



Impact of power line corridors on carbon stocks of upland forests in eastern Canada

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Abstract

Aim Hydroelectricity is considered a renewable energy source with low greenhouse gas emissions. However, little attention has been given to the impact of the construction and maintenance of power line corridors on soil and vegetation. This study assessed the effect of power line corridors on terrestrial carbon pools in upland forests across a 750-km gradient from deciduous forests to open-canopy taiga in Quebec (Canada). **Methods** Quantification of ecosystem carbon pools (live biomass, dead organic matter and soil) was

carried out using transects of plots located in: i) the power line corridor, ii) the forest located at 20 m from the corridor edge (forest edge), and iii) the forest located at 60 m from the corridor (interior forest, considered as a control).

Results Ecosystem carbon stocks were significantly lower in the corridors, with reductions of 36–133 Mg of C ha⁻¹ compared with interior forests. Yet, corridors still stored substantial carbon (average of 108.5 ± 13.7 Mg of C ha⁻¹). While biomass of trees and shrubs was significantly reduced in the corridor compared with interior and edge forests, total soil carbon stocks (which varied from 80.11 to 169.91 Mg of C ha⁻¹ across bioclimatic domains) were similar between corridors and forests. We also observed similar carbon stocks between edge and interior forests, suggesting that the influence of power line corridors is not noticeable at 20 m and beyond in adjacent forest ecosystems.

Conclusion Our study improves the quantification of carbon storage across forested landscapes and the GHG footprint of hydroelectricity.

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Boreal and temperate forests · Hydroelectricity

Introduction

Hydroelectricity is an energy source with lower greenhouse gas (GHG) emissions than other important energy sources (IPCC 2011). The accuracy of the

carbon footprint of hydroelectricity production has been improved over the past decades, through studies that estimate GHG emissions from dam construction, flooding and reservoir dynamics (Prairie et al. 2018; Rust et al. 2022; Song et al. 2018). However, little attention has been paid to the impact of establishing and maintaining power line corridors on land biogenic carbon stocks (Levasseur et al. 2021).

The guidance for reporting GHG emissions to the United Nations Framework Convention on Climate Change (UNFCCC) indicates that the installation of power lines in forested lands is considered a conversion from forest to establishments (i.e., a land-use change) (IPCC 2006). In Canada, the emissions associated with forest land-use conversion are estimated with the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3), a forest carbon cycling model (Environment and Climate Change Canada 2026). The methodology follows the guidelines for reporting emissions from the land use, land-use change and forestry sector (Kurz et al. 2009; Penman et al. 2003).

The disturbance associated with the establishment of the right-of-way of power lines (hereafter referred to as power line corridors) is simulated as deforestation in CBM-CFS3: all merchantable trees within the corridor are assumed to be removed with no further vegetation biomass growth, and the decay of dead organic matter is simulated over the following 20-year period (Kurz et al. 2009). Yet, evidence suggests that such corridors harbour shrub, forb and herbaceous vegetation communities, the carbon dynamics of which are ignored in current estimations (Luken et al. 1992; Mercier et al. 2001; Wagner et al. 2014). Depending on the latitude (and hence vegetation growth), vegetation in the corridors is controlled approximately every five to eight years to prevent tree encroachment and ensure the safety and reliability of the electrical network (Létourneau et al. 2013). Vegetation maintenance consists of mechanically cutting or chemically treating trees and shrubs that could otherwise cause electric arcs if allowed to reach their mature height. The absence of a tree canopy cover and the managed low-vegetation state could promote the persistence and dominance of species with distinct carbon allocation strategies (e.g., carbohydrate storage in roots), which may influence the above- and belowground carbon inputs to the soil within corridors

compared with forests undisturbed by the corridor (Landuyt et al. 2020; Luken et al. 1992; Wagner et al. 2014).

Moreover, linear disturbances, such as power lines, create forest edges along the corridor border (Reinmann and Hutrya 2017; Smith et al. 2018). In such edges, changes in microclimate are expected to occur compared with ecosystems located further from the edge (i.e., interior forests), as well as increased exposure to wind, light, drought, fire and nutrient inputs such as nitrogen (Smith et al. 2018). Depending on site climatic conditions, but also edge aspect (i.e., cardinal orientation) (Hofmeister et al. 2019), these changes can be either favorable for vegetation growth (for example, due to increased availability of light and nitrogen), or unfavorable i.e., creating a carbon drawdown (Meeussen et al. 2021; Smith et al. 2018). Yet, the global analysis of Yang et al. (2025) found consistently lower aboveground biomass density near forest edges (measured at 30 m from an opening) compared to interior forests, with larger effects in tropical, subtropical and temperate forests compared with boreal ecosystems.

Most studies evaluating the impacts of power lines on ecosystems have focused on vegetation mortality during the construction phase and biodiversity changes during the operational phase, but not on carbon dynamics specifically (Biasotto and Kindel 2018; Richardson et al. 2017). The province of Quebec, in eastern Canada, possesses the largest power transmission network in North America, spread over 30 000 km and 170 000 ha, largely across forested landscapes (Hydro-Québec 2026; Létourneau et al. 2013). Here, we evaluated the impact of power line corridors on the distribution and variability of carbon stocks (including live biomass, dead organic matter and soil) across a bioclimatic gradient of upland forested landscapes in the province of Quebec. Specifically, we quantified and compared ecosystem carbon stocks according to their position relative to the power line corridors and the bioclimatic gradient of Quebec forests and investigated how the impact of the power line corridors varies with this gradient. The findings from this study can be used to refine estimates of carbon stocks and stock changes associated with the conversion of forests to settlements, such as power lines, in national GHG inventories, and to inform future monitoring, reporting, and verification activities aimed at characterizing land-use changes.

Materials and methods

Study area

The study area covered four bioclimatic domains of the Quebec province on a 750 km gradient from North to South; bioclimatic domains are named after the climax forest vegetation types found on sites of medium drainage and fertility: 1) black spruce–moss (*Picea mariana* (Mill.) BSP -*Pleurozium schreberi* (Brid.) Mitt) [BS-M], a bioclimatic domain covering 391 700 km², i.e., 26% of Quebec territory, 2) balsam fir–white birch (*Abies balsamea* (L.) Mill—*Betula papyrifera* Marsh.) [BF-WB], which covers 139 700 km², 3) balsam fir–yellow birch (*Abies balsamea*—*Betula alleghaniensis* Britton) [BF-YB], which covers 99 700 km², and 4) sugar maple–yellow birch (*Acer saccharum*—*Betula alleghaniensis*) [SM-YB], which covers 65 400 km² (Saucier et al. 2009) (Fig. 1). These domains broadly represent the three major forested sub-zones of vegetation in Eastern North America: the sub-zone of the continuous boreal forest (BS-M and BF-WB), the sub-zone of the mixed forest (BF-YB), and the sub-zone of the Northern deciduous forest (SM-YB). The mean annual temperatures of the four bioclimatic domains range from 4.6 to −1.6 °C, with degree-days above 5 °C from 955 in the northeast to 1738 in the southwest, and the mean precipitation from 430 to 639 mm (1991–2018 observed values) (Environment and Climate Change Canada 2022).

