested poor fen, 5.6 for open rich fen, −7.8 for treed rich fen, and −11.1 for forested rich fen. This model explains 52% of the variance in water table depth, has a residual standard error of 13 cm, and is highly significant ($p < 0.001$). As the acrotelm can be approximated by the peat layers exposed to aerobic conditions during periods of deep water table decline, we calculated the annual maximum DC for every point in the forested area of Canada using a 9-km weather grid to compute DC from the NARR reanalysis (Jain et al. 2018) from 1979–2015. The 80th percentile of the maximum DC value was used to compute the water table using Eq. (1), and that distance below the surface was taken as the acrotelm thickness. Note that the relationship (Eq. ([1])) used to predict water table depths only predicts a range of estimated annual minimum water table depths (Table A14) that correspond to the observed range of potential Drought Codes (78.5 to 411.8) for Canada at the time of the peatland water table observation (Supplement S1).

We acknowledge that a more complex approach to water table modeling could have been employed via advanced and surface models such as that described by Bechtold et al. (2019) that can simulate daily to monthly decline in water table depth, but this was deemed too detailed and parameter-heavy for the first stage of the CaMP model framework development. In addition, the model of Bechtold et al. (2019) was not available when the CaMP model design was created. Instead we chose to start with a peatland-type and weather driven model of water table depth but more sophisticated process-based models could be explored for use in future versions.

3.3. Carbon density curves

The C that is transferred to the oxic catotelm and anoxic acrotelm holding pools, as the annual water table fluctuates, is calculated based on standardized C density curves for each peatland category. Carbon density curves were modeled on 336 peat cores with 5017 core segments with bulk density data from Zoltai et al. (2000). Relationships from Bauer et al. (2006) were used to convert bulk density to C density. The Zoltai database sites were classified into the 11 CaMP peatland categories based on vegetation observations and peat properties. Sites where more than one peat core was collected were treated as fully independent as these sites were sampled across different topographic forms (e.g., strings or flarks of a patterned fen). Peat samples at differing depths from the same core are not truly independent samples, so linear mixed effects models should be used (Treat et al., 2016; Morris et al., 2015) in studies of peat properties to accounted for within-core effects. Therefore, we used linear mixed effect modeling via the *lme4* package in R (Bates et al., 2015) with depth, wetland class, and tree cover class as fixed effects to model C density; both the core as well as sampling year were included as random effects in the model. Depth class (number of 15 cm intervals from the surface) was transformed by the natural logarithm to account for the large increase in bulk density just below the surface in most peatland ecosystems (Thompson and Waddington 2014). A leave-one-out cross-validation analysis was used on individual samples ($n = 4096$), but excluding any samples from the same core in the cross-validation.

Predictions of C density at 15 cm increments down to 3.0 m depth were made for each of the 11 peatland categories using the complete version of the linear mixed effect model. Note that we do not limit the depth of the peat profile to 3.0 m elsewhere in the CaMP, and that the C density curves limit of 3.0 m is irrelevant because curves are used to calculate transfers between oxic and anoxic pools and the annual water table depth never approaches a 3.0 m depth. Both instantaneous C density (per 15 cm depth interval by peat class) and cumulative (top-down) C density were calculated. Instantaneous C density (kg C m^{-3}) from the linear mixed effects models was fitted to a continuous function to facilitate predictions for depth increments different than the 15 cm used in the model:

$$[C_{w,d,z}] = d(ln(z)) - e \tag{2}$$

where *w, d*, and *z* represent the wetland class, canopy cover class, and depth interval, respectively. The coefficients *d* and *e* were fitted using a non-linear least squares procedure from the R package *nls* with the median and standard error of each parameter estimate recorded (Table 2). Similarly, cumulative C density was fit to the simple exponential function:

$$\sum C_{w,d,z} = a(z^b) \tag{3}$$

A 95% confidence interval for the fitted functions was calculated for cumulative C load up to 3.0 m, and compared to other peatland types in order to determine meaningful differences in standard density curves between peatland types (Table 3). Individual C density curve functions (Table 2, 3) using Eqs. ([2]) and [3] above had very low standard errors for the parameters *a* through *d*, owing to the use of model prediction data in the curve fitting, rather than the population of cores from the 336 cores used to build the linear mixed effects model. Cumulative C density curves generally followed nutrient gradients, with open bogs have the lowest C loading (154 ± 19.7 kg C m^{-2}) compared with the aggregate of all rich fens and swamps, which as an aggregate had a C

Table 2
Instantaneous carbon density curve fitting model parameters from equation [2] by the Canadian Model for Peatlands (CaMP) peatland categories.

CaMP category	d value	SE	e value	SE
open poor fen	0.0089245	0.000186	−0.0078673	0.0008946
open bog	0.0126573	0.0001793	0.008555	0.0008627
treed poor fen	0.0088893	0.0002065	−0.0107837	0.0009935
treed bog	0.0127787	0.0002465	0.0064283	0.0011858
open rich fen	0.0081488	0.0001798	−0.0164793	0.0008652
forested poor fen	0.0089245	0.000186	−0.0128673	0.0008946
forested bog	0.0126573	0.0002606	0.003555	0.0012534
treed rich fen	0.0080689	0.0001922	−0.0196068	0.0009248
forested rich fen	0.0081488	0.0001798	−0.0214793	0.0008652
treed swamp	0.0110348	0.0002312	−0.0141545	0.0011124
forested swamp	0.011051	0.000221	−0.016177	0.001063

SE = standard error of the estimate of variable.

Table 3
Cumulative (surface down) carbon density curve fitting model parameters from Eq. (3) by the Canadian Model for Peatlands (CaMP) peatland categories.

CaMP category		a				b				Cumulative C load up to 3m[1] (kg C m^{-2})		
	# cores	value	SE	CI lower limit	upper limit	value	SE	CI lower limit	upper limit	median	CI upper limit	lower limit
open bog	13	0.142	0.0019	0.146	1.234	1.229	0.0019	1.224	1.234	158.2	166.9	149.5
treed bog	42	0.165	0.0037	0.172	1.221	1.213	0.0037	1.204	1.221	167.1	182.6	151.6
forested bog	35	0.184	0.0044	0.193	1.209	1.200	0.0044	1.191	1.209	173.8	190.8	156.8
open poor fen	25	0.231	0.0072	0.246	1.151	1.140	0.0072	1.128	1.151	154.7	174.5	135.0
treed poor fen	12	0.260	0.0086	0.277	1.141	1.128	0.0086	1.116	1.141	163.5	185.5	141.5
forested poor fen	10	0.284	0.0094	0.302	1.132	1.120	0.0094	1.108	1.132	170.2	193.1	147.3
open rich fen	95	0.313	0.0108	0.334	1.116	1.103	0.0108	1.090	1.116	169.9	193.9	146.0
treed rich fen	44	0.345	0.0121	0.369	1.107	1.094	0.0121	1.081	1.107	178.6	204.2	153.0
forested rich fen	37	0.370	0.0133	0.397	1.102	1.088	0.0133	1.075	1.102	185.4	212.6	158.3
treed swamp	4	0.328	0.0107	0.350	1.139	1.127	0.0107	1.115	1.139	204.6	231.9	177.4
forested swamp	19	0.352	0.0116	0.375	1.133	1.120	0.0116	1.108	1.133	211.4	240.0	182.8

SE; standard error of the estimate of variable, CI; 95% confidence interval.

[1] The above values are fitted to the model prediction from the Linear Mixed Effect model, and do no represent individual cores, but rather the median model prediction of C density per depth interval per peat type.

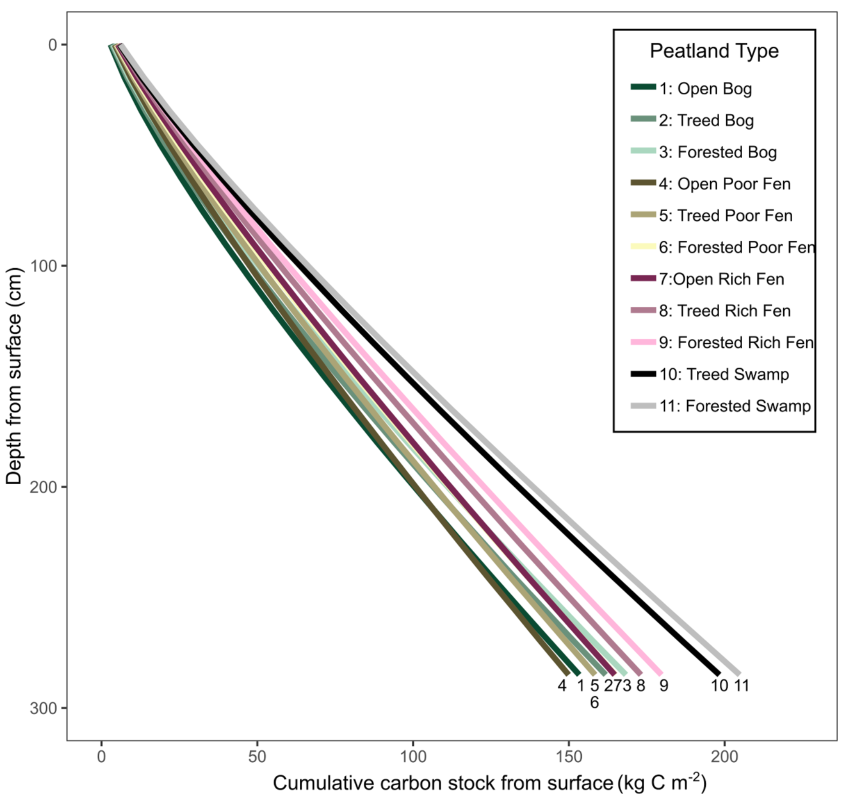

Fig. 2. Cumulative carbon density from the surface down to 300 cm for the CaMP peatland categories. Note that the curves are the result of the inversion of the linear mixed effects model, and represent the mean model response at various depths, and not raw observed data from the Zoltai et al. (2000) database. See Table 3 for parameters and number of cores used.

loading between 178 and 211 kg C m^{-2} (Table 3, Fig. 2).

3.4. Growth

3.4.1. Tree layer growth

The GCBM uses merchantable stemwood volume yield curves as input to simulate stand-level merchantable sized (> 5 m height potential) tree growth over time. Then allometric equations are used to calculate other aboveground tree biomass pools (Boudewyn et al., 2007) as a function of stem volume (Kurz et al., 2009). We use the following default types of curves for each forested peatland category, but in the future users will have the ability to provide their own yield curves. For the forested bog peatland category, we use a default pure black spruce *(Picea mariana* (Mill.) B.S.P.) yield curve, for poor fens we use a black spruce forest type curve which assumed black spruce as a lead species but includes a small proportion of larch (*Larix laricina* (du Roi) K. Koch), and for forested rich fens we use a larch forest type curve which includes a small proportion of black spruce (Penner et al., 2008).

Small tree growth (≤ 5m height potential) was simulated in GCBM similarly to merchantable-sized trees, except that total tree volume yield curves were used instead of merchantable wood volume yield curves. Plot data from stands located on peatlands in Northern Alberta were compiled for total stem volume, and an empirical volume over age yield curve was developed using a model based on Hoerl's special function (Daniel and Wood 1980).

