



Mapping temperate forest stands using mobile terrestrial LiDAR shows the influence of forest management regimes on tree mortality

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ABSTRACT

With global change, forest trees will be exposed to increasing stress in the coming decades with various studies demonstrating that stress-related mortality will increase in forests. While tree death can be triggered by a single factor, it is often caused by the accumulation and the complex interaction of various stressors. Several silvicultural strategies have been developed to cope with global change but very few studies have addressed the ways in which silvicultural regimes interact with various stressors to influence tree mortality. This lack of research on the effects of forest management regimes on tree mortality may be due to the challenge of acquiring large (or long-term) datasets to assess tree mortality in forests. Within this context and using a mobile terrestrial LiDAR approach for rapid 3D-mapping of forest stands, we aimed to (i) compare recent tree mortality patterns in temperate forests among contrasting forest management regimes (even-aged silviculture, uneven-aged silviculture and unmanaged forests), and (ii) evaluate the relative influence of regeneration harvest severity on tree mortality compared to other spatially explicit factors (i.e., localized competition and slope position) and non-spatially explicit factors (i.e. tree DBH -diameter at breast height- and tree species group). In a permanent sample plot network, we mapped 15 508 dead and living trees (>9.1 cm DBH) within 37 sugar maple-dominated stands: 14 even-aged, 16 uneven-aged and 7 unmanaged stands. We separated the relative role of forest management on individual tree mortality from other factors such as size, species, slope position, and localized competition by modeling the probability of mortality for each tree. Localized competition or slope position were not significant factors describing tree mortality. Results showed that tree mortality was influenced by tree species, DBH and forest management regime. Models indicated that forest management regime, by itself, had a strong effect on tree mortality proportion. Results also indicated that trees in uneven-aged stands had a higher probability of dying than the those in even-aged or unmanaged stands. However, we do not advocate for the replacement of uneven-aged regimes in favour of even-aged ones. Instead, we believe that, in the context of global change, adjustments should be made to uneven-aged regimes to reduce risk of mortality. Moreover, our study reinforces the idea that silviculture can be applied in such a way as to enhance stand resistance and resilience to cope with global change; for example, by influencing size structure and species composition, which greatly influence tree mortality.

1. Introduction

Global change is modifying the dynamics of forest ecosystems through numerous and diverse environmental stressors (e.g., climatic, chemical, physical and anthropogenic) (Freedman, 2015). Unlike disturbances, such as fire and windthrow, which cause sudden mortality, stress-related mortality is an accumulative process (Holzwarth et al., 2013) linked to the injuries (Mangel and Bonsall, 2004) and stressors that occur during a tree's lifetime to adequately describe these dynamics

(Anderegg et al., 2015a). The use of long-term data is crucial to understand the processes driving stress-related mortality (Maringer et al., 2021). These stressors can impact the natural processes of forests in various ways: for example, growth decline triggered by drought (Camarero et al., 2015), decreases in plant establishment through lower nitrogen availability due to more intense and frequent fires (Stirling et al., 2019), forest composition shifts caused by species growth response to elevated CO₂ (Anderson-Teixeira et al., 2013), increases in seed production with rising spring temperatures (Caignard et al., 2017),

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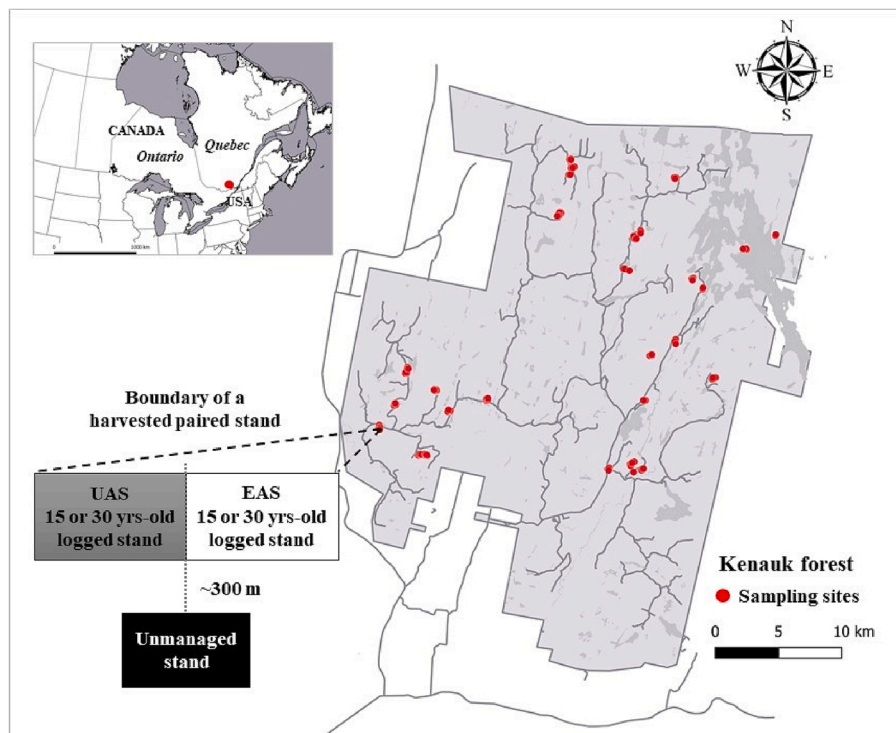


Fig. 1. Location of the study area, Kenauk forest (Quebec, Canada). The 37 stands were sampled within an area covering 25 000 ha. Harvested and unmanaged stands were at least 200 m × 250 m. The minimum distance between them was at least 300 m.

and elevated mature tree mortality (Allen et al., 2010) resulting from heat waves. Moreover, it has long been acknowledged that the complex interaction between these stressors is usually the cause of tree mortality in forests (Hartmann, 2008; Neely and Manion, 1991). Recently, some studies have reinforced Manion's (1981) idea that trees generally die from the accumulation of stress rather than from a single stressor (Anderegg et al., 2015b; Cailleret et al., 2017; Deschênes et al., 2019; Galiano et al., 2011).

In response to global change, foresters and forest ecologists have developed a variety of strategies to enhance forest resistance and resilience to increasing stress intensity, frequency and interaction (D'Amato et al., 2011). Such strategies focus on, for example, mixed plantations to spread risk among species (Guerra-De la Cruz and Galicia, 2017), assisted migration to establish species likely adapted to new climatic conditions (Gómez-Pineda et al., 2021), and commercial thinning to lower competition among trees for water consumption (Comeau, 2021). Most of these strategies aim to improve tree growth or regeneration via changes in (or by adapting) silvicultural methods, yet very few have addressed the ways in which silvicultural systems interact with various stressors to influence tree mortality. This is highly surprising given that tree mortality is expected to increase with global change as reflected by the vast literature on the subject (Allen et al., 2015; Hartmann et al., 2018; Neumann et al., 2017). The general absence of research on the effects of silvicultural systems on tree mortality may be in part due to the challenge of acquiring large (or long-term) datasets to assess tree mortality in forests (Guillemette et al., 2017; Holzwarth et al., 2013).

In North American northern hardwood forests, two of the most commonly used silvicultural systems are even-aged and uneven-aged silviculture (hereafter referred to as EAS and UAS) regimes. Forests developing under UAS are characterized by a mixture of individuals of all ages (from seedlings to mature trees) and of varying sizes (Majcen, 1994; Schuck et al., 1989). The predominant UAS regimes in northern hardwood-dominated forests involves applying different silvicultural treatments that promote this stand structure through repeated selection cuttings, thus favoring regeneration recruitment more frequently and

maintaining a permanent forest cover (Nolet, Kneeshaw et al., 2018). In contrast, within the EAS regime, age does not differ much amongst individuals; the silvicultural treatments promote the regrowth of a stand dominated by trees mostly of the same age. These treatments are often in the form of clearcuts, which remove the previous mature stand in a single harvest to bring it back to the regeneration stage (Nolet et al., 2018). Both predominant silvicultural regimes differ in severity and frequency of treatments, making any comparison between them difficult (Kuuluvainen et al., 2012). At the stand level, these regimes will influence species composition and forest structure. At the tree level, in a UAS regime, a dominant tree may have experienced irregular growth episodes given the temporal variations of available light and other resources. Thus, it is possible for a tree to go through multiple periods of *suppression and release* as demonstrated by Canham (1989) in unmanaged forests. Conversely, dominant trees in an EAS regime will more rarely go through such episodes. Therefore, the lifetime growth patterns of trees depend on the regime in which they develop, making it even more difficult to compare their resistance (e.g., as measured by tree mortality) to environmental stressors. In fact, few studies have actually compared forest resilience and resistance between these two regimes (O'Hara and Ramage, 2013).