Experimental design

The experimental design comprised transects of inventory plots that ran perpendicularly to the power line corridor (Fig. 1). A total of 18 transects were studied. The transect selection was based on a series of requirements. Transects had to be located in relatively flat, upland areas (i.e., on mineral soils) to ensure that differences in vegetation were not primarily driven by slope. At least 150 m of undisturbed forests had to be present on each side of the power line corridor, i.e., forest stands with no recent signs of harvest or other major stand-replacing disturbance (e.g., wildfire or severe insect epidemics; however, small-scale disturbances such as localized windthrows were possible). These undisturbed forests also had to be broadly representative of the general vegetation

characteristics of the area in terms of species composition; the potential vegetation of the selected forests was representative of 6% (for SM-YB3) to 47% (for BF-WB2 and BF-WB3) of the area of their respective ecological region (Table 1).

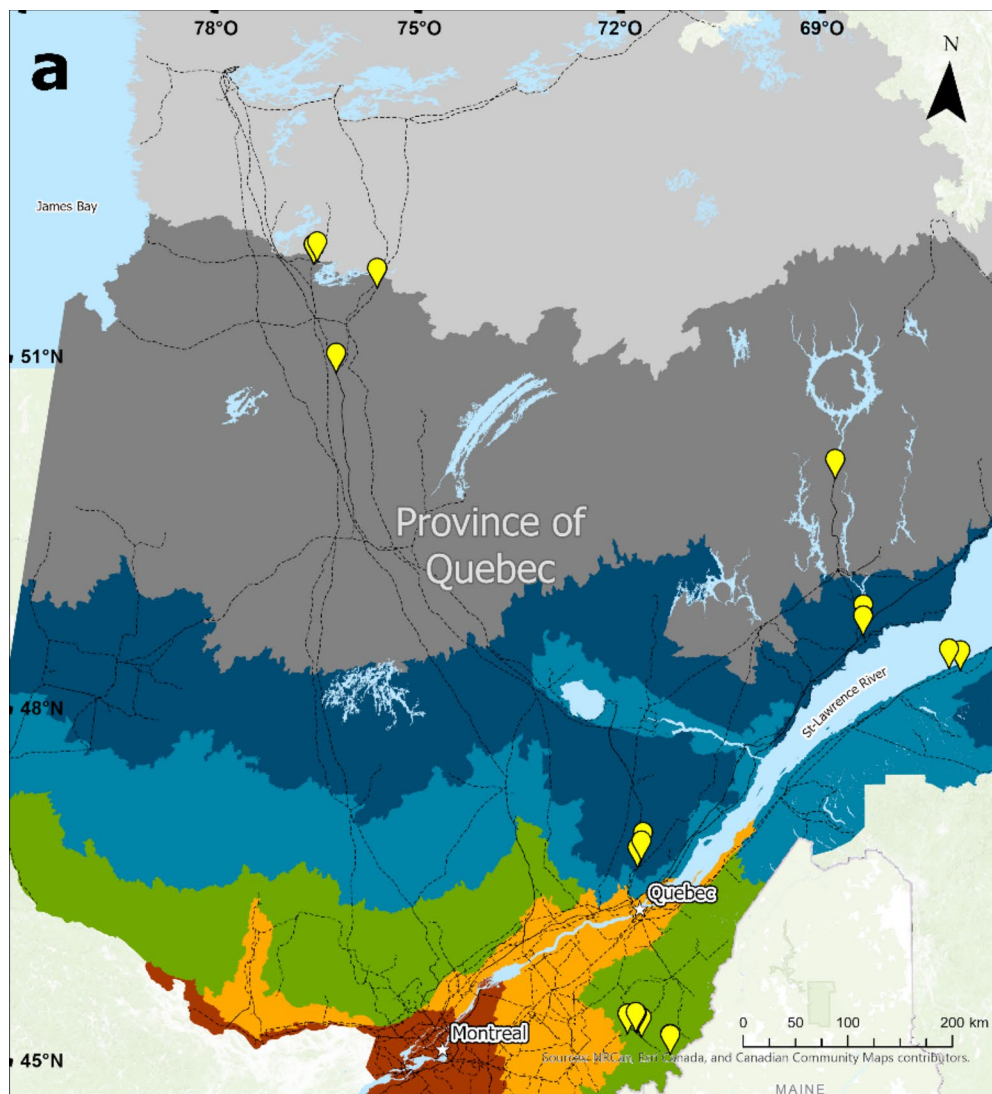
Most power lines from selected corridors were roughly oriented North–South, as this is the most common power line aspect in Quebec (Supplementary material, Table S1). The date of the power line corridor and the last vegetation maintenance treatment of the corridor varied across transects (Supplementary material, Table S1).

Inventory plots were located at three different positions along each transect: within the corridor, within the corridor edge and within the interior forest, the latter considered as a control (Fig. 1). For each transect, two to three plots were positioned within the corridor of the power line. The center of another plot was located at 20 m from the border of the corridor towards the interior forest and was considered the edge plot, and the center of another plot was located at 60 m from the border and was considered the interior plot. When possible, one edge plot and one interior plot were established on each side of the corridor (2–3 corridor plots + 1–2 edge plots + 1–2 interior plots = 4–7 inventory plots per transect).

In total, four transects were established in the spruce–moss (BS-M) bioclimatic domain, six in the balsam fir–white birch (BF-WB) domain, three in the balsam fir–yellow birch (BF-YB) domain and five in the sugar maple–yellow birch (SM-YB) domain, for a total of 18 transects (Table 1). Among the 18 transects, soil drainage varied from well to somewhat poorly drained, but drainage was similar among plots of a given transect. For all transects, surficial deposits were generally classified as glacial till.

Sampling procedure

The sampling procedure was adapted from Canada's National Forest Inventory guidelines (National Forest Inventory 2008). A schematic representation of the sampling can be found in Figure S1 in Supplementary material. A large inventory plot with a radius of 11.28 m (area of 400 m²) was used for large trees (see below for definitions). A smaller plot with a radius of 3.99 m (50 m²) located at the center of the larger one was used for small trees, shrubs and stumps. For woody debris, two



Bioclimatic domains

■ Spruce-lichen

■ Spruce-moss

■ Balsam fir-yellow birch

■ Balsam fir-white birch

■ Maple-yellow birch

■ Maple-bitternut hickory

■ Maple-basswood

☆ Cities

--- Power line

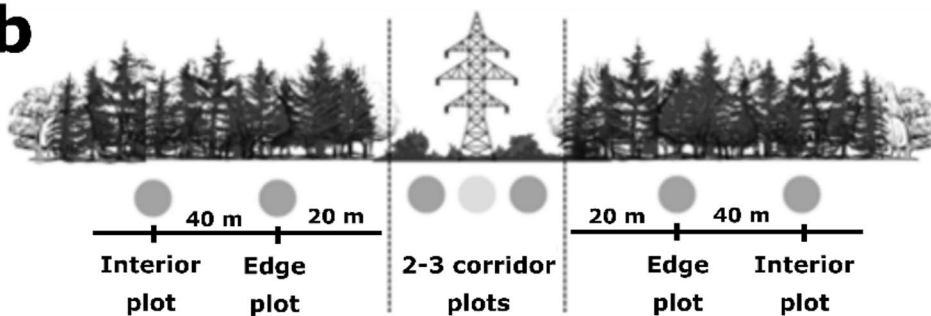
● Study sites

Louis-Lou Tardif-Samson

Source : Hydro-Québec, Données Québec.