3.4.2. Shrub layer growth

In keeping with the CBM-CFS3 framework, the CaMP models shrub layer growth using simple shrub net growth curves where aboveground mass increases linearly to a maximum value and then levels off. Values estimated for maximum biomass (g biomass m^{-2}) vary by peatland category and were calculated as the mean of values reported in the literature (Bona et al., 2018) (Table A1). Tall shrub maximum biomass data were not available for each peatland category therefore a mean for all peatland categories was used and scaled for each peatland category by multiplying the mean biomass per unit area by the relative cover of tall shrubs for each peatland category from Zoltai et al. (2000) (Tables A2, A3). It is assumed that the maximum biomass after establishment is reached at three years for low shrubs (Johnston et al., 2015; Table A1) and at 20 years for tall shrubs (Wieder et al., 2009; Table A2, A3). The aboveground shrub C is split between the shrub stems and branches pool, and the shrub foliage pool, by using a foliage to aboveground biomass ratio (set at a default value of 0.265) calculated as a mean from Bona et al. (2018). This ratio is used for all shrubs, although it is based solely on low shrubs since data to calculate a tall shrub value were not available.

Net shrub layer root growth was calculated as a function of shrub layer stems and branches using Eq. ([4]) from Murphy et al. (2009),

$$\mathrm{rootproduction} = a(\mathrm{stembiomass}) + b, \tag{4}$$

where $a = 0.75$ and $b = 0.02$, $R^2 = 0.62$. This equation was chosen because it does not vary by water table or site type and included fine roots as well as some coarse roots. Similar allometric equations by Murphy and Moore (2010) that were derived for the Mer Bleue bog were tested because of geographical proximity, but eventually were rejected in favor of a function from a study in Finland (Murphy et al., 2009) because the Mer Bleue study did not include coarse roots.

3.4.2. Sedge growth

Annual net sedge growth was calculated as an adjusted and scaled NPP value (aNPPs) based on a review of peatland sedge dynamics. First, a total mean maximum standing crop biomass of 220 g biomass m^{-2} for open bogs and fens was calculated using the data in Bona et al. (2018). There were few data for treed systems so they were not included in this mean. Instead values for treed systems were calculated as a mean maximum standing crop biomass resulting from scaling the total mean maximum standing crop biomass (220 g biomass m^{-2}) using percent ground cover values from Zoltai et al. (2000) for each CaMP peatland category (Table A4). The scaled standing crop biomass was then multiplied by an adjustment factor currently set at a default value of 1.1 (Bernard and Hankinson 1979). This adjustment factor is used to correct for intra-annual mortality. Authors Bernard and Hankinson (1979) found that the quadrat method for sampling sedge biomass (the method used for all compiled data in Bona et al., [2018]) only accounts for live biomass in the growing season of the measurement year. However, since a portion of the sedge crop grows and then dies during that season one must not only account for the observed growth, but also the growth from sedges that have grown and died within the growing season, to get a true measure of NPP. In accounting for this the biomass:productivity ratio increases from 0.85 to 1.1 (Bernard and Hankinson et al. 1979, *Carex rostrada*), and in another study of *Carex lucustris* authors cited an increase from 0.82 to 1.5 (Bernard and Macdonald 1973). These studies were conducted in marshes, but no similar studies were found for bogs and fens so the default value for bogs and fens is based on the reported values for marshes. An adjustment factor of 1.1 is used in the CaMP to adjust the scaled mean live sedge standing crop biomass C assuming that intra-seasonal mortality for *Carex rostrata* and other sedge species is similar within fens and bogs.

To calculate the net growth of sedge roots, aNPPs is divided by a mean aboveground to belowground ratio for sedges found in the literature, initially set at a default value of 0.3. This was estimated from a range of literature-cited values (0.09–0.56) where most studies observed that roots contribute a relatively large portion to the total NPP (Bona et al., 2018).

3.4.3. Moss growth

Mean NPP for *Sphagnum* and feather mosses by peatland category were estimated using data compiled in Bona et al. (2018) (Table A5) and used to model moss growth. These baseline NPP rates were scaled for some peatland categories (e.g., rich fens) because field studies citing moss NPP rates in the literature were not scaled-up to the plot level. Estimates of NPP from field studies for bogs and poor fens were used without scaling because in these sites peat mosses cover a large portion of the ground, and are more spatially homogenous therefore making them relatively accurate estimates of the plot-level NPP. However, for peatland categories where mosses typically have low ground cover, baseline NPP rates were scaled (Table A6) by multiplying them by the relative% cover values calculated from Zoltai et al. (2000).

3.5. Biomass turnover

As in the CBM-CFS3, the CaMP represents biomass mortality and litterfall using an annual turnover rate (% mortality yr^{-1}) that determines the amount of C transferred annually from each live pool into the appropriate dead pool.

The shrub branches and stem pool turnover rate is calculated based on the assumption that the net growth of shrub branch and stems would reduce to zero after being established (i.e., after three years for low shrubs and after 20 years for tall shrubs). Net growth is equal to NPP minus mortality, so when net growth equals zero then annual NPP should approximately equal the annual mortality rate for any mature site. Therefore, annual NPP rates from Bona et al. (2018) were averaged for each peatland category (Table A7) and assumed equal to annual mortality. These mortality rates by peatland category were then divided by mean biomass values for branches and stem biomass in Bona et al. (2018) for each peatland category to solve for the annual turnover rate (% mortality yr^{-1}) for each of these biomass components. Shrub root mortality was set at a default value of 0.56 from the study by authors Laiho et al. (2003).

Litterfall from the shrub foliage pool is also simulated annually by the CaMP. The proportion of evergreen to non-evergreen foliage was calculated as a mean of ratios in Bona et al. (2018) for each CaMP peatland category (Table A8). The current default in the CaMP is that 100% of non-evergreen leaves and 25% of evergreen leaves (Aerts et al. 1989) transfer to dead pools annually.

The CaMP sedge turnover rate is currently set to a default value of 0.61 yr^{-1} cited by authors Saarinen et al. (1996) that is also similar to the turnover rate of 0.6 yr^{-1} cited in Grigal et al. (1985). The CaMP assumes that C accumulated in the live *Sphagnum* and feather moss pools in the previous year (t_1) is completely transferred as litter to the acrotelm in the next year (t). The assumption that total aboveground moss NPP is equal to moss litter input is also used in other peatland models (e.g., Frolking et al., 2002, 2010).

3.6. Decay dynamics

As in the CBM-CFS3, the CaMP has a built in temperature-dependent decay rate that calculates the amount of organic matter that decomposes in each dead pool annually. The applied decay rate (a_k) of any dead pool is calculated using Eq. ([5]),

$$a_k = k \times \mathrm{TempMod}, \tag{5}$$

where k is the base decay rate for each dead pool at the reference temperature of 10 °C, and *TempMod* is a temperature modifier. *TempMod* is calculated using Eq. ([6]),

$$\mathrm{TempMod} = e^{((\mathrm{MATi}-\mathrm{RefTemp})\times\ln(Q_{10})\times0.1)}, \tag{6}$$

where MAT is the mean annual temperature for the *i*th spatial unit (MAT_i), *RefTemp* is the reference temperature (10 °C), and Q_{10} is the temperature coefficient.

Exponential decay rates (y^{-1}) for different litter types were compiled (Bona et al., 2018) (Table A9–A13). Appropriate Q_{10} values were difficult to determine from the literature, especially since the Q_{10} value for peat have been shown to vary greatly between sites (between 2.2

and 19 in Chapman and Thurlow 1998) and to vary with changing water table depth (Silvola et al., 1996). Therefore, decay rates and Q_{10} values for the acrotelm and catotelm pools were estimated using an Bayesian Markov Chain Monte Carlo technique to find values of decay rates and Q_{10} that maximized the likelihood function, which was expressed as a function of the difference between modeled and measured estimates of C stocks in each of these pools (see Section 6. Calibration of peat decay parameters). After the amount of decayed C in each dead pool is calculated, a proportion of the C is transferred to the atmosphere (P_{atm}), and the rest is transferred to the appropriate pool of lesser chemical potential energy as humified material (P_t). The proportion of C released to the atmosphere due to decomposition is split into CO_2 and CH_4.

3.7. Methane emission

Modeling peatland CH_4 emissions posed a significant challenge because flux rates reported in the literature have high spatial and temporal variability and are dependent on highly complex dynamics (Arneth et al., 2010; Wania et al., 2010). The amount of CH_4 that reaches the atmosphere from any peatland is a function of how much of it becomes oxidized while being transported through the peat column (Edwards et al., 1998). This transport process can be very complex and depends on factors such as the rate of diffusion (Moore et al. 1990), ebullition (Shannon and White 1994) and plant-mediated transport (Schimel 1995). Taking into account these complex dynamics in a process-based CH_4 model across several regions in Canada would require a large number of parameters that are not available. Instead, for the CaMP (v2.0) we use the approach described in a meta-analysis study by Turetsky et al. (2014), whereby an optimal water table depth that corresponds to maximum CH_4 flux (F_{CH4max}) is defined for each wetland type (bog, poor fen, rich fen). The optimal water table depths and their corresponding F_{CH4max} rates grouped by wetland type (bog, rich fen, and poor fen) within the boreal described by Turetsky et al. (2014) are used here. Data from a peatland drainage study by Strack et al. (2010) are then used to model response to deviations from the optimal water table depths, which we refer to as the F_{10}. The F_{10} value is analogous to a Q_{10} value, such that it solves for a decrease in CH_4 production within a system that corresponds to a 10 cm decrease or increase in water table depth. In the CaMP (v2.0) we assume a symmetrical response for all peatland categories (and use two F_{10} values, one for water table increase from optimum (0.32) and one for water table decrease (2.6) based on within-site responses to water table change from Strack et al. (2010) rather than the between-site water table-CH_4 production responses from Turetsky et al. (2014).

Initially the CH_4 model used regionally-specific F_{CH4max} rates collected from the literature for each ecozone (Ecological Stratification Working Group 1996) and wetland type, in order to improve expression of regional variation. However, this version of the model consistently underestimated CH_4 likely because of low sample sizes to estimate F_{CH4max} rates for each wetland type (bog, poor fen, rich fen) and ecoregion combination, so the decision was made to use the F_{CH4max} rates reported in Turetsky et al. (2014) for the entire boreal until more data can be compiled. The drawback of using this approach is that the values provided by Turetsky et al. (2014) are based on daily measures of both water table and CH_4 flux. To make them useful for the CaMP the flux rates had to be scaled to annual rates following the method of Webster et al. (2018). On a daily scale, water table varies significantly more than on an annual scale, meaning that daily optimal water table depths sometimes fall outside the range of the mean annual water table positions (Supplement S1). The result is that optimal water table depths for maximum CH_4 flux used here (daily) for poor fen and rich fen systems (2 cm, 16 cm) are never reached within the range of potential (annual) water table depths for these systems that correspond to the annual minimum water table (-3 to -35 and -3 to -30, respectively; Appendix Table A14), though the peatland systems can be observed at water tables above the surface (Supplement S1). Despite this issue, the CH_4 scheme used in the CaMP (v2.0) provided better results on a large-scale than any other schemes we tested including a simple linear relationship with water table depth defined in the original CaMP design document (Shaw et al., 2016).