Old, unmanaged forests are often viewed as natural benchmarks to compare with managed ecosystems (Amanzadeh et al., 2013), and are generally characterized by large and old trees, dead trees in different stages of decay and a greater overall amount of deadwood than managed ecosystems (Kirby et al., 1998; Spies, 2004). Given older unmanaged forests often have multi-cohort structures arising from a long history of partial canopy disturbance, UAS regimes are often viewed as a strategy to emulate these dynamics and structures (Kuuluvainen and Grenfell, 2012) in an attempt to maintain levels of ecosystem services associated with older forest conditions (Joelsson et al., 2017). Therefore, the use of UAS regimes as an integrative approach (Doerfler et al., 2017; O'Hara and Ramage, 2013) for achieving both timber production and conservation goals (Bollmann and Braunisch, 2013) has been increasing in recent decades. However, there are apparent conflicting objectives

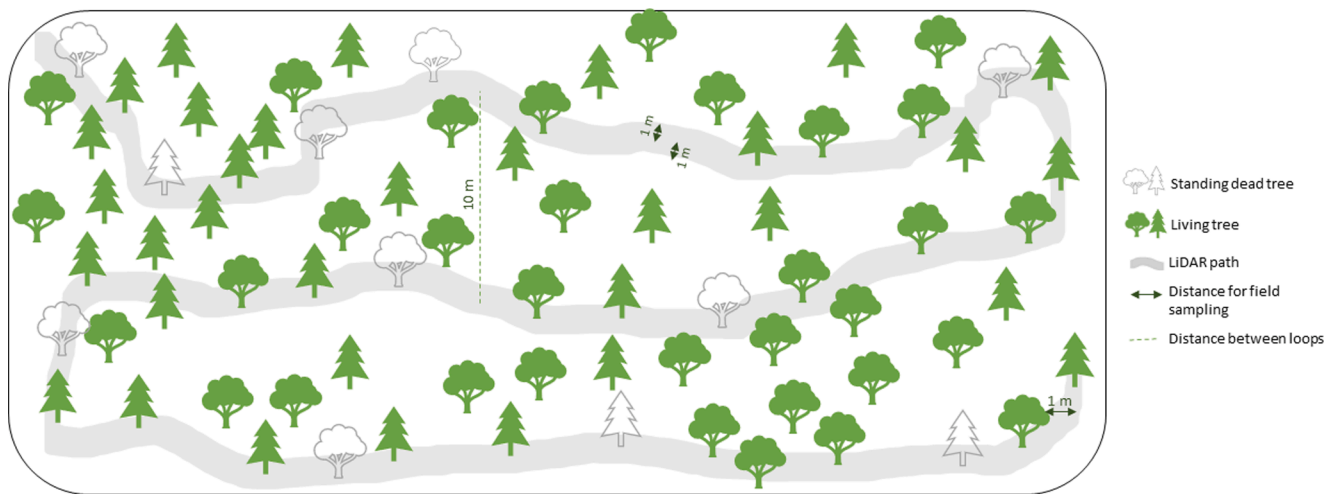


Fig. 2. Schematic representation of a mobile terrestrial LiDAR path in a stand. The path was delineated and flagged in each stand before circulating with the LiDAR sensor. The LiDAR path in grey represents the field sampling area.

between forest management and conservation (biodiversity) goals regarding tree mortality and associated deadwood recruitment. On the one hand, timber-focused management regimes aim to maximise wood production, which often leads to a decrease in deadwood volumes; on the other hand, ecological silviculture approaches focus on restoring and maintaining biodiversity by creating and retaining high levels of snags and coarse woody material (among other legacy features) in the ecosystem (Palik et al., 2020). These conflicting objectives regarding tree mortality in UAS regimes have largely been overlooked (but see Kenefic and Nyland, 2007).

In this study we first aimed to compare recent tree mortality patterns between contrasting forest management regimes (even-aged silviculture, uneven-aged silviculture and unmanaged forests); second, we aimed to evaluate the relative influence of regeneration harvest severity on tree mortality compared to other factors, either spatially explicit (i.e., localized competition and slope position) or non-spatially explicit (i.e., tree DBH and tree species). As spatially explicit factors require tree mapping, we used a mobile terrestrial LiDAR approach for rapid 3D-mapping of forest stands. Our expectations are that tree mortality probabilities will be higher (for a same DBH) in uneven-aged and unmanaged stands than in even-aged stands, largely owing to differences in tree growth history.

2. Materials and methods

2.1. Study area

Our study took place in the Kenauk forest (45°42'N, 74°53'W), a private forest territory that covers 25 000 ha, located in the southern part of the Outaouais region (Quebec, Canada) (Fig. 1). The area belongs to two bioclimatic domains: sugar maple-yellow birch and sugar maple-American basswood Majcen et al. (2003). Forest stand composition varies with soil characteristics and past disturbances, leading to a mixture of numerous species such as sugar maple (*Acer saccharum* Marsh.), yellow birch (*Betula alleghaniensis* Britton), American basswood (*Tilia americana* L.), American beech (*Fagus grandifolia* Ehrh.), balsam fir (*Abies balsamea* [L.] Mill.) and red oak (*Quercus rubra* L.) (Forget and Bouffard, 2006; Varin et al., 2015). Mean annual temperature is 4.8 °C, while mean annual precipitation is 985.5 mm (including 178.1 mm of snowfall and 807.4 mm of rainfall). From 1981 to 2010 the average number of degree-days above 0°C was 2888 (Environment Canada, 2018). The landscape is characterized by a varied relief with hills and valleys, altitude ranges between 200 and 300 m. Soils in the study area are chiefly well-drained Dystric Brunisols (USDA: Typic Dystrichrepts)

with moder-type humus and a loamy texture, which developed on glacial till deposits mainly composed of gneiss, quartzite and granite (Forget and Bouffard, 2006; Lajoie, 1967; Soil Classification Working Group, 1998). There is little documentation of the details of forest management in the Kenauk forest prior to 1985–1990 but given prevailing forest conditions, we can assume that it was very similar to other portions of the Quebec northern hardwoods: high grading and diameter-limit cuts (Nolet et al., 2008). As such, we assumed the Kenauk forest had been harvested several times with these types of selective cuts from the beginning of the 18th century to the end of the 20th. According to Forget et al. (2006), from the early 1980 s to the early 2000 s, strip cutting became the main silvicultural approach used on the territory, with clearcutting in strips (about 60 m wide) and selection cutting in between these strips (about 60 m wide). This approach led to the development of numerous side-by-side EAS- and UAS stands. It is worth mentioning that 30–35 years ago, Quebec foresters were at the very beginning of the implementation of uneven-aged management as a forest management regime, with selection methods gaining in popularity in response to abusive selective cuts and clearcuts. While the first guidelines (Majcen et al., 1990) for sugar maple-dominated forests emphasized residual DBH structure (J-shaped) and basal area (approximately 20 to 35 % of the basal area was removed) as well as improving tree vigor (Majcen et al., 1990), it is impossible to know how strictly those guidelines were followed in the Kenauk forest. Also, there are stands in the Kenauk forest that have not been managed for 50 years or more. Not managing these forests is not considered a management regime *per se*, but in our study we consider it as a regime within Kenauk forest management regimes. Henceforth, the term “forest management regime” (FMR) will include even-aged silviculture, uneven-aged silviculture, and unmanaged stands.

2.2. Experimental design

We sampled 23 sites located within a permanent sample plot network that was established in 2016 to monitor forest health and development. The sampled sites comprised seven unmanaged stands (where the last selection cutting occurred 50 years ago or more and are no longer managed); 14 harvested paired stands consisting of an EAS stand (clearcut) adjacent to UAS stand selection cut); and 2 additional UAS stands (within the study area). Among the harvested paired stands, 6 were logged about 15 years ago, as were the two additional UAS stands, and the other 8 paired stands were logged about 30 years ago according to Kenauk’s records. Therefore, the EAS stands had trees mostly of the same age (15 or 30 years old) and DBH class as shown in Fig. A.1, some

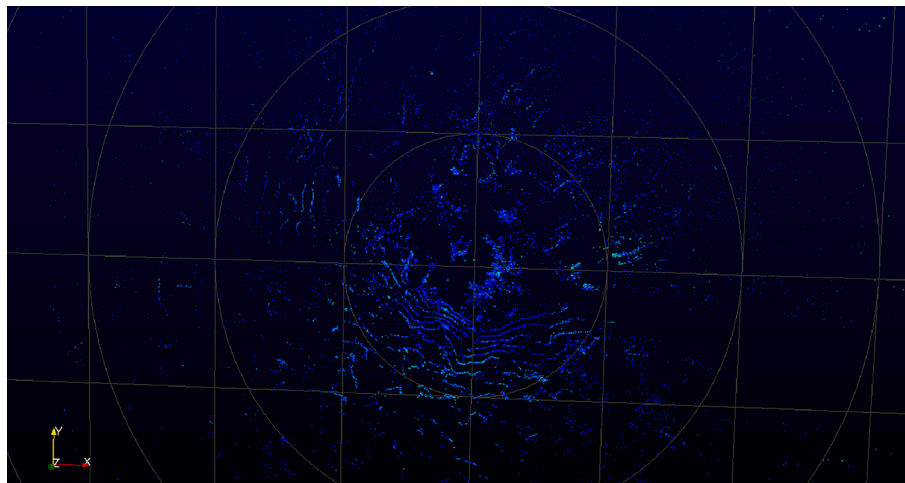


Fig. 3. Raw data (not spatially organized) captured by LiDAR in Kenauk forest (Quebec, Canada). A frame of one instant displayed on VeloView (by Velodyne LiDAR, version 3.5.0).