Projected Coordinate System : NAD 1983 Quebec Lambert

b



◀**Fig. 1 a** Location of transects (1 transect=1 study site) throughout the power line network across four sampled bioclimatic domains: Black spruce–moss, Balsam fir–white birch, Balsam fir–yellow birch and Sugar maple–yellow birch (Saucier et al. 2009). **b** Transect of inventory plots. The center of the Edge and Interior plots were located at 20 and 60 m from the border of the power line right-of-way, respectively

perpendicular transects of 20 m for edge and interior plots, and 10 m for corridor plots, and crossing at the center of plots were used. Three (for corridor plots) or four (for edge and interior plots) 1-m² microplots for tree regeneration and understory vegetation were located 5 m from the plot center along woody debris transects. One soil sampling station was located within each microplot; for edge and interior plots, two additional stations were placed at the ends of the woody debris transects, for a total of 3 soil stations for corridor plots and 6 soil stations for edge and interior plots.

Within the 400-m² plot, all live and dead trees with > 1.3 m in height and diameter at breast height (DBH, measured at 1.3 m height) ≥ 9 cm were measured. For each tree, species, status (alive or standing dead) and DBH were recorded; the height of broken trees was also recorded.

Within the 50-m² plot, small trees with a height ≥ 1.3 m and a DBH < 9 cm were measured. Shrubs with a diameter at stump height (DSH) ≥ 1 cm were also recorded. The following variables were collected: species, status (alive or standing dead) and DBH or DSH. In the same plot, stumps with a height < 1.3 m and a top diameter ≥ 4 cm were inventoried; their species, diameter under bark at stump height, height at the highest ground point and decay class were recorded. The decay classes had 5 levels according to the state of wood texture, branches, bark and shape (1: wood intact and hard with branches and bark; 2: wood hard to partially decomposed with branches sometimes present and bark partially intact); 3: large wood portions decomposed, no branches, trace of bark; 4: decomposed but still hard in the centre, no branches and no bark; 5: totally decomposed and soft) (National Forest Inventory 2008).

Along the two sampling transects, two categories of woody debris lying on the ground were inventoried. Coarse woody debris with a diameter > 3 cm that intersected the transects were individually tallied along the full length of the two transects (20 m for edge and control plots, 10 m for corridor plots);

their species, decay class (using the classification described above) and diameter were recorded and measured at the intersection with the transect. On the first 5 m and last 5 m of each transect for edge and interior plots, and on their full length for corridor plots, small woody debris with a diameter > 1 cm and ≤ 3 cm were counted, and their average species and decay class were recorded.

Within each 1-m² microplot, all herbaceous and woody plants with DSH < 1 cm were clipped at the base over the entire microplot. The clipped vegetation was placed in paper bags, oven-dried at 65 °C for 72 h and then weighed.

Within each microplot, the L horizon (mostly consisting of live mosses) and fine woody debris (diameter ≤ 1 cm) were sampled over a 400-cm² quadrat, when present. The FH horizons, i.e. the soil organic layers consisting of partially or totally decomposed organic matter (corresponding to Of and Oh in IUSS Working Group WRB (2022)) were then sampled over the same quadrat, and their depth was recorded. The mineral horizons were sampled volumetrically using a metal tube 4.8 cm in diameter at three depths: 0–15, 15–35, and 35–55 cm. In total, ten soil samples were collected for edge and interior plots: six samples at 0–15 cm of depth, three samples at 15–35 cm and one sample at 35–55 cm. For corridor plots, six soil samples were collected: three samples at 0–15 cm of depth, two samples at 15–35 cm and one sample at 35–55 cm.

Quantification of carbon stocks

The ecosystem carbon stocks in this study were based on the IPCC definitions of carbon pools for agriculture, forestry, and other land uses (IPCC 2006) and adapted to the specific features of the vegetation dynamics of corridors.

For large trees, small trees and shrubs, total dry biomass was estimated using allometric models based on DBH or DSH and species, following Lambert et al. (2005) and Tremblay et al. (2006). Aboveground dry biomass estimates of trees and shrubs were converted to carbon stocks using species-group-specific carbon fractions from Martin et al. (2018). For boreal forests, carbon fractions of 0.468 and 0.492 were applied to hardwood and conifer species, respectively. Shrub biomass, which was mainly composed of broadleaf

Table 1 Characteristics of the study sites, including bioclimatic domain (BS-M: Black spruce (*Picea mariana*)-moss; BF-WB: balsam fir (*Abies balsamea*)-white birch (*Betula papyrifera*); BF-YB: balsam fir-yellow birch (*Betula alleghaniensis*); SM-YB: Sugar maple (*Acer saccharum*)-yellow

birch (Saucier et al. 2009), transect number, geographic coordinates, altitude, tree density (trees ≥ 9 cm in DBH), mean annual precipitation (MAP), mean annual temperature (MAT), potential vegetation at climax and dominant tree species (mean relative abundance, %)