4. Model initialization

Before running a CaMP simulation, C pools need to be given initial C stock values, which in the CaMP (v2.0) are the steady state C stocks (with the exception of the catotelm pool) adjusted for the effects of the last stand-replacing wildfire. In the CBM-CFS3 C pools are initialized by simulating repeated cycles of forest stand development until pools reach a semi-equilibrium where the annual change in soil pools is less than 1% (Kurz et al., 2009). For the CaMP, we calculate the steady state C pools analytically, because it would be too computationally taxing and require too long to spin-up the peat pools with repeated disturbance cycles due to low decay rates of the peat pools.

Following the logic in Weng et al. (2012), C dynamics across all ecosystem compartments can be expressed as

$$\frac{dX(t)}{dt} = U(t) - AKX(t) - fSMX(t) \tag{7}$$

where $X(t)$ is a row vector of C pools in ecosystem compartments; $U(t)$ is a row vector of C inputs to live pools (tree leaves, wood, and roots; sedges; feather, and *Sphagnum* mosses); A is a matrix of C partitioning coefficients among the ecosystem compartments; K is a matrix of C turnover rates; f is the frequency of the stand-replacing fires; S is the diagonal matrix of the fractions of C pools removed by a stand replacing fire; and M is a matrix of C partitioning coefficients that allocates C that was lost from a donor pool as a result of fire among the receiving pools (e.g., when fire kills the aboveground biomass, live roots, or a donor pool, are transferred to a litter pool, which is a receiving pool). To obtain steady state X_{ss}, we first calculated steady state C inputs (U_{ss}) from GCBM simulations of tree foliage, woody, and root pools; feather moss and *Sphagnum* NPP's (Table A5); and NPP derived from steady state shrub biomass (Table A1,2) and associated partitioning coefficients and biomass turnover rates. Afterwards, we set the values in the vector $\frac{dX(t)}{dt}$, which represented the annual changes in C pools, to 0, set $U(t)$ to U_{ss}, and solved Eq. ([7]) for X_{ss} as described in Xia et al. (2012). Afterwards the catotelm pool was adjusted to represent the C pool size after 8000 years of C accumulation that is the approximate time since peatland vegetation initiated in Canada (Koropchak et al. 2012). As for other pools, dynamics for the catotelm pool is formulated as

$$\frac{dx}{dt} = C_{in,ss} - (k + f \times s)x \tag{8}$$

where x is the C pool, $C_{in,\ ss}$ is the steady stead C input from the donor pools, k is the decay rate of the pool, f is the frequency of disturbance, and s is the severity of disturbance of the given pool expressed as a fraction ranging from 0 to 1. Eq. ([8]) is a separable differential equation and therefore can be analytically solved for x at the time t, yielding

$$x(t) = \frac{C_{in,ss}}{k + f \times s} - e^{-(k+f\times s)\times t+const} \tag{9}$$

where $\frac{C_{in,ss}}{k+f\times s}$ is the steady state pool size, and $e^{-(k+f\times s)\times t+const}$ is the correction to the steady state pool to obtain the pool at the given time t. Because initial pool size, or pool size when $t = 0$, for the catotelm is 0, e^{const} will equal to the steady state pool size, $\frac{C_{in,ss}}{k+f\times s}$. Therefore, the new expression for $x(t)$ becomes

$$x(t) = \frac{C_{in,ss}}{k + f \times s}(1 - e^{-(k+f\times s)\times t}) \tag{10}$$

From Eq. ([10]) it follows that the time it takes for the given pool to reach a value within 0.1% of its equilibrium is dependent on the decay rate of the given pool as well as the frequency and severity of disturbances. Unlike for other pools in the CaMP, which reach equilibrium after less than 4400 years, it takes more than 8000 years for catotelm C pool to reach equilibrium, because the catotelm is assumed to remain undisturbed after fire ($s = 0$) and its decay rates are extremely slow (see Table 5 for k magnitudes). Therefore to avoid overestimation of the catotelm C pool we needed to adjust the steady state estimate for the catotelm in the calculated vector X_{ss} as follows:

$$X_{ss,catotelm} = X_{ss,catotelm} \times \left(1 - e^{-k_{catotelm} \times 8,000}\right) \tag{11}$$

Lastly, once all C pools in X_{ss} were obtained, we simulated a stand replacing wildfire, ran the model forward to the time since the last stand-replacing disturbance assigned to the site, and used the $X(Age)$ as the initial values for model simulations. This accelerated initialization method was validated by running the model for 30,000 annual time steps, including fire at the appropriate fire return interval, and comparing the steady state values from the solution with the spin-up values from the method used in the CBM-CFS3 (Kurz et al., 2009) but applied in the CaMP (v2.0).

5. Wildfire disturbance

In the CaMP (v2.0) wildfire is the only disturbance type included at this time, that affects biomass turnover and transfers between dead pools as well as the atmosphere. Each time a fire is simulated, a wildfire disturbance matrix is applied that determines the amount of C transferred from one pool to another or emitted from a pool to the atmosphere (as CO_2, CO or CH_4). A disturbance matrix is a table where columns are designated for each potential C source pool in the model (e.g., live sedge foliage or dead sedge roots), and rows are assigned to each potential sink pool (e.g., anoxic catotelm or atmosphere pool) and the intersection of the source and sink (column × row) defines the proportion of C moved from source to sink for a given disturbance event, such as fire in this case (Kurz et al., 2009; Kurz et al., 1992).

We assume that the live sedge roots and live shrub roots will not be combusted by fire because all, or most of them, will be protected by water saturated peat layers. Instead, 100% of live shrub roots are assumed to die and directly transfer to the acrotelm C pool, while live sedge roots remain alive. All other live pools are assumed to be combusted by fire: live shrub stems and branches, live shrub foliage, live sedge foliage, live *Sphagnum,* and live feather moss. Dead pools that are easily consumed by fire such as dead tree and shrub foliage, dead sedge foliage and dead feather moss (Shetler et al., 2008) are completely combusted and the C emitted to the atmosphere. The oxic acrotelm (and oxic catotelm holding pool) is assumed to be only partially combusted at a rate of 12.5% for bogs and 11.4% for poor and rich fens. Rates were calculated based on data from Lukenbach et al. (2015) and Turetsky et al. (2011) (Table 4). Other dead pools that are assumed to be partially combusted by a fire at the same rate as the oxic peat are: Dead Fine Woody Debris, dead coarse woody debris, dead tree and shrub roots, and dead sedge roots. None of the anoxic catotelm (or anoxic acrotelm) pool is combusted by fire since it is water saturated and protected from burning.

Each time C in a CaMP pool is combusted by a wildfire simulation the C is emitted to the atmosphere as either CO_2, CO or CH_4. The ratio determining the split between these three gases depends on whether the pool is assumed to combust in a flaming or smouldering fire. Dead tree and shrub, sedge foliage, and dead fine woody debris pools are assumed to burn in the flaming combustion phase and emit at a ratio of CO:CO_2 by mass (t ha^{-1}) of 0.0499 and CH_4:CO_2 by mass (t ha^{-1}) of 0.00194 (Ottmar 2014). The rest of the dead and live pools are assumed to smoulder and emit at a ratio of CO:CO_2 by mass (t ha^{-1}) at 0.254, and the CH_4:CO_2 by mass (t ha^{-1}) at 0.038 (Kohlenburg et al. 2018). These emission ratios were converted to molar ratios of C in a series of calculations such that the total amount of C emitted is multiplied by a factor of 0.8788, 0.1083, 0.01285 to solve for the amount of C emitted as CO_2, CO and CH_4, respectively for flaming, and 0.523, 0.327, and 0.149 to solve for the amount of C emitted as CO_2, CO and CH_4, respectively for smouldering.

Table 4
Data used to approximate the proportion of the dead peat pools consumed by fire in the Canadian Model for Peatlands (CaMP).

	Ecosystem Treed bog	Shrub Fen
Reference:	Lukenbach et al. 2015	Turetsky et al. 2011
Drought Code at time of fire	290	410
Water table (cm)	−36	−24
Oxic C load (kg C m^{-2})	12.1	17.5
C Consumed	1.52	2
% C consumed of oxic layer[a]	0.125	0.114

[a] proportion of C consumed was calculated for the oxic acrotelm and catotelm layers only, as it is assumed that none of the anoxic layer is consumed. Values highlighted in grey were those used in the CaMP for all dead peat pools and the acrotelm. Data for the treed bog ecosystem are used for the following CaMP peatland categories: open bog, treed bog, forested bog, and open rich fen. Data for the shrub fen ecosystem are used for the following CaMP peatland categories: open poor fen, forested poor fen, treed rich fen, and forested rich fen.

6. Calibration of peat decay parameters

Decomposition rates for the acrotelm and catotelm pools are rarely measured. Because these data are extremely limited, the exponential decay rates (y^{-1}) and Q_{10} values for these pools were calibrated (Table 5) using peat profile C stock data. Parameters were calibrated using a Bayesian Markov Chain Monte Carlo technique that was previously used to calibrate decay parameters for the CBM-CFS3 (Hararuk et al., 2017). To calibrate parameters for the CaMP acrotelm and catotelm, observed and modeled acrotelm and catotelm C stocks were compared across several thousands of iterations to generate a posterior distribution for each parameter value. Prior parameter distributions were assumed to be uniform and bound by minimum and maximum values obtained from the literature (Bona et al., 2018), and calibration runs were continued until posterior parameter distributions became stable (Hararuk et al., 2017).

We recognize that since the CaMP is built for the purpose of generating estimates of C emissions and not C stocks, that calibrating against C stock changes instead of one-time C stock pool data would be ideal. However, because flux data are not split between acrotelm and catotelm peat pools they could not be used for calibration, and were instead reserved for model validation. The purpose of this calibration procedure is to parameterize the decomposition rates of the long-term acrotelm and catotelm pools, therefore the calibration procedure was conducted using the initialization spin-up of the CaMP (see Section 4 Model initialization), where long-term water table is used instead of annual water table fluctuations. Over the long timeframe required to calibrate decomposition of peat, only large-scale temporal variation in parameters such as water table is required.

The calibration dataset was comprised mainly of 370 peatland sites from Zoltai et al. (2000) where sufficient data were available to estimate acrotelm and catotelm C stocks, and enough vegetation and wetland metadata were available to classify the sites into the appropriate CaMP peatland category. Because the data of Zoltai et al. (2000) are mainly from western Canada additional data from eastern Canada (Quebec and Ontario), originating from various peatland researchers' unpublished data, were included to fully represent peatlands across Canada (Garneau unpublished data; Packalen unpublished data; Tarnocai unpublished data).

Table 5
Calibrated peat decay parameters.