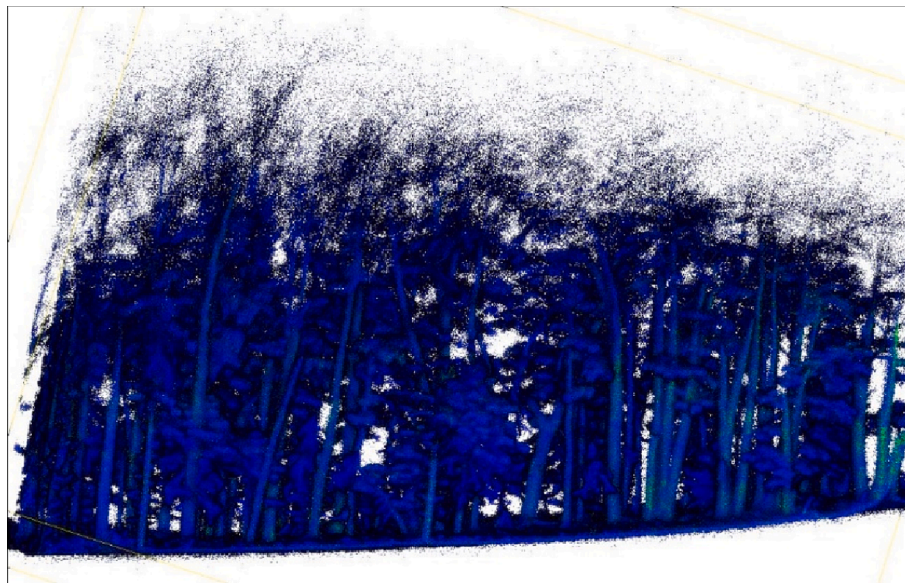


Fig. 4. LiDAR frame spatially organized by the SLAM (Simultaneous Localization and Mapping) algorithms developed by Outsight®. The picture extracted from CloudCompare (version 2.11.3) shows the 3D point cloud map finalized and ready for use in determining tree-level attributes.

Table 1
Description of 6 independent variables for the logistic regression analyses.

Independent variables	Description	Model
Forest management regime	3-level categorical variable: Even-aged, Uneven-aged, Unmanaged forests	Spatially explicit/ Non-spatially explicit
Forest management regime/time	5-level categorical variable: Even-aged-15yrs, Even-aged-30yrs, Uneven-aged-15yrs, Uneven-aged-30yrs, Unmanaged	Spatially explicit/ Non-spatially explicit
Tree DBH	Continuous variable	Spatially explicit/ Non-spatially explicit
Competition index	Continuous variable	Spatially explicit
Slope position	4-level categorical variable: Lower slope, Middle slope, Upper slope, Ridge	Spatially explicit
Tree species group	5-level categorical variable: Beech, Conifers, Intolerants, Mid-tolerants, Sugar maple	Non-spatially explicit

of these stands had older (and larger) residual trees because there was no market for some species. Based on stand structure, we estimated that there were 10 residual trees per hectare immediately after harvesting. No other treatments were performed in the EAS stands (such as pre-commercial thinning) and they all developed from natural regeneration. The intentions of the forest managers for these even-aged stands remain unclear. The UAS stands had trees of different ages as represented by their DBH distribution (Fig. A.1). All the UAS stands were subjected to a single selection cut (performed uniformly across stands rather than in gaps according to our observations) no matter if the treatment was performed 15 or 30 years ago. Before this treatment and as already mentioned, these stands underwent several episodes of exploitive partial cuts in the last centuries. Unmanaged stands were randomly distributed in the study area and served as a benchmark for comparison with harvested paired stands (mean stand area was 0.50 ha and minimum distance to paired stands was 300 m (Fig. 1).

Table 2

Characteristics of 37 stands scanned in Kenauk forest (Quebec, Canada) as a function of forest management regime and time since last logging. Data extracted from mobile terrestrial LiDAR. * Considering all standing dead trees (mortality codes 1 to 4) ** Considering only trees recently dead (codes 1 to 3). Values in parentheses are standard deviation of the mean.

Forest management regime	Time since logging (yrs)	Number of scanned stands	Mean area, ha	Mean of basal area, m ² •ha ⁻¹	Mean DBH (cm)	Number of standing dead trees detected*	Total number of trees detected (dead and alive)	Percentage of recent mortality*	Percentage of most recent mortality**
Even-aged	15	6	0.45 (0.18)	18.6 (4.01)	13.5 (0.80)	42	3111	1.35	0.90
Even-aged	30	8	0.44 (0.10)	23.4 (9.95)	14.1 (0.80)	80	4724	1.69	1.26
Uneven-aged	15	8	0.44 (0.10)	23.5 (2.65)	16.8 (1.40)	142	2328	6.10	2.24
Uneven-aged	30	8	0.51 (0.16)	23.5 (7.81)	18.9 (2.23)	119	2940	4.12	1.67
Unmanaged	>50	7	0.63 (0.15)	23.0 (4.63)	19.8 (2.16)	209	2405	8.69	5.10

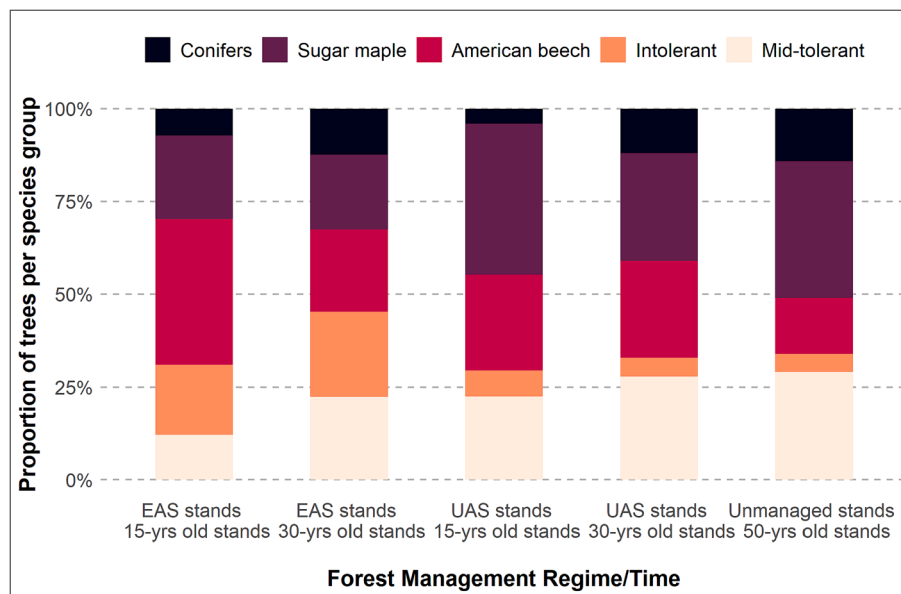


Fig. 5. Proportion of trees per species group as a function of Forest Management Regime and time since logging based on conventional tree inventory taken while circulating on the path with the LiDAR sensor. Kenauk forest (Quebec, Canada).

2.3. General approach and rationale

To achieve our objectives, we localized and mapped every living and standing dead tree in each stand. To do so, we used a mobile terrestrial laser scanning technique (LiDAR) complemented by a more conventional tree inventory (field sampling). Thousands of trees per hectare can be found in some stands; mapping all these trees using conventional methods could take weeks for a single stand. Hence, this two-fold approach allowed us to carry out a large-scale forest inventory and to describe the surroundings of each tree (i.e., coordinates, neighbouring vegetation, slope position). This study is the first to our knowledge that uses mobile terrestrial LiDAR to fully inventory natural forests to examine ecological research questions. An overview of the different steps of the methodology is explained in the following sections and schematically represented in Fig. C.1.

2.4. Forest inventory

To scan the forest trees, we walked in each stand carrying a VLP-16 (PUCK™ from Velodyne) LiDAR sensor mounted on a modified backpack above the user’s head (to avoid blocking the laser beam) (Fig. 2). The VLP-16 provides a full 360-degree environmental view to deliver accurate real-time 3D data (Kidd, 2017). The data captured by the LiDAR were recorded using the VeloView (by Velodyne LiDAR, version

3.5.0) software running on a field-computer connected to the VLP-16 sensor. A voice recording file was synchronized with the LiDAR recording. Indeed, as the VLP-16 sensor does not have a built-in GPS, the exact location of the standing dead trees was later obtained by matching the timestamp of the voice recording with the timestamp of the LiDAR data, both taken while walking in the stands.