Bioclimatic domain and Transect number	Coordinates	Altitude (m)	Tree density (Nb ha ⁻¹)	MAP (mm)	MAT (°C)	Potential vegetation at climax	Dominant species (% average abundance)
BS-M1	51.243N –75.675W	342	850	890	–0.9	Black spruce–Moss-Ericacea (RE2; 61% of ecological plots in region 6d)	<i>Picea mariana</i> (45%) <i>Pinus banksiana</i> (32%) <i>Salix</i> spp. (16%)
BS-M2	52.161N –76.141W	242	1750	836	–1.5	Black spruce–Lichens (RE1; 14% of ecological plots in region 6d)	<i>Picea mariana</i> (81%) <i>Salix</i> spp. (6%) <i>Populus tremuloides</i> (3%)
BS-M3	52.188N –76.098W	290	1975	823	–1.4	Black spruce–Lichens (RE1; 14% of ecological plots in region 6d)	<i>Pinus banksiana</i> (99%) <i>Picea mariana</i> (1%)
BS-M4	52.012N –75.222W	305	638	841	–1.4	Black spruce–Lichens (RE1; 14% of ecological plots in region 6d)	<i>Picea mariana</i> (92%) <i>Pinus banksiana</i> (8%)
BF-WB1	47.164N –71.258W	592	1450	1338	2	Balsam fir-Yellow birch (MS1; 9% of ecological plots in region 5e)	<i>Betula papyrifera</i> (56%) <i>Abies balsamea</i> (44%)
BF-WB2	47.296N –71.194W	812	1638	1382	0.8	Balsam fir-White birch (MS2; 47% of ecological plots in region 5e)	<i>Abies balsamea</i> (89%) <i>Betula papyrifera</i> (10%) <i>Picea glauca</i> (1%)
BF-WB3	47.222N –71.213W	739	1775	1368	1.3	Balsam fir-White birch (MS2; 47% of ecological plots in region 5e)	<i>Abies balsamea</i> (70%) <i>Betula papyrifera</i> (27%) <i>Picea glauca</i> (3%)
BF-WB4	49.295N –68.401W	119	575	1006	2	Balsam fir-Black spruce (RS2; 21% of ecological plots in region 5g)	<i>Salix</i> spp. (47%) <i>Populus tremuloides</i> (23%) <i>Picea mariana</i> (21%)
BF-WB5	49.200N –68.401W	111	1250	1008	2.2	Black spruce–Moss-Ericacea (RE2; 18% of ecological plots in region 5g)	<i>Picea mariana</i> (100%)
BF-WB6	50.552N –68.779W	434	250	968	–1.1	Balsam fir-Black spruce (RS2; 21% of ecological plots in region 5g)	<i>Betula papyrifera</i> (74%) <i>Picea mariana</i> (23%) <i>Abies balsamea</i> (3%)
BF-YB1	48.908N –67.278W	16	1325	986	3.4	Balsam fir-Yellow birch (MS1; 26% of ecological plots in region 4f)	<i>Abies balsamea</i> (38%) <i>Betula papyrifera</i> (33%) <i>Acer rubrum</i> (11%)

Table 1 (continued)

Bioclimatic domain and Transect number	Coordinates	Altitude (m)	Tree density (Nb ha ⁻¹)	MAP (mm)	MAT (°C)	Potential vegetation at climax	Dominant species (% average abundance)
BF-YB2	48.889N –67.129W	394	888	1107	1.9	Balsam fir-White birch (MS2; 32% of ecological plots in region 4h)	<i>Acer spicatum</i> (30%) <i>Populus tremuloides</i> (22%) <i>Betula papyrifera</i> (21%)
BF-YB3	48.890N –67.130W	377	963	1102	1.9	Balsam fir-White birch (MS2; 32% of ecological plots in region 4h)	<i>Betula papyrifera</i> (31%) <i>Acer spicatum</i> (25%) <i>Prunus pensylvanica</i> (12%)
SM-YB1	45.729N –71.303W	332	1025	1157	4.6	Yellow birch-Balsam fir-Sugar maple (MJ1; 28% of area of region 3d)	<i>Acer rubrum</i> (39%) <i>Abies balsamea</i> (33%) <i>Betula alleghaniensis</i> (28%)
SM-YB2	45.566N –70.770W	420	850	1160	3.9	Yellow birch-Balsam fir (MJ2; 26% of area of region 3d)	<i>Populus tremuloides</i> (50%) <i>Abies balsamea</i> (s) (36%) <i>Acer rubrum</i> (7%)
SM-YB3	45.703N –71.129W	414	1325	1172	4.2	Eastern white-cedar (RB1; 6% of area of region 3d)	<i>Abies balsamea</i> (67%) <i>Populus tremuloides</i> (11%) <i>Acer saccharum</i> (5%)
SM-YB4	45.737N –71.188W	372	625	1167	4.4	Sugar maple-Yellow birch (FE3; 20% of area of region 3d)	<i>Acer rubrum</i> (38%) <i>Betula alleghaniensis</i> (19%) <i>Picea glauca</i> (19%)
SM-YB5	45.745N –71.203W	365	1600	1166	4.5	Yellow birch-Balsam fir (MJ2; 26% of area of region 3d)	<i>Acer rubrum</i> (77%) <i>Betula alleghaniensis</i> (9%) <i>Abies balsamea</i> (7%)

species, was converted using the hardwood carbon fraction (0.468).

Clipped understory vegetation samples were oven-dried, and their dry mass was multiplied by 0.5 based on the general value of Penman et al. (2003) to estimate carbon content. For stumps and woody debris, carbon stocks were estimated using models from Canada's National Forest Inventory based on volume and wood density by decay class and species, and the 0.5 carbon fraction of Penman et al. (2003). To estimate the root biomass, the sum of the biomass of large trees, small trees, shrubs and understory vegetation was multiplied by 0.256 based on Penman et al. (2003).

Soil samples were air-dried for at least one week, after which fine soil was separated from the coarse content by sieving at 6 mm for the FH horizon samples and 2 mm for mineral soil samples. The coarse portion of the soil was divided into organic fractions (roots, dead wood and other organic material) and mineral fractions (rocks). All components of dried soil samples were weighed before and after sieving to calculate soil bulk density (Federer et al. 1993). The fine portion of soil samples was ground to 250 µm, and its carbon concentration (%) was measured by a Leco TruMac CNS Analyzer (LECO Corporation, USA). The carbon content of the fine fraction of soil was calculated via its C concentration and fine soil

bulk density (estimated as the weight of the fine fraction on the total volume of the core); this corresponds to the hydrid method described in Lang et al. (2025). The carbon content of the coarse organic fractions was estimated at 0.5 of their oven-dried mass.

All carbon stocks were finally converted into megagrams of carbon per hectare (Mg of C ha^{-1}).

Statistical analyses

All statistical analyses were performed in R version 4.4.2 (R Core Team, 2024). Linear mixed-effects models (LMMs) were fitted using the *lmer* function from the ‘lme4’ package in R (Bates et al., 2023) to evaluate differences among the carbon pools and sub-pools, including aboveground live biomass of live trees and shrubs, aboveground biomass of understory vegetation, litter (L horizon, mostly live mosses), live roots, total above- and belowground live biomass, standing dead trees, stumps, woody debris, total dead wood (dead trees + stumps + woody debris), total aboveground biomass (live trees and shrubs + understory vegetation + litter + dead trees + stumps + woody debris), organic FH horizons, mineral soil (0–55 cm), soil coarse organic fraction, total soil (FH horizons, mineral soil and soil coarse organic fraction), and total ecosystem carbon stocks.

For each response variable, fixed effects included transect position (three levels: corridor, edge, and interior forest), bioclimatic domain (four levels: black spruce–moss, balsam fir–white birch, balsam fir–yellow birch, and sugar maple–yellow birch), and their interaction. The site was included as a random intercept to account for the non-independence of observations collected within the same sampling site.

Carbon stock data were log-transformed using $\log(x+1.1)$ to accommodate zero values and satisfy the assumptions of residual normality and homogeneity of variance. Model assumptions were verified by visual inspection of residual and quantile–quantile plots using the *check_model* function in the ‘performance’ R package (Lüdtke et al. 2021).