Calibration grouping	k_b [a] Acrotelm Oxic	Catotelm Anoxic	Q_{10} Acrotelm Oxic	Catotelm Anoxic	Pt
Calibration group 1: Open bog Poor fen	0.0283	0.000890	4.25	1.21	0.42
Calibration Group 2: Treed bog Forested bog Treed poor fen Forested rich fen	0.0401	0.0003007	3.99	2.11	0.18
Calibration Group 3: Open rich fen Treed rich fen Forested rich fen	0.0861	0.001235	4.05	1.54	0.39

[a] k_b is the base decay rate used along with mean annual temperature and the Q_{10} value to calculate the applied decomposition rate (Eq. (6)). Pt is the proportion of decayed C transferred to the downstream pool, where as 1-Pt is the proportion of decayed C transferred to the atmosphere. During calibration water table depth remains at a constant long-term value, therefore only decay parameters for the acrotelm oxic and catotelm anoxic pools (shown here) are calibrated. The acrotelm anoxic and catotelm oxic k_b values were solved for by linear interpolation between the calibrated values. The Q_{10} for the acrotelm oxic pool was used for the acrotelm anoxic pool, and of the catotelm anoxic pool for the catotelm oxic pool. Pt values are held constant for all peat pools within the same calibration group.

7. Model sensitivity and behavior

Model sensitivity and behavior were assessed by examining the responses of emission (CO_2, CH_4) estimates, and initial peatland C stock estimates, to spatial input variables that are either directly (MAT) or indirectly (Fire return Interval, FRI; DC) related to climate. Carbon dioxide emission rates are affected through two separate mechanisms in the model; one that helps to determine the size of the initial peat C stock resulting from the model initialization (affected by MAT and FRI), the other determines the annual proportion of the peat C stock that is decomposed and then either released to the atmosphere or transferred to another pool (affected by MAT (see Section 3.6 Decay Dynamics) and by the annual water table depth. Methane emissions are primarily driven by water table depth that is predicted by DC.

A theoretical spatial test grid was developed to represent the full range of values, where peatlands occur in Canada, for each spatial variable; Fire return intervals ranged between 35 and 500 years, MATs between −10.1 °C and 7.8 °C, and the long-term and annual DCs between 78.5 and 411.8 (corresponding to a different range of predicted water table depths for each peatland category; Appendix Table A14). Readers are reminded that the CaMP (v2.0) does not include frozen soils typical of regions with MAT of −10.1, and that this sensitivity test is being used to understand the limits of the current model's response to the full range of extremes in climatic variables observed in Canada. When a single variable was tested, all other spatial variables were held constant at their mid-range value. We used NEE (net ecosystem exchange of CO_2) to assess CO_2 emissions estimates for ease of comparison to observed flux measures that are commonly reported in peatland C dynamics studies. We also assessed the 100-year global warming potential (GWP) of total heterotrophic respiration (T_Rh), calculated as the sum of CO_2 and CH_4 emissions after conversion to CO_2 equivalents (CO_2e), using a conversion factor of 1 for CO_2 and 25 for CH_4 (Myhre et al., 2013).

7.1. Sensitivity of initial c stocks to MAT and FRI

The MAT is applied to the base annual decomposition rates of each of the dead pools using a Q_{10} relationship (see Section 3.6 Decay Dynamics), which will directly influence the amount of C emitted to the atmosphere annually. The same approach is used to arrive at an applied decay rate to use in the model initialization (i.e., spin-up procedure). In the CaMP MAT can have a large effect on the amount of initial C in each of the peat pools within a peatland type (Table 6). This in turn can affect emissions estimates, as pools with larger stocks will tend to have larger emissions. The absolute differences in C stocks between the highest (7.8 °C) and lowest (−10.1 °C) MAT tested across peatland types ranged from 240 to 1641 Mg C ha^{-1} for the acrotelm and from 163 to 1677 Mg C ha^{-1} for the catotelm (Table 6). In general, peat pool C stocks were highest at the lowest MAT, and lowest and the highest MAT. This relationship is consistent with that of another large-scale (global) peatland soil organic C model (Wania et al. 2009) that uses a similar initialization procedure. Wania et al. (2009) report a predicted range of total soil organic C stocks from approximately 400 Mg ha^{-1} (at MAT = 8.7 °C) to 2000 Mg ha^{-1} (at MAT = −6.7 °C) when testing a MAT range of 8.7 °C to −6.7 °C. CaMP predicts a similar but higher range in total peat C when the same range in MAT is tested; 520 Mg ha^{-1} at MAT = 8.7 °C, to 3365.2 Mg ha^{-1} at MAT = −6.7 °C. Wania et al. (2009) caution that their estimates tend to be lower than those found in the literature. They attribute the low values to limiting the analytical solution for their spin-up to 1000 years, whereas we assume that the catotelm never reaches a true equilibrium and use a factor of 8000 years in the initialization of the catotelm pool (see Section 4 Model Initialization). While our total C stocks reach higher values than reported by Wania et al. (2009) they are within the range of field measurements reported in the literature that were compiled here for model calibration, which ranged from 385 to 4300 Mg ha^{-1} (Zoltai et al., 2000; Garneau unpublished data; Tarnocai unpublished data; Packalen unpublished data). In summary, the CaMP is demonstrating expected model behavior by having a consistent correlation between organic matter accumulation and MAT, that is in general agreement with others (Wania et al. 2009; Clymo et al. 1998). Other relevant studies, which examined peatland sensitivity to temperature, generally focused on peat (soil) temperature rather than air temperature, demonstrating a strong correlation between CO_2 (LeRoy et al. 2017), and CH_4 (Hargreaves and Fowler 1998, LeRoy et al. 2017) fluxes to peat temperature. This relationship has been shown to decrease with peat profile depth (Jusczak et al. 2013) which is consistent with our calibrated Q_{10} values which were uniformly smaller for catotelm (deeper) pools than for acrotelm pools (Table 5).

The model's initial C stocks for peat pools also demonstrate an expected relationship with FRI; stocks are decreasing with increased fire frequency and visa versa. Keeping all other spatial variables constant at mid-range values (MAT = −1.15, annual and long-term DC = 300.7), initial total peat pool C stock shifts from a mean of 1500 Mg C ha^{-1} (at FRI = 35) to 1872 Mg C ha^{-1} (at FRI = 500). The acrotelm C is least sensitive to changes in FRI shifting from 180 Mg C ha^{-1} (at FRI = 35) to 185 Mg C ha^{-1} (at FRI = 500), whereas the catotelm C has a larger

Table 6
Sensitivity of a) modelled net ecosystem exchange (NEE) and total heterotrophic respiration (T_Rh), and b) initialized oxic acrotelm and anoxic catotelm peat C pools to national potential range in mean annual temperature (MAT) by peatland category. Five values for MAT were tested (3 shown) that are representative of the range in values for the area where peatlands occur in Canada.

Peatland categories		MAT at min. −10.1 °C	MAT at median −1.15 °C	MAT at max. 7.8 °C	Absolute Difference (max-min)	MAT at min. −10.1 °C	MAT at median −1.15 °C	MAT at max. 7.8 °C	Absolute Difference (max-min)
	(a)	NEE g C−CO_2 $m^{-2}y^{-1}$				T_Rh g CO_2e $m^{-2}y^{-1}$			
Open bog		−58	−47	−31	27	2449	2492	2550	100
Treed bog		−4	11	17	21	1265	1320	1343	78
Forested bog		−29	−7	7	36	2274	2357	2409	135
Open poor fen		11	38	81	71	1912	2015	2174	262
Treed poor fen		40	40	41	2	1657	1659	1664	6
Forested poor fen		−57	−4	20	76	2429	2623	2712	283
Open rich fen		42	42	67	25	2302	2304	2396	94
Treed rich fen		75	77	82	7	2617	2624	2644	27
Forested rich fen		88	160	167	80	3394	3661	3689	295
	(b)	Acrotelm Peat Mg C ha^{-1}				Catotelm Peat Mg C ha^{-1}			
Open bog		897	275	80	817	1109	946	980	163
Treed bog		271	89	26	245	753	707	520	233
Forested bog		448	143	43	405	1304	1206	869	434
Open poor fen		1712	338	71	1641	2309	1933	1585	724
Treed poor fen		436	118	32	404	1441	1289	908	533
Forested poor fen		288	144	48	240	1399	1265	904	496
Open rich fen		551	134	39	512	3116	2132	1439	1677
Treed rich fen		606	166	48	558	3121	2283	1584	1537
Forested rich fen		615	236	71	543	2658	2032	1475	1183

Note: All spatial model parameters other than MAT were set at mid-range values and held constant. Long-term and annual Drought Code is constant here at 300.7 predicting the following water table depths for each peatland type (Open bog WTD = −26 cm; Treed bog WTD = −39.4 cm; Forested bog = −42.7 cm; Open poor fen = −12.8 cm; Treed poor fen = −26.2 cm; Forested poor fen = −29.5 cm; open rich fen = −7.9 cm; treed rich fen = −21.3 cm; forested rich fen = −24.6 cm).

shift from 1320 (at FRI = 35) to 1687 (at FRI = 500). Because the oxic acrotelm is partially consumed by fire, whereas the anoxic catotelm is untouched, one might assume that the acrotelm should be more sensitive to FRI. However, that is only the case for shorter-term contemporary or forward looking simulations. Here we are examining the result from the 8000 year spin-up of the model initialization. The FRI has a larger impact on the catotelm pool than on the acrotelm pool during initialization because the rate at which the acrotelm transfers C to the catotelm is proportionally larger when the acrotelm pool is larger, and because this transfer rate has significant weight in determining the C stock of the catotelm, given that the decay rate within the anoxic catotelm is almost zero (Table 5). On average the rich fen total initial peat pool C tended to be least sensitive to shifts in FRI (14% difference between FRI 35 and 500), followed by poor fens (21% difference between FRI 35 and 500), and then bogs being the most sensitive (33% difference between FRI 35 and 500).

7.2. Sensitivity of NEE and total Rh to MAT and water table depth

Overall, the model's simulation of NEE is more sensitive to differences in MAT (Table 6) than it is to changes in annual water table position (Table 7), whereas the opposite is true for total heterotrophic respiration (T_Rh). This result is expected when the model is behaving correctly, since CO_2 emissions are highly dependent on the applied annual decay rates defined by a Q_{10} relationship with MAT, whereas T_Rh also included CH_4 emissions that are generally higher than CO_2 emissions (once converted to CO_2e), and are dependent on annual water table position.

A comparison of the relative shift in T_Rh and NEE (Table 7) in response to the range of annual DCs (corresponding to different typical ranges of water tables for each peatland category, Table A14) shows that CH_4 emission is the driving force of the sensitivity of T_Rh to water table movement. This is especially evident once the global warming potential of CH_4 is taken into account and CH_4 emissions are converted to CO_2e (Fig. 3). The bog sites shown here reach their maximum CH_4 emission potential since the optimal CH_4 flux water table depth for bogs are reached (−26 cm) within the full range of DCs tested, whereas the optimal depths for poor fen (−1.6 cm) and rich fen (16.3 cm) sites are not (See Section 9 for further discussion).