Before circulating through the stand with the LiDAR sensor, all the standing dead trees were localized and flagged. This was followed by delineation of a path within each stand to cover most of the stand area, and to get as close as possible to previously identified standing dead trees to facilitate their identification in the 3D point cloud map. The path included three loops within each stand, keeping a maximum distance of 10 m between two loops (Fig. 2) to limit occlusion (a tree hidden by another one) (Pierzchała et al., 2018). The mean distance walked in each stand was 400 m (SE). While circulating on the path with the LiDAR sensor, we identified all tree species and manually measured the DBH of all living and standing dead trees within one meter of the mobile terrestrial LiDAR user (Fig. 2). We also visually assessed and classified the mortality status of each standing dead tree within the one-meter distance based on level of decomposition (e.g., crown integrity, bark decomposition, presence of branches) to provide an approximate time since the death of the tree; a four-category classification code was used, 1 to 4, with 4 corresponding to trees with the highest decomposition as explained in Table B.1). While LiDAR is not yet able to perform species

Table 3

Comparison of spatially and non-spatially explicit models explaining the variation in the probability of tree mortality in 37 forest stands in Kenauk forest for all tree mortality codes (Quebec, Canada). Comparison based on the Akaike information criterion (AIC). The number of parameters k , the ΔAIC and the Akaike weight (w_i) of each model are shown. Site was considered as a random effect.

	Model	Fixed variables	k	AIC	ΔAIC	w_i
Spatially explicit models (a)	S14	FMR/time \times DBH \times CI	21	4751.86	0	0.965
	S10	FMR/time \times DBH	11	4758.56	6.706	0.034
	S13	FMR \times DBH \times CI	13	4767.17	15.32	0
	S11	FMR \times DBH \times Slope-pos	25	4768.56	16.703	0
	S9	FMR \times DBH	7	4770.15	18.292	0
	S12	FMR/time \times DBH \times Slope-pos	41	4771.1	19.248	0
	S8	DBH \times CI	5	4836.7	84.843	0
	S7	DBH \times Slope-pos	9	4859.68	107.822	0
	S2	DBH	3	4873.8	121.941	0
	S6	FMR/time	6	4934.52	182.669	0
	S5	FMR	4	4935.16	183.304	0
	S15	Slope-pos \times FMR	13	4942.67	190.815	0
	S16	Slope-pos \times FMR/time	21	4951.85	199.996	0
	S4	CI	3	5021.26	269.407	0
	S1	Intercept	2	5082.12	330.267	0
S3	Slope-pos	5	5085.88	334.03	0	
Non-spatially explicit models (b)	N7	Sp \times DBH \times FMR/time	51	4615.12	0	0.804
	N6	Sp \times DBH \times FMR	31	4617.94	2.821	0.196
	N5	Sp \times DBH	11	4753.3	138.176	0
	N3	Sp \times FMR	16	4842.44	227.321	0
	N4	Sp \times FMR/time	26	4845.12	229.998	0
	N2	Sp	6	5014.42	399.3	0
	N1	Intercept	2	5082.12	467.003	0

identification, our manual field sampling along the LiDAR path allowed us to measure and identify about 25 % of all the living trees per stand.

2.5. LiDAR data processing

Raw LiDAR data consists of a series of spatially disorganized frames (a frame being a picture of points at a given time: Fig. 3). To spatially organize the frames into a usable 3D point cloud, where geometric shapes can be recognized (Fig. 4), we used the SLAM solution (Simultaneous Localization and Mapping) developed by OutSight®. Once the 3D point cloud map was obtained, further analyses were performed to gather the digital elevation model (DEM) of each stand, to detect the trees within the stands, and to determine their DBH and spatial coordinates. The SLAM also allowed for reconstruction of the spatial location of the path followed by the sensor during the “inventory”, which was subsequently used later to map the location of standing dead trees.

For each stand we extracted the DEM from the point cloud using the package “lidR” (Roussel et al., 2020) on R 3.5.2 (R Core Team, 2020). It was then converted into slope positions using the Topographic Position Index to obtain four categories of landform (i.e., ridge, upper, middle and lower slope) (Jenness, 2006) which were then assigned to each tree (based on their spatial coordinates).

Tree detection and their DBH calculation and spatial coordinates were obtained using various successive filters and algorithms in the CompuTree platform (ONF, 2010; Othmani et al., 2011). Even though

CompuTree was able to automatically detect many of the trees and to calculate their DBH, a visual inspection of the 3D point cloud maps indicated that i) some trees were not detected, ii) CompuTree detected trees that did not exist and iii) the DBH of some detected trees was poorly estimated. Consequently, a visual inspection/validation of the 3D point cloud maps was needed to detect missing trees and to measure their DBH, to eliminate false tree detections, and to verify the DBH estimation of the trees provided by CompuTree. The visual inspection was performed using QGIS (Las Palmas 2.18) with embedded R scripts to facilitate both 2D and 3D map visualization of the point clouds (Nolet et al., 2022).

2.6. LiDAR- and field sampling-based tree metrics

For standing dead trees, while their DBH, species and mortality code were recorded during the field sampling, their spatial coordinates were obtained by matching the timestamp of their mention in the audio recording with the timestamp of the LiDAR path. For living trees, DBH and spatial coordinates were determined in various ways. For trees that were correctly detected and DBH correctly calculated by CompuTree, we used the DBH and spatial coordinates provided by CompuTree. In cases where the difference between the DBH measured during field sampling and the one obtained with CompuTree was low ($\pm 10\%$), no correction was performed. When this difference was high ($> \pm 10\%$), we kept the spatial coordinates obtained through CompuTree and we manually measured the DBH from the 3D point cloud map using QGIS. For trees not detected by CompuTree, both DBH and spatial coordinates were obtained through QGIS. We examined the correlation between the diameters manually measured (QGIS) from the 3D point cloud map and those calculated by CompuTree and found a systematic overestimation. As a result, a correction factor of 0.8975 was applied for those DBH, Fig. D.1.

2.7. Competition indices

There are a variety of competition indices that have been proposed that differ in terms of variables used to quantify degree of competition experienced by a given tree (species, DBH, height, distance, angle, etc.). Here, we used the general equation provided by (Hegyi, 1974) and modified this general form to include numerous competition indices (cf. Lorimer, 1983) reflecting the nature of tree-tree interactions. The Competition Index (CI) is formulated as:

$$CI_{ik} = \sum_{j=1}^n \frac{DBH_j^x}{DBH_i^x \times L^y} \quad (1)$$

where i is the target tree, j represents the competing tree, L_{ij} is the distance between trees i and j , n is the number of competing trees in a radius of k meters. Four radii were used in our calculations: 4, 6, 8 and 10 m. We attributed three values to \times (0, 1 and 2) to assess the degree of size-asymmetry of competitive interactions (cf. Schwinning and Weiner, 2012); attributing the value 0 provides the same weight ($DBH^0 = 1$) to each competing tree, whereas value 2 indicates size-asymmetric competitive interactions with larger competitors having a disproportionate effect on resource availability. Similarly, we attributed three values (0, 1 and 2) to y to assess the effect of various weights to the distance between trees; attributing the value 0 is equal to not incorporating a distance effect. We also performed the same calculations in which the size of the target tree was not considered:

$$CI_{ik} = \sum_{j=1}^n \frac{DBH_j^x}{L^y} \quad (2)$$

A first set of competition indices were computed using Equations (1) and (2) using all the competing trees in the radius considered and a second set was computed for only competing stems larger than the target tree. Hence, a total of 144 competition indices were computed for each dead and living tree.