The significance of fixed effects was assessed using F-tests with Satterthwaite’s approximation for denominator degrees of freedom. When the interaction between transect position and bioclimatic domain was significant ($p < 0.05$), pairwise comparisons were conducted among simple effects using estimated marginal means. When the interaction was

not significant, pairwise comparisons were performed only for significant main effects. Estimated marginal means were computed using the *emmeans* function of the ‘emmeans’ package (Lenth et al., 2024) and back-transformed to the original units for presentation (acknowledging that these values are likely biased following the back-transformation (Neyman and Scott 1960)). Differences were considered statistically significant at $p < 0.05$.

Results

The model p-values and F-values for each tested effect (position, bioclimatic domain and position $\times$ bioclimatic domain) and the mean values of carbon stocks for each carbon pool are presented by position and by bioclimatic domain in the Supplementary material (Tables S2 and S3, respectively).

We found that most ecosystem carbon pool estimates were significantly affected by transect position, with the exception of the soil coarse organic fraction, mineral horizons and total soil carbon pools (Table 2). Estimates of carbon stocks were also significantly different between the bioclimatic domains for understory vegetation, soil mineral horizons, total soil and total ecosystem stocks. Furthermore, significant interactions between transect position and bioclimatic domain were detected for live roots, total live biomass, dead trees, woody debris and total dead wood (Table 2).

Transect position had a mostly consistent, statistically significant effect on most estimates of ecosystem carbon pools (Table 2). Indeed, significantly lower carbon stocks were observed in the corridor plots compared with the edge and interior plots for live trees and shrubs, total aboveground biomass (see details of the latter in Fig. 2a), soil organic horizons and total ecosystem stocks. The estimates of carbon stocks in the litter were also significantly lower in the corridor plots than in the interior plots but did not differ from those in the edge plots. Conversely, the understory vegetation measured in the corridor plots had significantly higher estimates of carbon stocks than those of the edge and interior plots. Finally, the estimates of carbon stocks for soil mineral horizons, soil coarse organic fraction and the total soil carbon stocks (see details of the latter in Fig. 2b) did not differ according to the position along the transect (Table 2).

Table 2 Mean estimated carbon stocks for ecosystem pools (Mg C ha⁻¹) and their standard error. Significant differences ($p < 0.05$) among positions or bioclimatic domains are indi-

cated by different letters and bold characters. When the interaction between position and domain was significant, detailed results are presented in Fig. 3

Carbon pools	Position			Bioclimatic domain			
	Corridor	Edge	Interior	Black spruce-moss	Balsam fir-white birch	Balsam fir-yellow birch	Sugar maple-yellow birch
Live trees and shrubs	0.25 ± 0.17a	46.87 ± 6.35b	43.39 ± 6.17b	9.75 ± 2.77a	10.79 ± 2.44a	15.51 ± 4.97a	18.10 ± 4.37a
Understory vegetation	0.88 ± 0.13b	0.35 ± 0.09a	0.50 ± 0.11a	1.06 ± 0.26b	0.85 ± 0.19b	0.35 ± 0.2ab	0.15 ± 0.14a
Litter (L horizon) ^A	3.54 ± 0.49a	3.95 ± 0.55ab	4.18 ± 0.56b	4.08 ± 1.17a	4.44 ± 1.01a	4.32 ± 1.43a	2.85 ± 0.79a
Live roots	Significant interaction between position and bioclimatic domain (Fig. 3)						
Total live biomass ^B	Significant interaction between position and bioclimatic domain (Fig. 3)						
Dead trees	Significant interaction between position and bioclimatic domain (Fig. 3)						
Stumps	0.10 ± 0.06a	0.36 ± 0.08b	0.27 ± 0.08ab	0.10 ± 0.11a	0.22 ± 0.1a	0.30 ± 0.15a	0.36 ± 0.12a
Woody debris	Significant interaction between position and bioclimatic domain (Fig. 3)						
Total dead wood ^C	Significant interaction between position and bioclimatic domain (Fig. 3)						
Total aboveground ^D	7.40 ± 0.95a	56.35 ± 7.50b	71.26 ± 8.73b	24.86 ± 5.65a	33.08 ± 5.99a	33.62 ± 6.80a	36.54 ± 9.63a
Soil organic (FH) horizons	17.20 ± 4.58a	25.40 ± 6.71b	23.90 ± 6.56b	34.01 ± 20.65a	34.31 ± 16.13a	13.46 ± 10.51a	14.29 ± 7.89a
Soil mineral horizons	82.33 ± 7.56a	80.80 ± 7.87a	80.56 ± 8.57a	49.52 ± 8.80a	117.51 ± 16.64c	71.37 ± 14.69ab	104.47 ± 16.59bc
Soil coarse organic fraction	8.63 ± 1.24a	12.14 ± 1.80a	10.92 ± 1.80a	11.64 ± 2.96a	12.00 ± 2.44a	9.57 ± 2.91a	8.94 ± 2.13a
Total soil ^E	84.90 ± 13.47a	128.64 ± 23.24a	120.49 ± 24.72a	80.11 ± 21.02a	169.91 ± 35.16b	102.15 ± 31.83ab	103.63 ± 24.75ab
Total ecosystem ^F	108.55 ± 13.68a	217.82 ± 30.88b	194.63 ± 31.08b	126.14 ± 26.94a	231.88 ± 39.42b	164.62 ± 41.49ab	159.08 ± 30.90ab

^AMostly consisting of live mosses^BLive trees and shrubs + Understory vegetation + L horizon + Live roots^CDead trees + Stumps + Woody debris^DLive trees and shrubs + Understory vegetation + L horizon + Dead trees + Stumps + Woody debris^ESoil organic horizons + Soil mineral horizons + Soil coarse organic fraction^FTotal live biomass + Total dead wood + Total soil

The bioclimatic domain effect differed among pools (Table 2). For live trees and shrubs, estimates of carbon stocks showed no latitudinal effect, with mean estimates ranging from 9.75 to 18.10 Mg of C ha⁻¹ (Table 2). The estimates of carbon stocks of the understory vegetation were significantly higher in the northernmost domains (BS-M = BF-WB ≥ BF-YB = SM-YB). The litter and stumps showed uniform carbon stocks across all domains (2.85 to 4.44 Mg of C ha⁻¹ for litter and 0.1 to 0.36 Mg of C ha⁻¹ for stumps). Furthermore, estimates for total aboveground carbon stocks and soil organic horizon carbon stocks did not differ between bioclimatic domains (Table 2; see Fig. 2a for details of total aboveground carbon stocks).

Mineral soil horizons contained the largest carbon stock in all transects, ranging from 49 to 117 Mg of

C ha⁻¹; the highest estimate of carbon content for this pool was found in the BF-WB bioclimatic domain (BS-M < BF-YB ≤ SM-YB ≤ BF-WB). Estimates for total soil carbon stocks (soil organic and mineral horizons + soil coarse organic fraction) followed a similar pattern, with significantly higher values in BF-WB compared to the other domains (BS-M ≤ BF-YB and SM-YB = BF-WB). Finally, total ecosystem carbon stocks were lowest in the BS-M domain (126 Mg of C ha⁻¹) and increased towards southern temperate forests, but were highest in BF-WB (231 Mg of C ha⁻¹) (BS-M ≤ BF-YB = SM-YB ≤ BF-WB) (Table 2).