In order to assess model behavior in response to changes in DC and the resulting shifts in water table depth, a set of seven-year simulations with different combinations of long-term and annual water table depths was tested. We present two examples (Fig. 4, open bog; Fig. 5, open rich fen) from these scenarios, where the annual water table starts above the long-term average water table, dips below the long-term average (typical of a drought or peatland drainage scenario) and then climbs back up above the long-term water table depth (Figs. 4A and 5A). In both examples (Figs. 4B and 5B), C is transferred into the acrotelm anoxic holding pool when the annual water table is above the long-term water table position, and C is also transferred into the catotelm oxic holding pool when water table dips below the long-term water table position. This dip results in a change in decay dynamics (and CO_2 emissions from Rh), however it is a small contributor to the change in T_Rh that is dominated by the change in CH_4 flux (Figs. 4C and 5C), especially after the global warming potential of CH_4 is taken into account (Figs. 4D and 5D). Again, the maximum CH_4 potential flux rate (F_{CH4max}) employed here is reached in the bog site at the optimal water table of −26 cm, which explains the symmetrical peaks in CH_4 at that water table depth (Fig. 4C) for the bog site, which is not evident in the rich fen site (Fig. 5C).

The trends in these results are consistent with our understanding of CH_4 and NEE dynamics in peatlands, and with the literature. For instance, Updegraff et al. (2001) demonstrated that CO_2 was insensitive to water table movement and only responded significantly to soil temperature. Methane, however, is shown to be highly sensitive to water table movement but is also impacted by a host of other variables such as soil temperature, plant productivity and N cycling dynamics (Updegraff et al., 2001) that we do not currently account for in this version of CaMP.

It is clear that emissions from some peatland types are more sensitivite to changes in MAT (such as forested bogs and forested poor fens) than others (such as treed poor fens and treed rich fens) (Table 6). The reason for these differences are difficult to isolate because they do not depend solely on MAT, but other parameters including annual applied decay rate, the amount of initial C in each peat pool, input rates from upstream pools, NPP, and turnover rates for each vegetation layer that

Table 7
Sensitivity of modelled net ecosystem exchange (NEE), and total heterotrophic respiration (T_Rh) to annual drought code (DC) by peatland category. Four values for annual DC were tested (2 shown) that are representative of the range in values for the area where peatlands occur in Canada.

Peatland categories	Annual DC at min. 78.5	Annual DC at max. 411.8	Absolute Difference (max–min)	Annual DC at min. 78.5	Annual DC at max. at 411.8	Absolute Difference (max–min)
	NEE g $C{-}CO_2\ m^{-2}y^{-1}$			T_Rh g $CO_2e\ m^{-2}\ y^{-1}$		
Open bog	−13	−24	−11	1449	1773	324
Treed bog	−14	17	31	2028	1137	891
Forested bog	−26	−3	22	2862	2226	636
Open poor fen	−13	50	62	3412	1653	1759
Treed poor fen	27	43	16	2014	1566	448
Forested poor fen	−13	−2	11	2877	2558	319
Open rich fen	26	46	19	2736	2191	546
Treed rich fen	72	77	5	2732	2594	139
Forested rich fen	153	150	−3	3713	3604	108

Note: All spatial model parameters other than annual DC were set at mid-range values and held constant. Long-term drought code is held constant at 189.7. Note that DC corresponds to different water table depths depending on the peatland type (see Section 3.2 Water Table Depth).

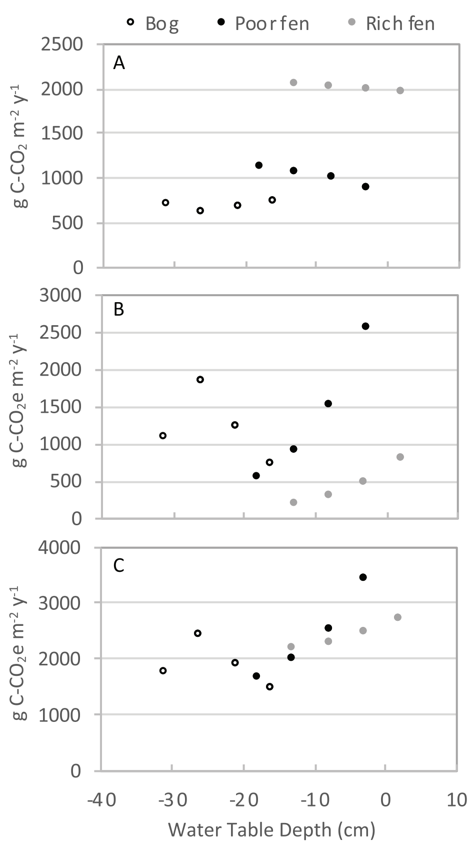

Fig. 3. Sensitivity of modelled CO_2 emissions to changes in annual water table depth using a theoretical example of an open bog, poor fen and rich fen in the Boreal Plains ecozone A) Modelled carbon dioxide emissions. B) Modelled methane emissions in carbon dioxide equivalents. C) Total heterotrophic respiration in carbon dioxide equivalents. Other spatial variables were held constant at their mid-range values: Fire return interval at 267 years, mean annual temperature at −1.15 °C, and historical drought code at 189 corresponding to a long-term water table depth of −21 cm for open bog, −7 cm for poor fen and −2 cm for rich fen. A full range of annual drought codes was tested in this simulation: 78.5, 189.9, 300.7, and 411.9. These correspond to different annual water table depths, depending on the peatland type (Table A14).

vary by peatland category. In general, peatland categories that are more sensitive to changes in MAT have higher C inputs, and larger initial C pools when decay rates and Q_{10}s are similar between peatland categories (Table 5).

8. Model evaluation

The CaMP (v2.0) is the first version of a comprehensive framework for modeling peatland C emissions and removals for Canada. The initial challenge in developing the model was to be able to represent the range of peatland types in Canada and to compile existing data and synthesize the best available knowledge to develop the basic model. The main goal of this evaluation is to assess how much variation is explained with the CaMPs base model structure, to consider the distribution and overall mean model residuals for large-scale estimation, and to identify and prioritize issues needing improvement.

This evaluation of the CaMP (v2.0) compared estimates from the model to those arrived at using data from peatland sample sites in 16 different locations across Canada with GHG flux measurements for multiple years or seasons. Eleven of the sites had data for CH_4 emission rates, and 12 of the sites had data for NEE for CO_2 either from eddy covariance flux towers or chamber measurements. Measured data were reported as daily or seasonal rates so we converted them to annual rates for comparison to annual rates predicted by the model, assuming winter emissions equaled 15% of growing season emissions (Tier 1 emission factors in the IPCC wetland Supplement) (See Webster et al., 2018 for more details on conversions). Peatland sample sites were mainly located in the provinces of Quebec, Ontario, Alberta and Manitoba, with one from Nova Scotia, and were located across the Mixedwood Plains, Hudson Plains, Boreal Shield, Boreal Plains and Taiga Shield ecozones (Ecological Land Classification Group [ELCG], 2005). Each site was classified according to a CaMP peatland category to the best of our ability with the information that was available. For NEE comparisons sites were mainly open bog ($n = 3$) and open poor fen ($n = 5$) sites, along with two open rich fen sites, and one of each treed bog and treed rich fen. None of the sites with NEE data were classified as forested based on the available information. For the CH_4 comparisons sites were mainly open bogs ($n = 4$), open rich fens ($n = 3$) or open poor fens ($n = 3$), and one site of each treed bog, forested bog and treed poor fen.

8.1. Overall model fit

Overall, mean model residuals (predicted-observed) for CH_4 and NEE are reasonably small (6 g $C{-}CH_4\ m^{-2}\ y^{-1}$, and 20 g $C{-}CO_2\ m^{-2}\ y^{-1}$, respectively), suggesting that on average the model is well calibrated for large-scale applications. A comparison of the distribution of observed and predicted CH_4 annual emissions shows that the CaMP is able to capture the full range of variation in CH_4 emissions (Observed: Sd = 15, range from 2 to 58 g $C{-}CH_4\ m^{-2}\ y^{-1}$ vs. Predicted: Sd = 21, range from 2 to 73 g $C{-}CH_4\ m^{-2}\ y^{-1}$), and that observed and predicted measures of central dispersion compare well (Observed mean = 15 and median = 9 vs. Predicted mean = 21 and median = 11) (Fig. 6A). This result suggests that the symmetrical

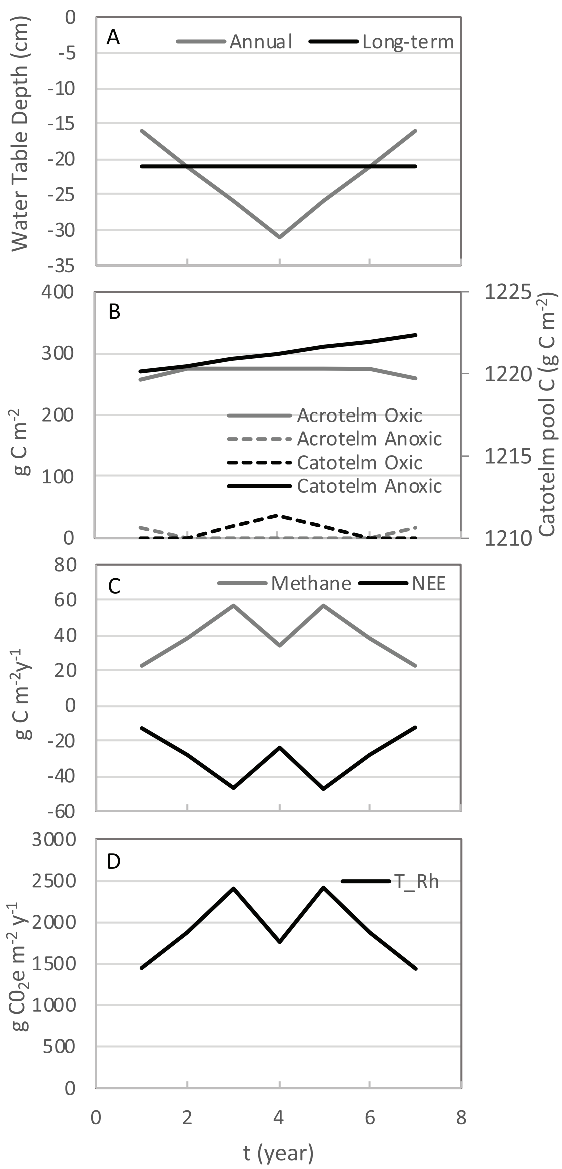

Fig. 4. A 7-year theoretical simulation demonstrating model behaviour in response to changes in annual water table for an open bog example. A) Annual and long-term water table position relative to the peat surface. B) The amount of C in each peat pool as C that is transferred as a result of annual water table movement relative to the long-term water table level. C) the response of annual methane and net ecosystem exchange (NEE). D) the response of total heterotrophic respiration (CO_2 + CH_4) in CO_2 equivalents.