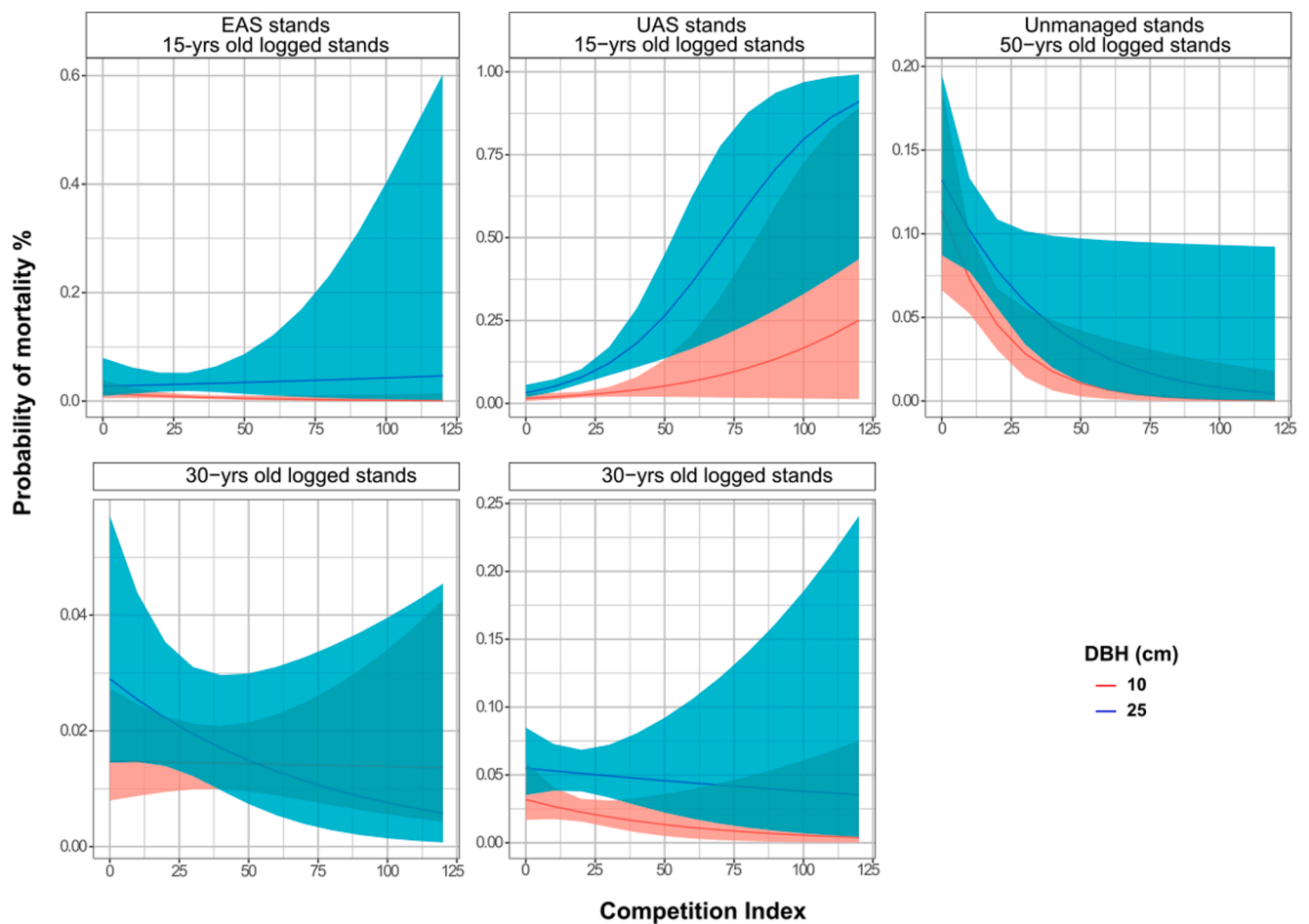


Fig. 6. Marginal effects of Competition index, DBH and Forest Management Regime/time since harvesting (EAS-and UAS stands logged 15 or 30 years ago and unmanaged stands) on tree mortality percentage in the Kenauk forest. The shaded area represents the 95% confidence interval around the fitted values.

2.8. Statistical analyses

We used logistic regression to compare recent tree mortality (binary variable) between forest management regimes and to evaluate the relative influence of forest management practices on mortality probability relative to other extrinsic spatially explicit and intrinsic factors, with site as a random effect (to account for the non-independence of data). We would have ideally run a single analysis using all explanatory variables (Table 1); however, this was not possible given the lack of species identification for the living trees detected in the point clouds (this only existed for the dead trees); for the living trees, species identification was only available for those recorded during the field manual sampling. As a result, we had to run two sets of analyses (Table 1), one using (among others) the competition index and slope position variables (spatially explicit variables) and another one using (among others) the species variable (non-spatially explicit variable). Each set of analyses was performed on two datasets, keeping all living trees (i.e., 14 916), and including either (1) all dead trees (mortality codes 1 to 4, i.e., 592 standing dead trees) or (2) trees recently dead (mortality codes 1 to 3, i.e., 299 standing dead trees).

A) Spatially explicit analyses

In the first set of analyses, we included all the explanatory variables, except the tree species group, applying either one-way, two-way, or three-way interactions. Therefore, the forest management practices variables (i.e., forest management regime: 3-level categorical variable and forest management regime/time: 5-level categorical variable), extrinsic spatially explicit factors (i.e., CI: continuous variable and slope

position: 4-level categorical variable), and tree DBH (continuous variable) were included as fixed effects (Table 1).

B) Non-spatially explicit analyses

Non-spatially explicit analyses could have been run using only the trees recorded in the field; however, choosing this option would have strongly limited our ability to compare the relative importance of spatially and non-spatially factors on mortality probability due to the use of different data sets for each analysis (i.e., AIC between two analyses cannot be compared when performed using different datasets). This problem was overcome by attributing a random species to the alive stems identified in the point clouds (the species of dead trees is known for dead trees). Based on the proportions of each species per diameter classes (i.e., 9–19, 19–29, 29–39, 39–49, 49–59, >60 cm) within each stand, we randomly attributed a species to each living tree in our database. For example, if sugar maple represented 40 % of the trees in the 10–19 cm diameter class in a given stand (LiDAR path), then 40 % of the living trees of that same diameter class were characterized as sugar maple in the stand. To remove uncertainty arising from this procedure, we performed this routine 100 times to create 100 random datasets; a similar approach was used by Royer-Tardif et al. (2017). However, when we assign a random species to a specific tree, we cannot assign the species to the spatial coordinates of the tree, which is why the competition and slope position variables could not be used in this analysis. Hence, in the second set of analyses, we included only the forest management practices variables (i.e., forest management regime and forest management regime/time) and intrinsic factors (i.e., tree DBH and tree species group: 5-level categorical variable) (Table 1). The explanatory

Table 4

Comparison of spatially and non-spatially explicit models explaining the variation in the probability of recent tree mortality in 37 forest stands in Kenauk forest for tree mortality codes from 1 to 3 (Quebec, Canada). Comparison based on the Akaike information criterion (AIC). The number of parameters k , the Δ AIC and the Akaike weight (w_i) of each model are shown. Site was considered as a random effect.

	Model	Fixed variables	K	AIC	Δ AIC	w_i
Spatially explicit models (a)	S10	FMR/time \times DBH	11	2845.8	0	0.571
	S14	FMR/time \times DBH \times CI	21	2847.5	1.705	0.243
	S9	FMR \times DBH	7	2849.11	3.31	0.109
	S11	FMR \times DBH \times Slope-pos	25	2850.52	4.723	0.054
	S13	FMR \times DBH \times CI	13	2852.21	6.413	0.023
	S12	FMR/time \times DBH \times Slope-pos	41	2866.46	20.662	0
	S7	DBH \times Slope-pos	9	2868.34	22.542	0
	S8	DBH \times CI	5	2873.75	27.955	0
	S2	DBH	3	2879.6	33.803	0
	S5	FMR	4	2917.35	71.552	0
	S6	FMR/time	6	2917.39	71.596	0
	S15	Slope-pos \times FMR	13	2929.09	83.289	0
	S4	CI	3	2933	87.207	0
	S16	Slope-pos \times FMR/time	21	2939.81	94.009	0
	S1	Intercept	2	2953.67	107.87	0
S3	Slope-pos	5	2957.97	112.17	0	
Non-spatially explicit models (b)	N6	Sp \times DBH \times FMR	31	2754.4	0	0.997
	N7	Sp \times DBH \times FMR/time	51	2765.89	11.491	0.003
	N5	Sp \times DBH	11	2807.94	53.542	0
	N3	Sp \times FMR	16	2876.46	122.063	0
	N4	Sp \times FMR/time	26	2883.44	129.046	0
	N2	Sp	6	2935.2	180.808	0
	N1	Intercept	2	2953.67	199.27	0

variables were included as one-way, two-way, or three-way interactions.

For both sets of analyses, our general hypothesis was that tree mortality probability would be related to the interaction between multiple factors associated with tree-level vigour, level of resource competition, and severity and frequency of harvesting activities. Hence, to avoid data dredging and testing all possible models (Franklin et al., 2000), we chose to test a priori models. These models included one-way or two- and three-way interactions among tested variables, because the influence of FMR practices on tree mortality may depend on the intrinsic or extrinsic factors, but no additive model was tested. Model comparisons were based on the Akaike's information criterion (AIC, Akaike, 1998). Concerning the second set of analyses, all tested models were run successively on each random dataset ($n = 100$) and AIC was computed for each model. We then computed the mean AIC of a model based on its 100 replicates.

Time since logging could not be tested alone because it does not have the same biological meaning when applied to a selection cut or a clear-cut as it implies a "reset" in the latter case. In our models, we then either used forest management regime alone (FMR) or with time (FMR/time).

Finally, we compared the spatially and non-spatially explicit models; this comparison was possible because there were the same number of trees (dead and alive) in both sets of analyses. The AIC ranking and the parsimony principle were used to choose the best model(s) to explain the probability of tree mortality (Franklin et al., 2000). We used R 4.0.0 for all the statistical analyses (R Core Team, 2020).