For all bioclimatic domains, estimated carbon stocks were significantly lower in the corridor plots compared with the edge and interior plots for live roots, total live biomass and dead trees (Fig. 3a, b

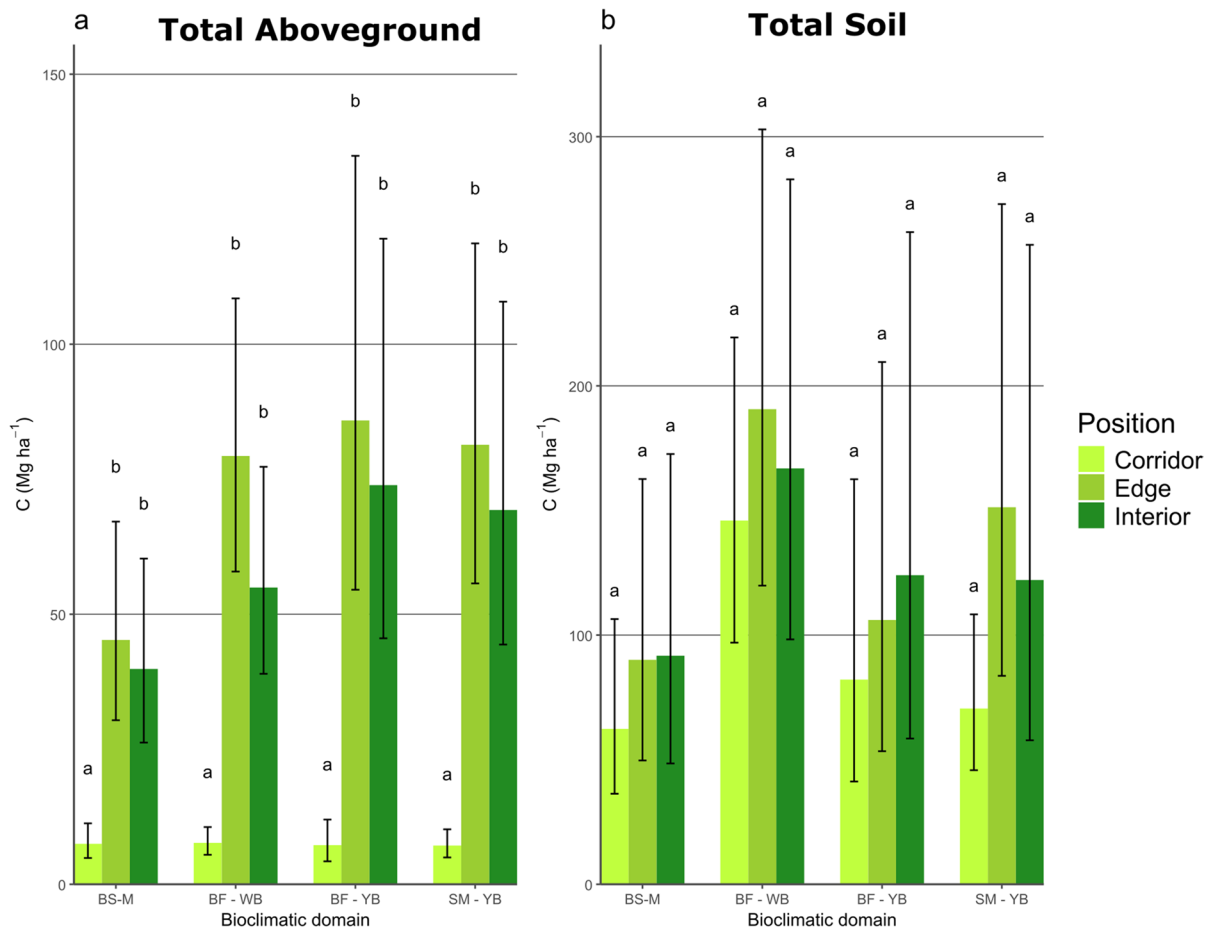


Fig. 2 Estimated carbon stocks between positions according to the bioclimatic domain for the (a) total aboveground and (b) total soil carbon pools. BS-M: black spruce–moss; BF-WB:

balsam fir–white birch; BF-YB: balsam fir–yellow birch; SM-YB: sugar maple–yellow birch

and c). For the woody debris and total dead wood pools, estimated carbon stocks were significantly lower in the corridor than in the edge and interior forests for the BF-WB and BF-YB domains, whereas there was only a significant difference between the corridor and edge positions for the SM-YB domain, and no significant difference for the BS-M domain (Fig. 3d and e, respectively).

Discussion

Ecosystem C storage in the corridors of power lines

Overall, our results showed that the presence of power line corridors primarily affected carbon stocks through changes in live biomass of trees and shrubs

and, to a lesser extent, in dead trees, but had no significant effect on total soil carbon stocks. According to the IPCC Tier 1 Good Practice Guidelines, the conversion of forests to settlements is associated with soil carbon losses ranging from 0 to 20%, with higher losses linked to paved surfaces (IPCC 2006; 2019).

The reduction in carbon stocks of aboveground live biomass of trees and shrubs in corridors was on average 99% compared to the interior forests; it amounts to carbon drawdowns of 31.7, 42.6, 57.9 and 62.4 Mg of C ha⁻¹, respectively for the black spruce–moss (BS-M), balsam fir–white birch (BF-WB), balsam fir–yellow birch (BF-YB) and sugar maple–yellow birch (SM-YB) bioclimatic domains. The carbon drawdown from aboveground live biomass of trees and shrubs caused by the establishment of power line corridors is therefore somewhat expected to decrease along a latitudinal/