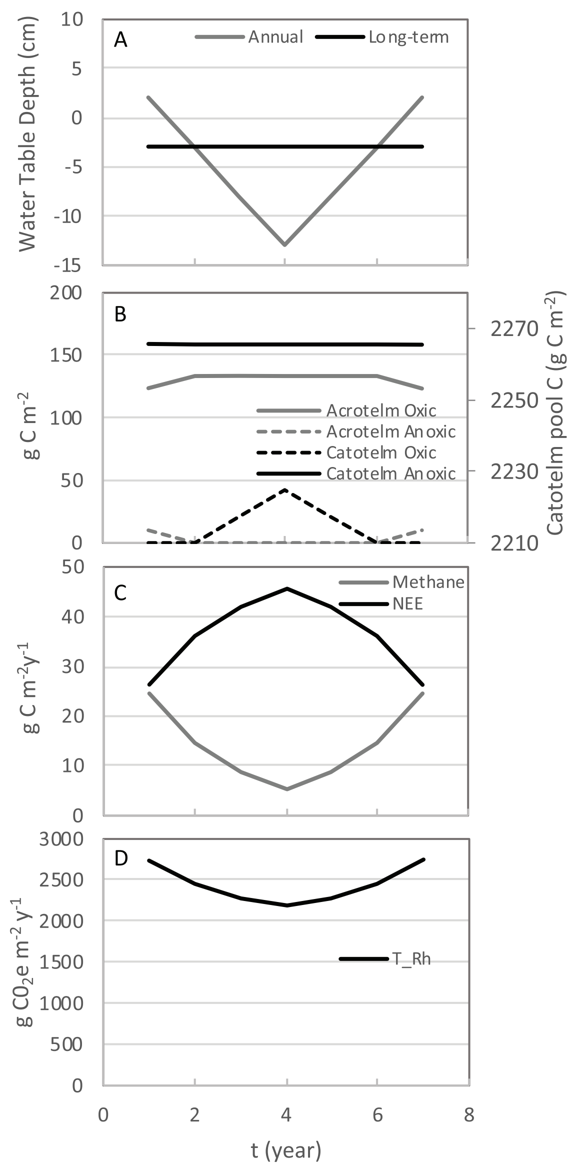

Fig. 5. A 7-year theoretical simulation demonstrating model behaviour in response to changes in annual water table for an open rich fen example. A) Annual and long-term water table position relative to the peat surface. B) The amount of C in each peat pool as C that is transferred as a result of annual water table movement relative to the long-term water table level. C) the response of annual methane and net ecosystem exchange (NEE). D) the response of total heterotrophic respiration (CO_2 + CH_4) in CO_2 equivalents.

scheme used to model CH_4 emissions, where mean maximum CH_4 emission values (F_{CH4max}) are associated with optimal CH_4 water table depths (see Section 3.7 Methane Emission) is well calibrated and performs well on average for large-scale analyses, especially considering the high complexity of CH_4 dynamics that complicates both prediction (see Section 3.7 Methane Emission) and measurement of CH_4 emissions (see Section 9 for further discussion). A comparison of the distribution of observed and predicted annual NEE did not agree as well as for CH_4. Overall, the CaMP overestimates NEE (Observed: mean = −39 g C−CO_2 m^{-2} y^{-1} and median = −58 g C−CO_2 m^{-2} y^{-1} vs. Predicted mean = −19 g C−CO_2 m^{-2} y^{-1} and median = −21 g C−CO_2 m^{-2} y^{-1}), indicating that the CaMP needs improvement in expressing the full variation in NEE (observed values have a much wider range and higher standard deviation (range from −145 to −61, $S = 67$) compared to predicted values (range from 21 to −61, $S = 21$) (Fig. 6B)). This result suggests that, in future versions of the model, efforts must be made either to increase the variation expressed in NPP and turnover rates for different regions and peatland categories, or increase the variation in annual applied decay rates, or most likely both. In the current version of the CaMP (v2.0), NPP is modeled as a constant for each peatland category, however we know that NPP is not static (Vitt et al., 2003; Thormann et al., 1997; Upgaff et al. 2001). Furthermore, we have already identified in the model behavior and sensitivity analysis that a simple Q_{10} relationship with MAT is likely insufficient to capture a realistic response in emissions, and that a response to soil temperature rather than air temperature would likely lead to better results (Upgaff et al. 2001; Leroy et al., 2017) (see Section 9 for further discussion).

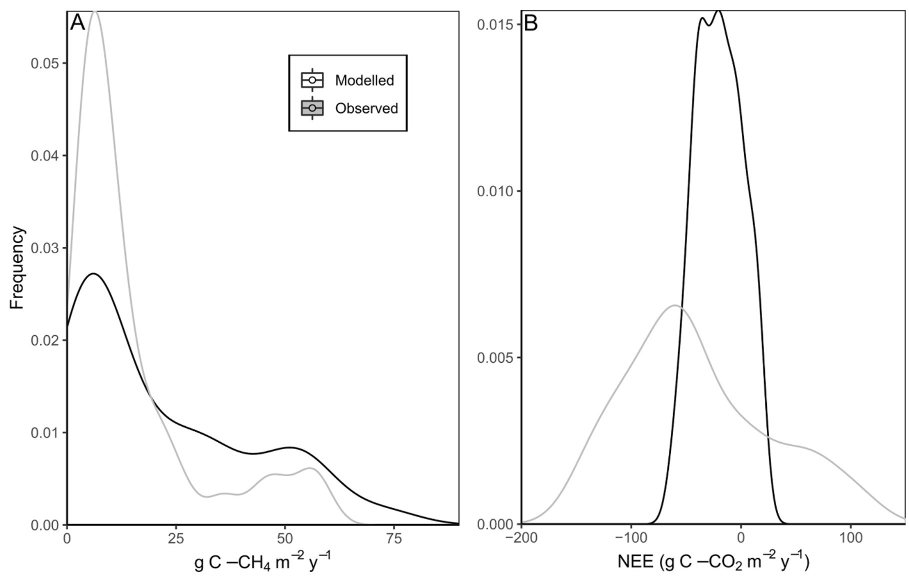

Fig. 6. Comparison of distribution of modelled and observed values for (A) methane fluxes and (B) net ecosystem exchange (NEE) for CO_2 fluxes.

The annualized measured and predicted emission rates for both CH_4 and NEE were not significantly correlated, despite the relatively decent fit of the distribution of observed and predicted values for CH_4 and low overall model residuals for both CH_4 and NEE. We attribute this to several different factors, such as the lack of soil temperature in the model, the need to increase variation in NPP rates (as discussed above), lack of confidence in scaled up measured CH_4 emissions, insufficient data for model calibration and parameterization for certain peatland categories, that will be discussed further in Section 9. Future Steps Towards Model Improvement.

8.2. Model fit by peatland category

The CaMP was designed to model C dynamics for each of the different peatland categories that occur across Canada. This will enable the model to take full advantage of existing data and knowledge and to easily integrate new peatland science as it becomes available. Because some peatland categories are better studied than others, some categories are better calibrated and parameterized in the CaMP. To evaluate the model's ability to estimate CH_4 emissions and NEE for different peatland categories, mean model residuals were split between peatland categories (Figs. 7 and 8). Even though the sample sizes for each peatland type are very low, given the high temporal and spatial variation within measured data available (Tables 8 and 9) the analysis can be helpful to inform future work, especially in identifying data gaps.

The analysis demonstrates that the model is overestimating CH_4 in both open bogs and open poor fens, and possibly treed bogs (small sample) (Fig. 7). The opposite was found for open rich fens, and possibly forested bogs (small sample size) (Fig. 7). The underestimation of CH_4 emissions for rich fens is likely due to issues arising from the use of F_{CH4max} and its corresponding optimal water table depth (2 cm) taken from Turetsky et al. (2014) that was based on a daily time-scale. When this relationship is applied in our annual time-scale model the optimal water table corresponding to F_{CH4max} is never reached (see Section 3.7 Methane Emission). Ideally, a new dataset with annual water table depths and annual F_{CH4max} values should be developed to improve CaMP predictions. Underestimation of CH_4 emissions for open bogs and open poor fens, likely results from the need to adjust the F_{10} response to changes in water table that could be implemented using data from peatland drainage studies in open bogs and poor fens. Also, the open bogs and poor fens in this analysis are mainly located in the relatively small (3.47×10^6 km^2) Mixedwood Plains ecozone (nine out of twelve years of data for open bogs, and four out of eight years of data for poor

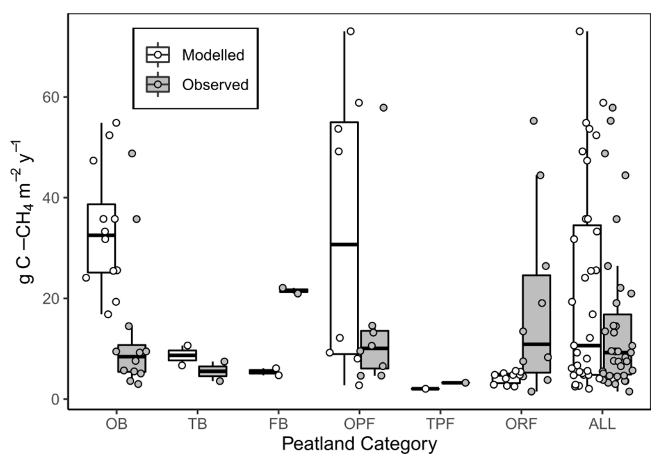

Fig. 7. Boxplot of observed and CaMP modeled annual CH_4 emission rates by peatland category. OB, open bog; TB, treed bog; FB, forested bog; OPF, open poor fen; TPF, treed poor fen; ORF, open rich fen; ALL, all validation sites.

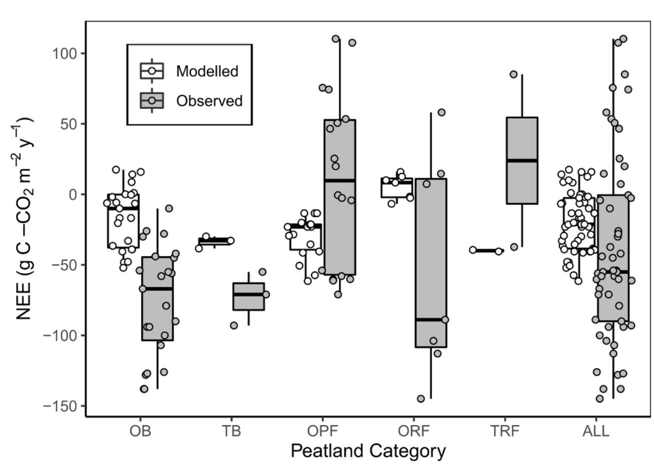

Fig. 8. Boxplot of observed and CaMP modelled annual net ecosystem exchange (NEE) CO_2 emission rates by peatland category. OB, open bog; TB, treed bog; OPF, open poor fen; ORF, open rich fen; TRF, treed rich fen; ALL, all validation sites.

Table 8
Comparison between observed and modelled mean methane flux by study location.