3. Results

3.1. Overview

A total of 14 916 live trees and 592 standing dead trees were detected and mapped within the 37 stands (Table 2). The mean area – by forest management regime and time since logging – covered by the mobile terrestrial LiDAR ranged from 0.44 ha (even-aged-30) to 0.63 ha (unmanaged). Average DBH varied from 13.5 cm (even-aged-15) to 19.9 cm (unmanaged) while the mean basal area varied from 18.6 m²•ha⁻¹ (even-aged-15) to 23.5 m²•ha⁻¹ (uneven-aged-30). The number of standing dead trees was the lowest (42) in young EAS stands and the highest in the unmanaged stands (209) (Table 2). Our conventional tree inventory (field sampling) also indicated that species composition varied as a function of forest management regime and time since last logging with the main difference being more intolerant species and less sugar maple in EAS stands than in the other forest management regimes (Fig. 5).

3.2. Model comparisons

Spatially explicit analyses.

Probability of mortality was best explained by the three-way interaction between Forest Management Regime/time, DBH and CI (model S14), based on AIC and associated model weight ($w_i = 0.965$) (Table 3 (a), model S14). Nevertheless, once the variable "Competition Index" (CI) was plotted (Fig. 6), we realized that its effect on the probability of mortality did not appear biologically plausible, especially for the EAS and the unmanaged stands (more neighbouring competition resulted in less tree mortality), suggesting a possible overfitting in the model (Fig. 6). The second-best model (model S10) considered only the two-way interaction between Forest Management Regime/time and DBH. Given the two best approximating models contained these two variables, we interpreted this to imply they had the most important effects on tree mortality probability. The role of CI appeared unclear and relatively weak.

Probability on recent tree mortality (codes 1–3) was best explained by the two-way interaction that considered FMR alone instead of FMR/time (Table 4 (a), model S9). Across all models, DBH and FMR were clearly the most important variables to explain tree mortality proportion; however, adding the component "time" to FMR appeared less informative (4 more parameters between models S10 and S9). Overall, for spatially explicit analyses, DBH, FMR (or FMR/time) were the most important variables to explain tree mortality proportion regardless of the whole tree mortality (codes 1 to 4) or most recent tree mortality (codes 1 to 3) were considered. The use of spatially explicit variables (CI and slope position) did not improve (or very weakly) the models.

Non-spatially explicit analyses

Based on the AIC, model N7 (three-way interaction between tree species group, Forest Management Regime/time and DBH) appeared as the best model to explain mortality proportion (codes 1 to 4) ($w_i = 0.804$). However, adding the component "time" to FMR was questionable, as it used 20 more parameters than model N6 for only a slight change in AIC (2.8 units) (Table 3 (b)). For recent tree mortality probability (codes 1 to 3), the best approximating model was clearly model N6 (three-way interaction between tree species group, Forest Management Regime and DBH), with a w_i of 0.997 and the use of 20 parameters fewer than the second-best model (Table 4 (b), model N6). In both analyses, the variables of tree species group, DBH and FMR were the most important variables to explain tree mortality proportion for the non-spatially explicit analyses.

Comparing spatially and non-spatially explicit models for the whole tree mortality (codes 1 to 4) showed that non-spatially explicit models outperformed spatially explicit models (improving the AIC by more than 140 units, Fig. 7) highlighting the importance of including the tree species group variable in the model more than competition indices or slope position. A similar scenario was observed for recent tree mortality probability (codes 1 to 3); the AIC of the best non-spatially explicit model outperformed

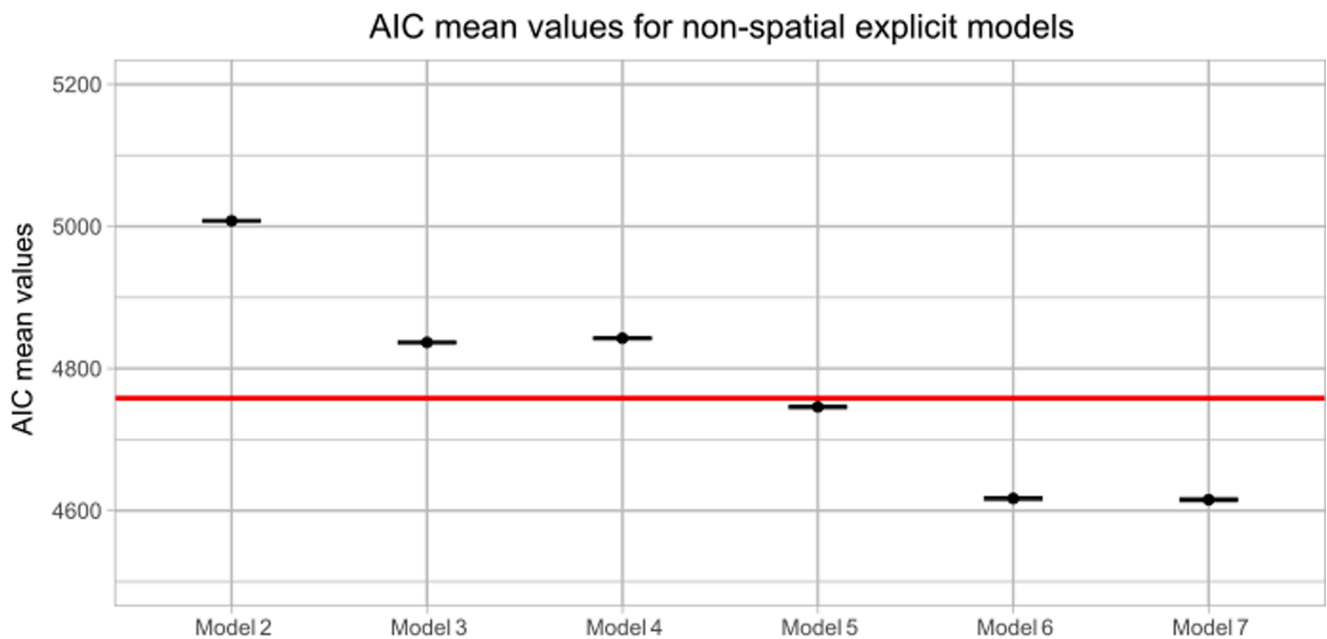


Fig. 7. AIC mean values for non-spatially explicit models explaining the variation in the probability of tree mortality in 37 forest stands in Kenauk forest for the whole tree mortality (codes 1 to 4) (Quebec, Canada). The red line represents the AIC for the best spatially explicit model (mortality codes 1 to 4, Table 2.3).