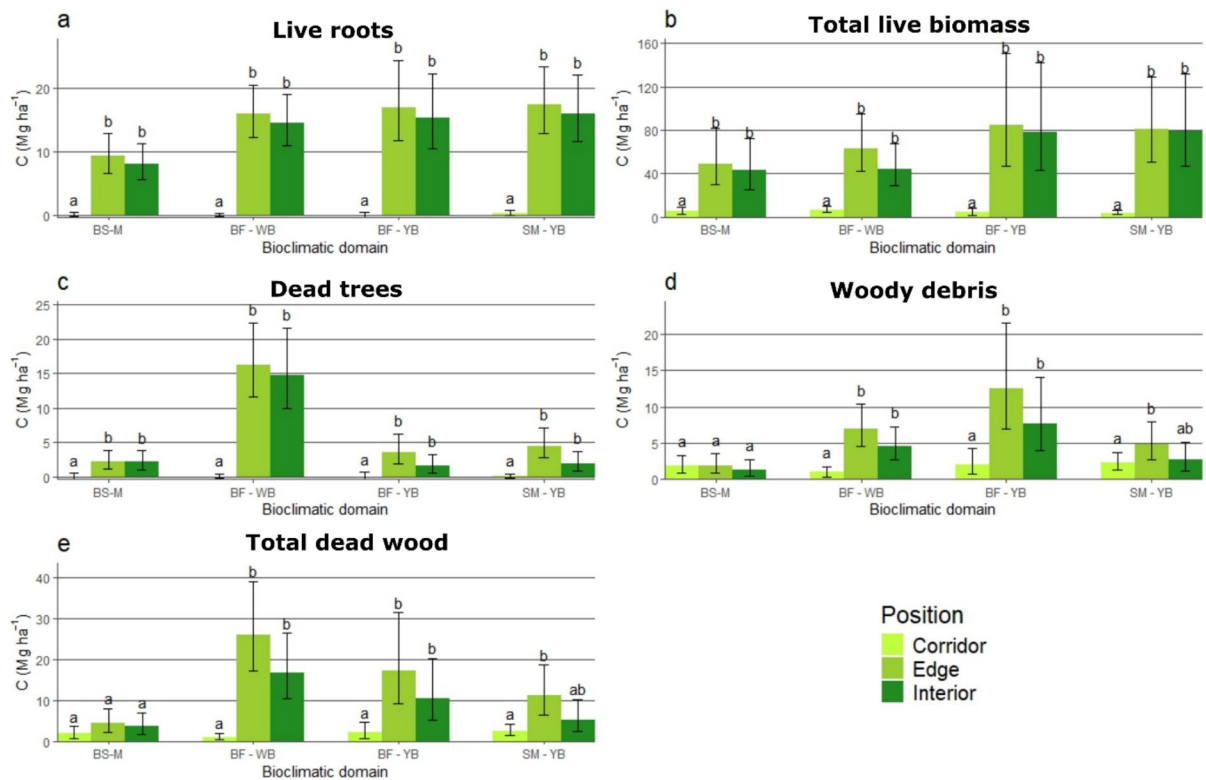


Fig. 3 Significant interactions of estimated carbon stocks between position and bioclimatic domain according to the bioclimatic domain for the (a) live roots, (b) total live biomass, (c) dead trees, (d) woody debris and (e) total dead wood carbon

bioclimatic gradient from our southernmost sugar maple-yellow birch domain (average mean temperature: 4.6 °C), to our northernmost black spruce-moss domain (average mean annual temperature: −1.6 °C).

The live and dead aboveground carbon stocks of the interior forests were relatively similar across all bioclimatic domains except for the black spruce-moss forests ($41.8 \pm 5.3 \text{ Mg of C ha}^{-1}$), in which it was about 58% the amount found in other domains (which varied from $68.7 \pm 9.9 \text{ Mg of C ha}^{-1}$ in the balsam fir-white birch domain to 74.9 ± 4.1 and $73.1 \pm 11.2 \text{ Mg of C ha}^{-1}$ in the balsam fir-yellow birch and sugar maple-yellow birch domains, respectively). The average loss in total (live and dead) aboveground carbon stocks caused by the creation of corridors from previously interior forests was estimated at $34.0 \text{ Mg of C ha}^{-1}$ for the black spruce-moss domain, while the average loss was higher for the other domains, i.e., 60.5, 65.0 and 65.0 Mg of C ha^{-1} for the balsam fir-white birch, balsam fir-yellow birch and sugar maple-yellow

birch bioclimatic domains, respectively. The expected average (site-level) carbon loss in total live and dead aboveground stocks, therefore, appeared to follow a latitudinal gradient from south to north: our results suggest that it would be lowest in boreal, more open-canopy stands dominated by black spruce, moss, ericaceous shrubs and lichens, intermediate in boreal balsam fir stands, and highest in balsam fir and sugar maple mixed and deciduous forests. The loss is likely driven by stand density and by growth, ingrowth, and mortality dynamics of vegetation in both corridors and control forests, which somewhat vary with latitude. These results also suggest that climatic conditions influence the magnitude of carbon losses primarily through their effects on forest productivity and stand structure, rather than through differences in corridor management.

As expected, initial deforestation and periodic vegetation management reduced most live and dead aboveground C stocks within corridors compared

with interior forests and corridor edges. Corridor management keeps the vegetation at a low height and prevents the ecological succession from progressing. In Quebec, about 70% of vegetation management operations consist of mechanical cuts (i.e., cutting trees and shrubs incompatible with the electrical network and preserving compatible species such as herbaceous and low shrubs), while the other practices involve spot chemical control of vegetation and combined mechanical-chemical methods (Létourneau et al. 2013). We also observed that the carbon pools of the understory vegetation were significantly higher in the corridor compared with the interior forest. Tree and tall shrub management within corridors likely provides more space and light to the understory vegetation, leading to higher biomass for this stratum compared to the interior forest.

Although carbon stocks of the organic FH layers were significantly lower in the corridors than in the edges and interior forests, total soil carbon pools (including both coarse and fine fractions of organic FH layers and mineral horizons down to a 55 cm depth) appeared not to be affected by the initial disturbance associated with the establishment of the power line corridor and by its periodic vegetation management. This held true across all studied power lines, which varied in age from old (80 years) to very young (<1 year). Deforestation for agriculture or urban development has been found to lead to a decline in soil carbon, but harvesting generally does not affect mineral soil carbon stocks (Nave et al. 2024), and has a transient effect on organic layer C stocks (Johannesson et al. 2025). The impact of the establishment and maintenance of power line corridors on soil carbon stocks is likely somewhere between that of deforestation and harvesting: the periodic vegetation management leaves ground vegetation and shrubs mostly untouched (Létourneau et al. 2013) but modifies the vegetation structure and composition enough to have a durable effect on the organic layers.

However, assessing management effects on soil carbon stocks can require large sample sizes due to their high spatial heterogeneity (Kravchenko and Robertson 2011; Stanley et al. 2023). A literature review on forest floor carbon stock changes observed that studies were unable to detect significant differences smaller than 15 to 20% because of their small sample sizes (Yanai et al. 2003). Based on our sampling

intensity and a power analysis using the Cohen's *d* test (Cohen 2016), we would only have been able to detect total soil carbon stock differences between transect positions superior to 34%. Consequently, the absence of detectable significant differences in soil carbon stocks between positions is not definitive; a higher number of observations would have been necessary to detect changes smaller than 34%.