Reference	Location[a]	n	Peatland Category[b]	Mean Methane Flux (standard deviation) g $C-CH_4$ m^{-2} y^{-1} Observed	Modelled	Residual
(a) Comparison of means by site location (intra-site temporal variation)						
Rouse et al. 2002	Churchill, Manitoba	5	Open rich fen	5 (2.2)	4 (1.0)	−0.7 (2.2)
Bubier et al. 1995, 2005	Thompson, Manitoba	5	Open to treed poor fen, and open rich fen	12 (5.9)	5 (3.4)	−7 (6.2)
Moore et al. 2001 and Humphreys et al. unpublished data	Mer Bleue, Quebec	9	Open bog	15 (16)	32 (14)	18 (17)
Strack and Waddington 2007	St. Charles-de-Bellechasse, Quebec	4	Open poor fen	22 (24)	37 (34)	15 (34.5)
(b) Total mean of all site locations (inter-site spatial variation)						
12 study sites	12 locations across Canada	12		14 (11)	22 (18)	8 (24)

[a] Only individual study locations with more than three years of data shown here.
[b] Peatland sites classified into the CaMP peatland categories using the best available data.

fens) in southern Canada, which is not representative of the spatial extent of open bogs and poor fen systems in the majority of Canada that are mainly located in northern continental (Boreal) ecozones (Webster et al., 2018). Only 0.3% of Canada's bog area and 0.003% of Canada's poor fen area are found in the Mixedwood Plains ecozone (Webster et al., 2018).

Model residuals by peatland category for NEE show that the CaMP overestimates NEE for open rich fens and open bogs, and possibly treed bogs (small sample) (Fig. 8) and underestimates it for open poor fens and possibly treed rich fens (small sample) (Fig. 8). It is difficult to assess whether these biases are related to unexpressed variation in NPP or decay rates, but is likely a combination of both where the relative importance of NPP or decay differ between peatland categories. Because the data for model parameterization of NPP rates for mosses, shrubs and sedges was more comprehensive for open bog sites than the other peatland types, it is safe to assume that for open bog systems the quality and accuracy of NPP rates are relatively high, so it is likely that it is the decay rates that need improvement, and probably need to be reduced. The reason that open bog decay rates could have turned out too high is that in the calibration process, they had to be combined with open poor fen sites to create a large enough sample size (Table 5) to calibrate decay rates. It is likely that the grouping also caused a similar, but opposite bias for poor fens, meaning that the open poor fen decay rate likely needs to be higher, which would improve the current underestimation of NEE in open poor fen systems. In general, data to parameterize NPP rates for vegetation layers in both poor fen and rich fen systems were limited (Bona et al., 2018). This data gap needs to be addressed. Lastly, introducing soil temperature to the decay function (rather than the current relationship with air temperature, MAT) would likely improve these results, but since we did not have access to measured soil temperature in our observed database, we could not test this hypothesis.

8.3. Site-level model fit

We have shown that, on average, means of observed and predicted emission rates are in agreement, and for national-scale estimation this is adequate. However, understanding finer-scale variation will lead to overall improved accuracy in estimates, and allow for future application of the model at finer scales. For this reason we performed an individual site-level evaluation of model fit. Temporal and spatial variation among observed values at fine scales (Tables 8 and 9) is very high. We do not expect this version of the CaMP to be able to express this high degree of variation, but a comparison will inform future research for development of the CaMP by showing where it succeeds or fails at this fine spatial scale.

First, we examined spatial (inter-site level) accuracy. To reduce the effect of inter-annual temporal variation, only site locations with more than three years of data were used, data were averaged over time, and compared to averaged model predictions for the same years (Tables 8 and 9). Methane was relatively well estimated by the model when averaged over a 4 to 9 year period for four different sites (residuals ranging from −0.7 to 18 g $C-CH_4$ m^{-2} y^{-1}), and the variation expressed was similar (compare Sd values in Table 8). It is also notable that the more northerly sites, that are more representative of ecozones in Canada that have large areas of peatlands, are better predicted by the model. For example, the Churchill site occurs in the Hudson Plains ecozone where peatlands make up an estimated 134.1×10^3 km^2 (Webster et al., 2018), and Thompson, Manitoba site occurs in the Boreal Shield ecozone where peatlands make up an estimated

Table 9
Comparison between observed and modelled mean net ecosystem exchange (NEE) by study location.

Reference	Location[a]	n	CaMP peatland category[b]	mean NEE (standard deviation) g $C-CO_2$ m^{-2} y^{-1} Observed	Modelled	Residual
(a) Comparison of means by site location (intra-site temporal variation)						
Griffis et al. 2000	Churchill, Manitoba	4	Open poor fen	22 (30)	−21 (0.6)	−43 (30)
Adkinson et al. 2011	North-Central Alberta	6	Open poor fen and rich fen	−43 (45)	−23 (22)	20 (43)
Humphries et al. unpublished	Victor MOE, Ontario	6	Open poor fen and treed bog	−67 (15)	−32 (3.8)	34 (14)
Strachen et al. 2016	Lac la Caron, Quebec	5	Open poor fen	−76 (40)	−46 (5.5)	30 (42)
Humphries et al. unpublished	Mer Bleue, Ontario	16	Open bog	−78 (42)	−4.8 (16)	74 (33)
Strack et al. 2006	St. Charles- de- Bellechasse, Quebec	6	Open poor fen	47 (40)	−18 (5.3)	−65 (40)
(b) Total mean of all site locations (inter-site spatial variation)						
11 study sites	11 locations across Canada	11		−29 (58)	−27 (19)	2.0 (68)

[a] Only individual study locations with more than three years of data shown here.
[b] Peatland sites classified into the CaMP peatland categories using the best available data.

275.6×10^3 km^2 (Webster et al., 2018) were better simulated by the model (mean residuals between -0.7 and -7 g $C{-}CH_4$ m^{-2} y^{-1}), as compared to the two southern sites (Mer Bleue and St. Charles-de-Bellechasse) (mean model residuals between 15 and 18 g $C{-}CH_4$ m^{-2} y^{-1}) in the Quebec Mixedwood Plains ecozone where peatlands areas are relatively small and occupy only 1.045×10^3 km^2 (Webster et al., 2018), making them much less significant at the national-scale. Overall, the total average model residual for CH_4 is low whether averaged by site location (8 g $C{-}CH_4$ m^{-2} y^{-1}) or not (6 g $C{-}CH_4$ m^{-2} y^{-1}), which suggests that spatial and temporal variation of CH_4 is of similar magnitude. This is supported by similar average observed intra-site temporal (Sd ranging from 2.2 to 24 g $C{-}CH_4$ m^{-2} y^{-1}) and inter-site spatial(Sd = 11 g $C{-}CH_4$ m^{-2} y^{-1}) variation (Table 8).

The site-level analysis for CO_2 emissions showed patterns for over and under-estimation that are similar to those for CH_4 emissions. However, interestingly, when the sites are averaged by site location the overall mean model residual is reduced to 2.0 g $C{-}CO_2$ m^{-2} y^{-1} from the previous mean (when rates were not weighted by location) of 20 g $C{-}CO_2$ m^{-2} y^{-1}. This result demonstrates that spatial variation across different sites is higher than temporal variation for CO_2 emissions, which is confirmed by the observed inter-site variation of (Sd = 58 g $C{-}CO_2$ m^{-2} y^{-1}) which tends to be higher than the observed intra-site temporal variation (range of 15–45 g $C{-}CO_2$ m^{-2} y^{-1}). This further supports the idea that including mean annual soil temperature in the peat decay function, a variable that generally correlates with a north to south gradient, provides a good opportunity to improve explained variation in NEE emissions. Higher spatial variation in observed NEE rates also supports the idea that NPP rates for different regions (whereas most NPP rates do not currently vary by region due to lack of data available) provides another opportunity for model improvement.

9. Future steps towards model improvement

The CaMP (v2.0) was developed as a first step towards a dynamic C estimation framework for peatlands, for application at the national-scale in Canada. The challenge was to build a simple model framework applicable to diverse peatland types and regions occurring across Canada by synthesizing expert knowledge and compiling the best available existing data for model parameterization and calibration. The results of the model evaluation demonstrate that although the modeling framework is appropriate for large-scale GHG emission and removal estimates, it will require further work and new data collected at finer scales to provide inputs and refine parameters for the model to enable explanation of fine-scale variation. The guidance of the IPCC states that uncertainties have to be identified, quantified and reduced as far as is practicable (IPCC 2003). Here we summarize and discuss areas within CaMP (v2.0) that have been identified as needing improvement, as well as data requirements and gaps, to help inform development of future versions of the CaMP (Table 10).

Although the overall mean model residual of NEE was reasonably low for large-scale estimation, it was evident that the model requires work to express the full variation in observed NEE. Since NEE is the balance of NPP and decomposition, there are several approaches that can be taken to improve the explained variation in model estimates. First, we recommend the integration of mean annual soil temperature, rather than MAT, into peat decomposition dynamics (Table 10), since CO_2 emissions have been shown to be correlated more strongly with soil temperature than air temperature (Upgaff et al. 2001; Leroy et al., 2017). It is known that air temperature does not have a simple, direct relationship with soil temperature in peatland systems (Weis et al. 2006), because of the thermally insulative properties of peat (Zhao and Bingcheng 2019), and a dependence on snow depth and precipitation (Zhang et al., 2005). Therefore, in order to integrate soil temperature into the CaMP, a Canada-wide national model for soil temperature in peatlands, at an annual time-scale, would need to be developed. One approach could be to use an existing peatland soil temperature model, once scaled to the appropriate level for interaction with CaMP. Model's such as the Northern Ecosystem Soil Temperature model (NEST; Zhang et al., 2003) that rely not only on air temperature as climate data, but also precipitation, wind speed and solar radiation would be appropriate. Since the NEST model focuses on active peat layer depth and permafrost thaw, having a NEST module could also support the integration of permafrost dynamics into the CaMP (Table 10). Second, we recommend increasing the complexity of modelled decay dynamics, and therefore the amount of explained variation in modeled NEE, by improving calibration of peat decomposition rates (Table 10). We are aware that the model calibration method is only as good as the input data. To provide a sufficiently large set of input data for calibration of decomposition rates, peatland categories (and therefore input data used for calibration) were combined into fewer groups because there were not enough data to calibrate each peatland category separately (Table 5). Model evaluation revealed that combining open poor fens and open bogs resulted in an averaged calibrated decay rate that introduced bias, since the evaluation suggested a lower decay rate for open bogs and a higher decay rate for poor fens would improve model estimation of NEE for each of these peatland categories. The Canadian Forest Service peatland profile database that was used for model calibration is still being expanded. Data are continuously being added to the database as new studies emerge and further collaborations with other teams of peatland scientists are formed. However, even with continued efforts to add to the database, it is evident that there is currently a lack of peat core data on poor fen systems. Considering that poor fens are significant peatland type, comprising approximately 20% of the total area covered by peatlands in Canada (Webster et al., 2018), it is likely new data will need to be collected to improve calibration of their peat decay rates.