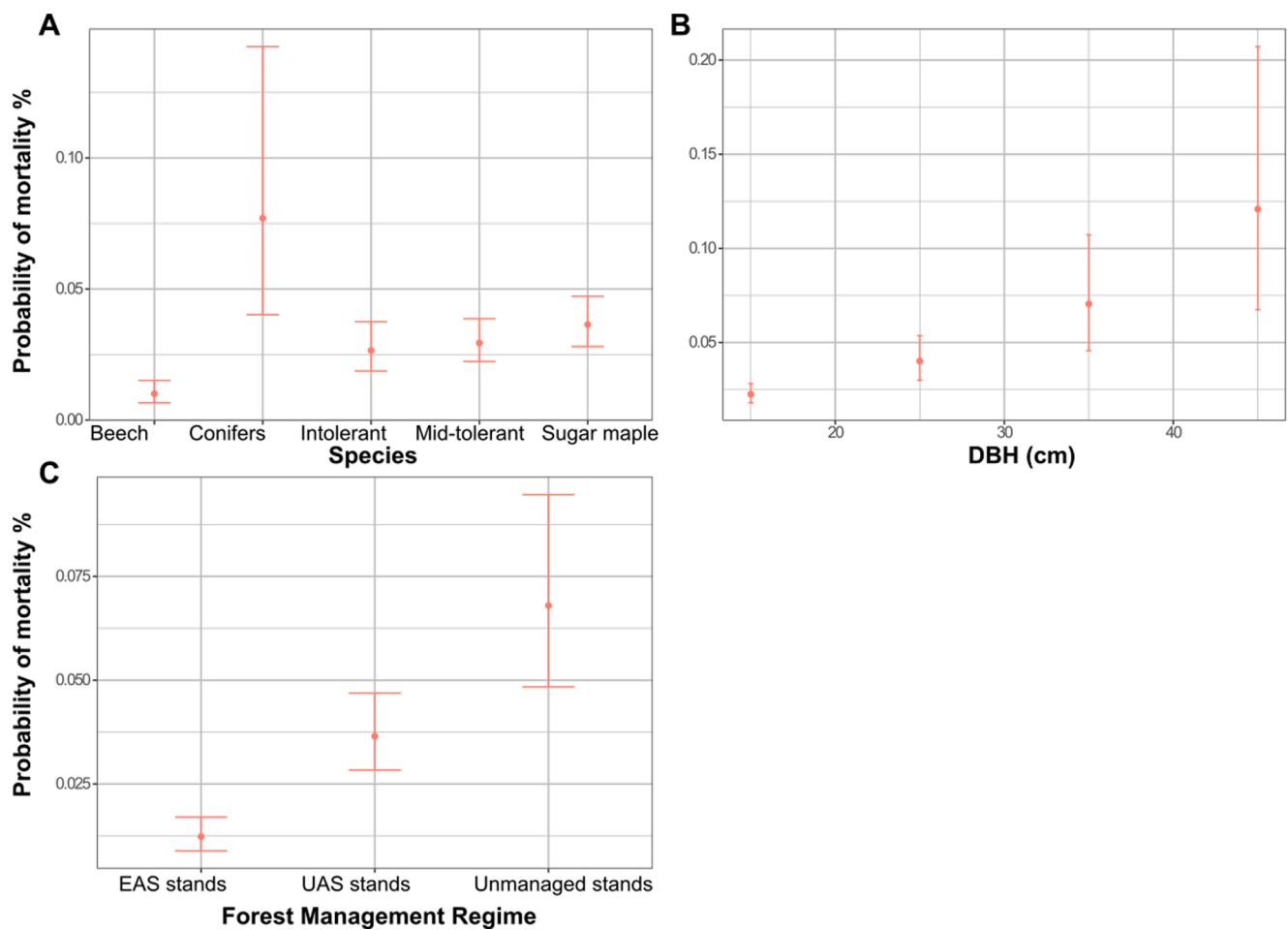


Fig. 8. Single effects of (A) tree species group, (B) DBH and (C) Forest Management Regime on tree mortality probability in 37 forest stands in Kenauk forest for all tree mortality codes (1 to 4) (Quebec, Canada). The 95% confidence interval is shown around the predicted values. Single effects were computed based on one of the 100 random datasets (see section 2.4 Forest inventory).

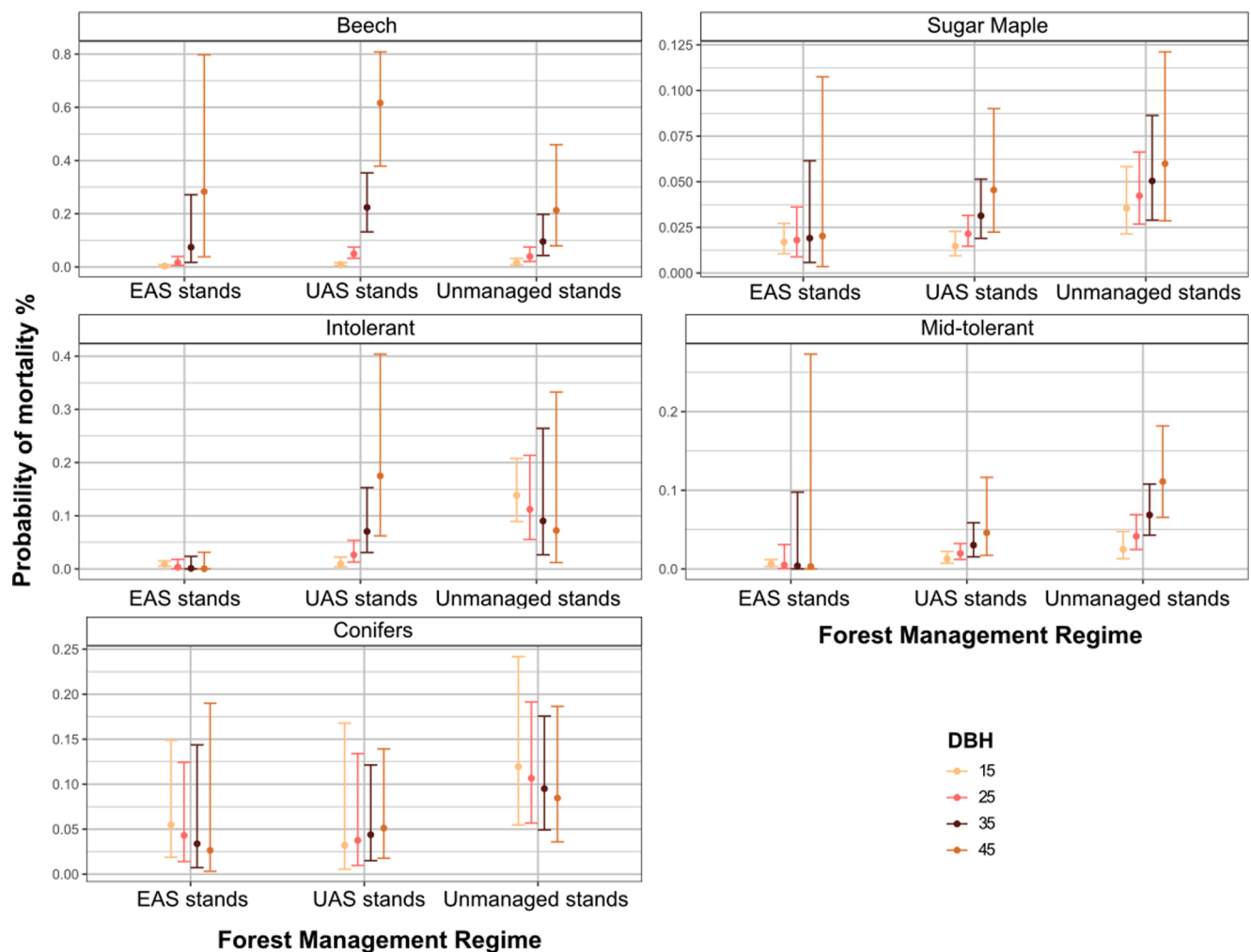


Fig. 9. Marginal effects of Forest Management Regime, DBH and tree species group on tree mortality probability in 37 forest stands in Kenauk forest for all tree mortality codes (Quebec, Canada). The 95% confidence interval is shown around the predicted values. Marginal effects were computed based on one of the 100 random datasets (see section 2.4 Forest inventory).

the best spatially explicit model (AIC improvement = 91.4 units, Table 4).

Finally, comparisons between the best approximating model (model N6, three-way interaction between tree species group, FMR and DBH) and a corresponding model that did not include FMR (model N5) indicated that FMR by itself, had a strong effect on tree mortality proportion (i.e., not only through its influence on species composition and DBH). To confirm the validity of the random species attribution approach for the non-spatially analyses, we performed regression analyses on the dataset of living and dead trees identified during the manual field sampling (for which there is no uncertainty for species). The results as shown in Table E.1 are very similar to those using the random species attribution approach and indicate that the model including the variables forest management regime in interaction with species and DBH is the most efficient model to predict tree mortality probability.

3.3. Influence of tree- and stand-level variables on mortality probabilities

Taking into consideration all models and the AIC comparison (Fig. 7), we found that tree mortality proportion was best explained by the three-way interaction between tree species group, DBH and Forest Management Regime (by model N6, Table 3 (b)). In this section we explored, according to this model, the marginal effects (once the random effect (site) was removed) of these variables on tree mortality probability.

First looking at one-way effects, EAS stands generally showed a lower mortality proportion 0.01 (95% C.I. 0.01–0.02) than UAS 0.04

(95% C.I. 0.03–0.05) stands and unmanaged stands 0.07 (95% C.I. 0.05–0.09) (C, Fig. 8). Among tree species groups, Beech generally showed the lowest mortality proportion 0.01 (95% C.I. 0.01–0.02) while Conifers and Sugar maple showed the highest proportion at 0.08 and 0.04, respectively (95% C.I. 0.04–0.14, 0.03–0.05) (A, Fig. 8). Mortality proportion also generally increased with DBH (B, Fig. 8). While these one-way effects are quite simple, their interactions are more complex.

Sugar maple, Beech and Mid-tolerant tree species groups showed similar patterns of mortality proportion as a function of FMR and DBH (Fig. 9); the main difference being that DBH effect is steeper for Beech, as mortality probability shifted from near 0% for smaller DBH classes to as high as 50 % for larger DBH, as shown in Fig. F.1. For these three tree species groups and for a specific DBH class, mortality proportion was generally higher in UAS and unmanaged stands than in EAS stands. For the Intolerant tree species group, mortality proportion was higher in UAS and unmanaged stands than in EAS stands. While mortality proportion increased with DBH in UAS stands, no such relationship was observed in EAS and unmanaged stands (Fig. 9). For the Coniferous tree species group, which showed the lowest abundance among tree species groups (Fig. 5), mortality probability appeared the lowest in the UAS stands among the FMRs. No clear relationship between DBH and mortality proportion was observed for this tree species group (Fig. 9).