Total ecosystem carbon stocks (including biomass, dead organic matter and soil) in power line corridors were quite similar between bioclimatic domains, hovering between 104 and 170 Mg of C ha⁻¹, 82 to 88% of which were soil carbon stocks. The loss of total ecosystem carbon stocks in corridors compared with interior forests was −38.7% on average (−27.3% for black spruce–moss, −34.7% for balsam fir–white birch, −56.3% for balsam fir–yellow birch and −36.7% for sugar maple–yellow birch domains). The absolute reduction in total ecosystem carbon storage was greatest in the boreal balsam fir domain, at 90 of C ha⁻¹, whereas the decrease amounted to 41 Mg of C ha⁻¹ in the black spruce–moss domain. The impact of the year of the power line establishment (which ranged from 1944 to 2024) and of their most recent vegetation management (from 2016 to 2023) (data provided by Hydro-Québec) was also examined on aboveground and total ecosystem carbon stocks (Figures S2 and S3 of Supplementary material). No significant trend was observed, as any temporal effect was likely overshadowed by bioclimatic differences. Indeed, the experimental setup was not designed to capture the influence of time. Nevertheless, the values for carbon drawdown caused by the presence of power lines represent averages broadly applicable at the landscape scale. For example, assuming an average corridor width of 42, 40, 64 and 109 m in the sugar maple–yellow birch, balsam fir–yellow birch, balsam fir–white birch and black spruce–moss domains (Table S1 of Supplementary material), the installation of 100 m of corridor represents a total carbon drawdown from aboveground live and dead biomass of respectively 27, 26, 39 and 37 Mg of C, and a total carbon drawdown from total ecosystem stocks of 33, 53, 58 and 45 Mg of C.

Our quantification of carbon stocks in corridors across a large bioclimatic gradient helps refine estimates of GHG emissions associated with the conversion of forests to settlements, such as power lines, in national GHG inventory reports. Our study

provides values for carbon drawdown at the bioclimatic domain level, which should eventually be replaced by finer categorizations when more detailed empirical data become available.

Yet, while the measured carbon drawdowns caused by the power lines are significant, they are likely less than what would be estimated by the current methodology used in Canada for its UNFCCC reporting of GHG emissions associated with the installation of power lines in forested lands (i.e., total removal of biomass of all merchantable trees, no further vegetation growth, and decay of dead organic matter over 20 years) (Environment and Climate Change Canada 2026), as live and dead aboveground carbon stocks remain non-negligible in corridors (i.e., 7.8 to 9.9 Mg of C ha⁻¹). Our results suggest that ecosystems within the corridors remain subject to vegetation and carbon dynamics, albeit along trajectories that are different from those observed in forests unaffected by the power lines. It is therefore an important aspect of further research, as it is not included in Canada's current GHG national inventory to the UNFCCC.

Edge effects on ecosystem carbon stocks

Our study suggested that the creation of a forest edge caused by the implementation of power lines does not significantly influence carbon stocks compared to the interior forests. The review of Smith et al. (2018) found a decrease of carbon stocks in edges of boreal ecosystems due to an increase of tree mortality and structural degradation, which could be partially compensated by an increased productivity from remaining trees. Yet, for temperate forests, Ziter et al. (2014) observed no edge effect on aboveground carbon stocks.

Edge effects are characterized by multiple gradients, such as microclimate, biogeochemistry, and resource availability, which vary in magnitude with distance from the edge (Harper et al. 2005; Smith et al. 2018). For instance, there should be greater exposure to light and wind near the edge. The conditions within the edges can be favorable or unfavorable to tree growth depending on the specific factors limiting productivity in a given ecosystem, and also on edge aspect (Biber and Pretzsch 2022; Smith et al. 2018). In our case, our studied forests varied from mostly coniferous, to mixed with a component of shade-intolerant hardwood and to moderately

shade-tolerant hardwood forests. Eleven of the power line corridors in our study were roughly oriented northeast-southwest, meaning their edges were either facing northwest or southeast, and the remaining seven were oriented northwest-southeast, with northeast or southwest-facing edges. However, there were no clear links between species composition and edge aspect and the resulting response of carbon stocks in edges.

Nevertheless, the edge effect on aboveground carbon stocks could be more subtle than what our experimental design could capture. Indeed, this design aimed to detect effects at the landscape level and only involved the comparison of forest edges and interior forests, respectively located at 20 vs. 60 m from the corridor opening, without strict control or corridor aspect. For example, a more intensive sister study performed on some of the same sites revealed that stand density and aboveground tree carbon stocks within 25 m of the corridor increased linearly with decreasing distance to the corridor, with the magnitude of the effect varying according to bioclimatic conditions (Harel et al. 2026). Similarly, Meeussen et al. (2021) detected an average 39% increase in total ecosystem carbon stocks, and a 95% increase in aboveground biomass, up to a distance of 4.5 m from the forest edge compared to interior stands in temperate deciduous forests across Europe. For their part, Buss et al. (2024) and Watkins et al. (2003) observed edge effects on plant composition only at the immediate edge (0 m) and up to a distance of 15 m. An investigation of forb and shrub communities in our study transects suggested that forb and shrub communities located 20 m from the corridor exhibited higher functional diversity related to biodiversity and habitat for pollinators than communities located either in the corridor or in interior forests (Roy Proulx et al. 2026). Alternatively, Yang et al. (2025) estimated that the effect of edges on aboveground biomass could go up to 235 m in temperate forests and 258 m in boreal forests, suggesting that what we considered interior forests might, in fact, still be under the influence of the corridor opening.

In our study, the quantification of tree biomass and carbon stocks was based on generic allometric equations developed for Canadian species (Lambert et al. (2005), which might have introduced a bias in the estimation of carbon stocks at forest edges. Indeed, tree allometric equations are typically built using data from productive stands and non-fragmented lands with a high proportion

of merchantable species (Boisvenue and White 2019; Tompalski et al. 2024). They might not properly capture the physiology of trees located near large openings. Using and adjusting DBH- and height-based allometric equations with field canopy measurements could potentially improve the accuracy and the detection of edge responses. Pairing our field data with remote sensing data could also improve the assessment and lessen specific biases and errors (Boisvenue and White 2019).

Conclusion

Our study provides one of the first field-based assessments of terrestrial carbon stocks associated with power line corridors across multiple forest bioclimatic domains. Our results suggested that, despite the initial disturbance and periodic vegetation maintenance in power lines on upland forests, upland ecosystems found within corridors are still important biogenic carbon stores (average of 108.6 ± 13.7 Mg of C ha⁻¹). While aboveground biomass of live trees and shrubs was, as expected, reduced in corridors, total soil carbon stocks were similar compared to those of forests located at the corridor edge (20 m from the corridor) and those of interior forests located 60 m from the corridor. Consequently, forest ecosystems with higher amounts of carbon stored in live biomass will be more impacted by the installation of a power line, while soils should be unaffected. Our results underline the fact that power line corridors in forested landscapes still maintain functioning, albeit anthropized, ecosystems.

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Author contributions Ariane Belleau: Data curation; Writing-Original Draft of the manuscript. Evelyne Thiffault: Funding acquisition; Writing-review and editing the manuscript. David Paré: Writing-review and editing of the manuscript. Jean-François Boucher: Writing-review and editing of the manuscript. Samuel Roy Proulx: Writing-review and editing of the manuscript. Grant Domke: Writing-review and editing of the manuscript.

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Data availability The datasets generated during the current study are available at: <https://doi.org/10.6084/m9.figshare.32660874>.

Declarations

Competing interests The authors have no relevant financial or non-financial interests to disclose.

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