More variation in modeled NEE may be explained by improving model estimation of NPP rates for different vegetation (e.g., shrubs, mosses, sedges) categories (Bona et al., 2018). The base structure of CaMP (v2.0) uses static NPP rates for combinations of vegetation layer and peatland category where sufficient data were available. We know that NPP is not static and has been shown to vary with pH (Chapin et al., 2006) and nutrient availability (Vitt et al., 2003; Chapin et al., 2006). Graminoids in a lacustrine sedge fen have been shown to increase productivity in response to water table rise, while bog productivity has been shown to respond to changes in temperature (Thormann et al., 1997) and water table depth (Upgaff et al. 2001). Introducing dynamic relationships between NPP and these controlling variables, such as water table depth or temperature, should increase the variation in modeled NEE and improve the model's ability to predict the effects of climatic changes in the future. Because development of these relationships will require time to execute new studies, and to synthesize results from existing and new studies, improvements could be achieved in the short-term by trying to expand and update the database of Bona et al. (2018). In particular, more data are required on sedge productivity, tree productivity, low shrub productivity and especially tall shrub productivity (included only in 28%, 27%, 46% and 14% respectively, of studies in the parameter database), as opposed to more readily reported moss NPP rates (included in 77% of study sites within the database) and attempting to develop separate rates for each peatland category. As previously mentioned for the decomposition rate calibration, there is a lack of poor fen data (comprising 14% of the sites within the parameter database), whereas data on bogs tend to be more comprehensive (27% of sites included in the database).

The results of the model evaluation showed that for large-scale estimation, the CaMP was better at estimating CH_4 emissions than NEE as evidenced by the very small overall mean residuals, and site-level residuals for CH_4. The bell-shaped relationship between maximum CH_4 flux and optimal CH_4 flux water table depths used in CaMP to predict CH_4 flux rates based on water table depth worked well, but requires refinement to explain more variation. The main challenge with CH_4 lies

Table 10
Summary of recommended model improvements.

Model improvement	Proposed changes in future versions of the CaMP	Data requirements	Potential data source
Increased CO_2 emission sensitivity to temperature and potential explained spatial variation in net ecosystem exchange (NEE).	Change Q_{10} relationship from mean annual temperature to mean annual soil temperature for shallow and deep peat layers.	(A) Peat temperature module that can be scaled up to national level.	Northern Ecosystem Soil Temperature (NEST) model Zhang et al., 2003
Increase explained spatial variation in NEE by improving regional variation in NPP rates	Introduce more regional and peatland specific growth rates for different vegetation layers.	(B) Expanded peatland parameter database (Bona et al., 2018). This may require collection of new data.	Data gap
Improved calibrated decay rates	Calibration decay rates for each CaMP peatland category starting with differentiating poor fens from bogs.	(C) Expand peat profile database particularly with additional data on poor fens	Expand on current work[1] Data gap for poor fens
Scale the CaMP methane model for the optimal water table depth to an annual time-step	*Re*-calibrate the relationship between maximum methane flux and optimal water table depths to the annual scale.	(E) Observed annual water table depth and corresponding annual methane flux rates for 50 to 100 peatland sites across Canada.	Data gap
Assess swamp peatland categories	Calibrate and assess swamp categories for use in future.	(C) Expand peat profile database[a] to include swamp data	Data gap
Improve fire matrices by including fire severity	Build different levels of severity into current fire disturbance matrix database.	(F) Proportion of C emitted from different peat pools as a result of fire severity	Current work (Nelson et al. unpublished)
Include oil and gas development disturbance effects	Test oil and gas development disturbance matrices developed at a workshop of experts hosted by the Canadian Forest Service.	(G) Proportions of C disturbed or removed from the peat profile as a result of different harvest disturbances	Current work/testing and evaluation to be completed
Include peatland and forest harvest disturbance effects	Build disturbance matrices for different harvest methods.	(H) Proportions of C disturbed or removed from the peat profile as a result of different harvest disturbances	Literature review required
Include permafrost effects	Include active and frozen layers in the CaMP model with simple thaw and water dynamics.	(A) Peat temperature module that can be scaled up to the national level.	NEST model Zhang et al., 2003
		(C) Update peatland profile database[1] to include permafrost peatland sites	Current work (Canadian Forest Service, unpublished)

[1] Peat profile database compilations to date used in this study: Zoltai et al., 2000; Garneau unpublished data; Packalen unpublished data; Tarnocai unpublished data.

in the high spatial and temporal variability such that the level of confidence in scaled-up numbers is low (Dinsmore et al., 2009). Therefore, when a residual is high for any one year, it is difficult to determine if the model requires adjusting, or if the measured annualized values are inaccurate. The CH_4 sub-model used in this version of CaMP employed optimal water table depth values taken from the literature (Turetsky et al., 2014). These were daily values, rather than annual values that would be more appropriate for the CaMP and should be developed. Ideally the CH_4 sub-model should be calibrated for annual-scale estimation by using data for annual WTD with corresponding CH_4 emission rates (Table 10). However, we found that annual water table depths were rarely reported in studies providing CH_4 fluxes, so this is a data gap that should be filled in order to improve accuracy of CH_4 emission estimates from the CaMP. Instrumentation for CH_4 flux tower measurement has only become readily available in the past decade (Olsen et al. 2013), which explains why data are relatively sparse and inconsistently scaled as compared to CO_2 flux data (Knox et al. 2019). Therefore, new programs such as the Global Carbon Project's CH4-FLUXNET are vital to setting global standards for gap filling, meta-data, and properly scaling CH_4 flux data. Inopportunely, CH4-FLUXNET does not plan to set a standard and account for winter emissions when only seasonal data are collected, which we argue is an important aspect to creating usable datasets for annual-scale and long-term projections. Still, continued funding and growth of programs such as these will work to expand datasets and baseline data collection and reporting standards that will greatly help to advance CH_4 science. Furthermore, we advocate for more funding towards flux towers in regions that are more representative of areas where peatlands are widespread in Canada such as in the Boreal Plains and Boreal Shield ecozones (Webster et al., 2018).

The current model's evaluation did not reveal any significant difference in model accuracy between peatlands in western and eastern Canada. That being said, we acknowledge that the data used to generate the C density curves, as well as much of the data for model calibration were taken from western Canada (Zoltai et al., 2000). As previously mentioned, the Canadian Forest Service is currently compiling more data in their peat profile C database and much of this work is focused on improving representation of eastern Canada. This additional data will be used to test, and if needed, re-calibrate the C density model for eastern Canada.

While the CaMP (v2.0) framework includes swamps as a wetland category, as stated previously, we lacked data to calibrate or evaluate the swamp class and it was therefore omitted from this study. Additionally, authors Webster et al. (2018), who built the map product designed to apply the CaMP nationally, did not include the swamp category, stating that it was too difficult to map mainly because of regional variation in overstory vegetation. Although swamps do not make up a large area of land nationally, we acknowledge that they can be important regionally (e.g., southern Ontario). Future work is required both to improve swamp mapping, as well as C profile and emission estimates for swamp ecosystems by region (Table 10) in order to calibrate and assess the performance of CaMP for swamps.

The CaMP (v2.0) only includes wildfire as a potential disturbance type, and uses average depth of burn. In order make use of the CaMP to estimate the impacts of natural and anthropogenic disturbances on GHG emission projections, future versions of CaMP need to include more disturbance types requiring the development of wildfire disturbance matrices representing differing fire severities and timber and peat harvest disturbance matrices, for each peatland type. Timber harvest on peatlands is of particular importance in the claybelt of Quebec and Ontario where a significant portion of harvest takes place on lowland or transition sites (Haavisto and Jeglum 1991). A literature review to address the development of harvest disturbances has been conducted, but there are gaps in data for depth and amount of C disturbed and for effects on water table depth, depending on the harvest method and associated disturbances (e.g., all season vs. temporary roads). Other disturbance types that must be addressed in a future version of CaMP are those associated with oil and gas development. A workshop attended by experts was held in November 2018 to develop a suite of 14 different disturbance matrices for the effect of oil and gas development on peatlands that could be used to model 21 different disturbance types including well pads, pipelines, seismic lines, and surface mining. The next step is to run these disturbance matrices with the CaMP (v2.0) in an appropriate region of Canada, and test their application by comparing modeled estimates to observed data. However, we know that appropriate observed data for this type of comparison are extremely limited and this represents another vital data gap that will need to be addressed (Table 10).

10. Summary and conclusion

A model framework for national GHG emission and removal estimation for Canadian peatlands (CaMP v2.0) was built and tested. It provides the National Forest Carbon Monitoring and Reporting System with a module that can work in tandem with the newly developed GCBM for upland forests, to provide a more complete approach to C reporting in Canada. The purpose of CaMP (v2.0) was to provide a simple model foundation that could be widely-applied across several peatland types and regions in Canada, and to evaluate different aspects of the model's structure and parameterization in order to inform the second phase of model development. The CaMP takes a novel approach to modelling peatland hydrology at a large spatial scale by using the nationally available fire weather index, Drought Code, to predict long-term and annual water table depth. The task of model parameterization and calibration was supported by compiling two large databases (Bona et al., 2018; Zoltai et al., 2000; Garneau unpublished data; Tarnocai unpublished data; Packalen unpublished data). A simple sub-model for CH_4 was also tested, and performed relatively well considering the high complexity of CH_4 emission dynamics and the high spatial and temporal variability in measured CH_4 fluxes. The CaMP's sensitivity to climatic variables was assessed, and the model was evaluated against observed data. Results suggest that the CaMP (v2.0) structure is appropriate for large spatial and temporal scale estimation and behaves as expected in relation to environmental variables, but that several areas require further attention in order to increase the amount of finer-scale explained variation in NEE (CO_2) and CH_4 emissions (Table 10). Future steps include expanding existing peatland databases to re-calibrate decay rates, and better parameterize NPP rates for certain peatland classes and regions. We recommend that a national soil temperature module that fits within the CaMP framework be built in order to replace the current simple air temperature Q_{10} relationship for peat decomposition. Doing this will also facilitate inclusion of permafrost dynamics in the CaMP by modeling an active and non-active layer, and permafrost thaw. Several data gaps were identified that must be addressed (Table 10). In particular we emphasize the importance of standardizing annual-scaled CH_4 flux rates and the inclusion of basic meta-data around flux tower sites (such as annual water table depth) in order to improve the science on modeling CH_4 at large-temporal and spatial scales. Future steps are also planned to include a wide-array of natural and anthropogenic disturbances in the CaMP.

CRediT authorship contribution statement

Kelly Ann Bona: Conceptualization, Formal analysis, Data curation, Software, Validation, Methodology, Visualization, Writing - original draft, Writing - review & editing. **Cindy Shaw:** Conceptualization, Methodology, Validation, Data curation, Writing - review & editing, Funding acquisition, Project administration. **Dan K. Thompson:** Conceptualization, Methodology, Validation, Data curation, Formal analysis, Visualization, Writing - original draft, Writing - review & editing. **Oleksandra Hararuk:** Conceptualization, Formal analysis, Software, Validation, Writing - original draft, Writing - review & editing. **Kara Webster:** Conceptualization, Methodology, Data curation, Writing - review & editing, Funding acquisition, Project administration. **Gary Zhang:** Software. **Mihai Voicu:** Formal analysis, Resources. **Werner A. Kurz:** Conceptualization, Writing - review & editing, Resources, Funding acquisition, Project administration.

Declaration of Competing Interests

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

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