4. Discussion

4.1. Forest management regime effects

Our study showed that tree mortality probability was influenced by tree species, DBH and forest management regime. While several studies have already demonstrated the influence of tree species (Deschênes et al., 2019; McDowell et al., 2011) and size (Deschênes et al., 2019; Van Mantgem et al., 2009) on tree mortality in the hardwoods of North America (Fortin et al., 2008; Guillemette et al., 2017) and other forest types (Negrón-Juárez et al., 2018; Neumann et al., 2017), our study highlights the effects of the forest management regime. As forest management regimes can influence both species composition (Martin et al., 2014) and tree size distribution within a stand, any observed effects of forest management regime on tree mortality may seem trivial. However, our analyses showed that, even after considering the species and DBH effects, the forest management regime still impacted tree mortality probability; for example, according to our results, there was a higher probability (almost twice) of a 15 cm DBH sugar maple dying in UAS stands than in EAS stands. These two forest management regimes were compared using an adjacent paired stand design, which ensures that the differences observed between EAS and UAS stands are not site-dependent, thus ensuring the robustness of our results.

Although we did not study the reasons why tree mortality varies as a function of the forest management regime, some explanations can be put forward. The reasons why a tree dies (severe disturbances, such as fire, excluded) can be grouped under three main headings: i) senescence (Gill et al., 2015), ii) competition (Das et al., 2011; Peet and Christensen, 1987), and iii) miscellaneous stressors (Anderegg et al., 2015b; Neumann et al., 2017). In our study, senescence can be considered negligible as very few trees (0.71 %) in our stands had a DBH of over 50 cm. As for competition, our results did not show a strong relationship (if any) with competition indices. It is possible that our indices failed to capture the complex phenomenon of competition; however, competition - or at least recent competition - does not seem to be the main factor responsible for the difference in tree mortality probability in EAS and UAS stands. This leaves us with the third possible explanation: miscellaneous stressors. In contrast to trees in EAS stands, trees in UAS stands undergo a series of suppression and release throughout their lifespan (Canham, 1985). In our UAS stands, trees experienced multiple partial cuts, the last one being a selection cut and the previous ones being selective cuts. While this growth rate fluctuation can be considered as a stress in itself (due to the tree making continuous adjustments), it also means, for example, that a tree with a DBH of 20 cm in UAS stands has a high probability of being much older than a tree of the same size in an EAS stand. Because they are older, these trees are also more likely to have undergone several episodes of environmental stress, such as drought or insect defoliation, within their lifespan. Moreover, trees in UAS stands have a higher probability of being injured by forest operations. In brief, trees in UAS stands may be subjected to more stresses and for a longer period of time than trees in EAS stands, increasing their likelihood of reaching a threshold where any additional stress would cause mortality. Such impacts of the cumulative effects of previous stress making trees more susceptible to a recent stress have been demonstrated in similar forests (Hartmann et al., 2008; Moreau et al., 2019).

4.2. Uneven-aged versus unmanaged forests

As it has been proposed by various authors that an UAS regime emulates natural disturbance in forests (O'Hara and Ramage, 2013); mortality patterns across DBH classes and species could be expected to be similar in UAS and unmanaged stands. In our study, this was only true for Beech, whereas Sugar maple and Intolerant species showed a much steeper DBH mortality relationship in UAS stands than in unmanaged stands, and Mid-tolerant species, as well as Conifer species, showed much higher mortality probabilities in unmanaged than in UAS stands. Hence, the natural disturbance emulation feature that is often attributed

to uneven-aged management (more specifically to tree-selection) must be put into perspective. According to our results, mortality patterns in UAS stands are closer to those in unmanaged stands than those in EAS stands; however, UAS and unmanaged stands cannot be considered the same in terms of mortality dynamics across species.

4.3. DBH and species effects

The influence of tree size and tree species on the probability of mortality was also confirmed by our results. Independent of the tree species, tree mortality probability increased in our stands as the DBH increased. Nevertheless, the DBH-mortality relationship differed among species, as observed in other studies (Neumann et al., 2017). Beech, for example, had a very low probability of mortality when its DBH was under 35 cm, but it drastically increased when its DBH exceeded 35 cm. The beech bark disease (BBD - necrotic fungal infection caused by the exotic beech scale insect *Cryptococcus fagisuga* Lindinger; Houston, 1975; Nolet et al., 2015) can only partially explain this drastic change as the "arrival" of BBD in the region is quite recent (Roy and Nolet, 2018). Moreover, according to our personal observations, Beech stems of over 40 cm in DBH were relatively rare in the region before the arrival of BBD. For the Intolerant group, the probability of mortality decreased with increasing size. This can be easily explained by the shade intolerance of these species where smaller trees are more likely to be shaded by taller trees. Furthermore, the relatively high mortality rates observed for coniferous species in our study appear to be related to the high mortality rates of Balsam fir (a short-lived species) in unmanaged forests and to hemlock remnants in EAS stands.

4.4. Benefits related to the approach and caveats

The mobile terrestrial LiDAR allowed us to carry out a massive tree inventory in a very short time compared to other techniques (e.g., manual inventory). In addition to delimiting the Lidar path and clear branches or other obstacles, it took approximately 30 min to scan each site. The outcome is a 3D-map database with all trees geolocated and a detailed digital elevation model. This enabled us to identify the most important variables in tree mortality in managed forests using contrasting approaches; for example, DBH appears to be a more important variable than competition for predicting tree mortality. Nevertheless, mobile terrestrial LiDAR data treatment (extraction of all metrics) is time consuming, involving a great deal of additional manual work, half a day was needed to extract all metrics from areas averaging roughly a half hectare. While the approach is currently being improved, human input might always be necessary. Moreover, species identification cannot be performed directly with LiDAR yet; developments in this area are needed if any significant progress is to be made, as species is often a key variable in forest studies. In this study, the EAS stands were relatively young, the oldest originating from a 30-year-old clearcut. Ideally, older EAS stands should be sampled, but such stands were not available in the study area. The long-term monitoring of the stands mapped in the current study should allow us to verify whether the differences we observed in tree mortality probability patterns between even- and uneven-aged silviculture remain the same as EAS stands get older. While our study provides insights into mortality patterns as a function of forest management regimes and many other variables, further studies are needed to identify the exact causes of tree mortality. Accordingly, we plan to carry out a study on the growth patterns (e.g., Cailleret et al., 2017) prior to death of the dead trees identified in the current study.

4.5. Forest management implications and conclusion

For this study, we did not have access to the detailed information regarding the method for tree selection applied in the UAS stands, and therefore it is hard to evaluate how our results can be extrapolated to all UAS regimes in American north hardwood stands. However, UAS stand

management varies highly depending on the jurisdiction, and it can even evolve over time within a given jurisdiction, making it impossible to perform a study that will apply to all its variants. It remains that certain conclusion drawn from our study, such a higher stress level experienced by trees in UAS stands compared to EAS stands and the limitations of considering UAS as an emulation of natural disturbance regimes, may apply to a certain degree in other forests managed under uneven-aged regimes.

Even though we observed a higher mortality probability in UAS stands than in EAS stands, we do not advocate for the replacement of uneven-aged regimes in favour of even-aged ones. Instead, we believe that, in the context of global change, adjustments should be made to the management of UAS systems. We suggest that because a substantial part of the observed mortality was stress-related, and that stressors are expected to diversify and increase with global change (Lindner et al., 2010), tree health monitoring between rotations should be included in UAS management plans to meet expected yields. Should observations from such monitoring reveal signs of a decrease in tree health and vigor, cutting cycles could be shortened to minimize tree death between harvests (by harvesting unhealthy trees). The effect of past stressors (Bre-shears et al., 2009), as well as longer periods of suppression and release in tree growth (Canham, 1985) are now referred to as the ecological memory of trees (Ogle et al., 2015). Such differences in the ecological memory of trees have been reported to affect growth and mortality (Itter et al., 2019; Pretzsch, 2021), and we suggest that mortality probability was greater in the UAS compared to EAS stands due to a greater accumulation of stressors and longer periods of suppression and release.

In the American northern hardwoods forests and broadleaved forests in Europe, the trade-off between timber production and tree mortality-induced biodiversity (snags and coarse woody debris) objectives is mainly achieved via UAS regimes (Doerfler et al., 2017; Joellsson et al., 2017; Kuuluvainen and Grenfell, 2012). In view of our results, this practice makes sense as mortality patterns in the UAS stands were more similar to those in the unmanaged stands compared to the EAS stands. However, the timber production objectives of UAS regimes could be threatened if global change- and stress-related mortality were to increase (as suggested by our results) and negatively impact stand growth. Finally, by showing that contrasting silviculture treatments, EAS vs UAS, influence tree mortality, our study reinforces the idea that silviculture can be applied in such a way as to enhance stand resistance and resilience to cope with global change, for example, by addressing DBH structure and species composition, which greatly influence tree mortality.

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CRediT authorship contribution statement

Rebeca Cordero Montoya: Investigation, Formal analysis, Methodology, Writing – original draft. **Anthony W. D’Amato:** Supervision, Writing – review & editing. **Christian Messier:** Funding acquisition, Writing – review & editing. **Philippe Nolet:** Project administration, Supervision, Conceptualization, Methodology, Writing – review & editing.

Declaration of Competing Interest

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.

Data availability

Data will be made available on request.

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Appendix A

Diameter Distribution Within Kenauk Forest.

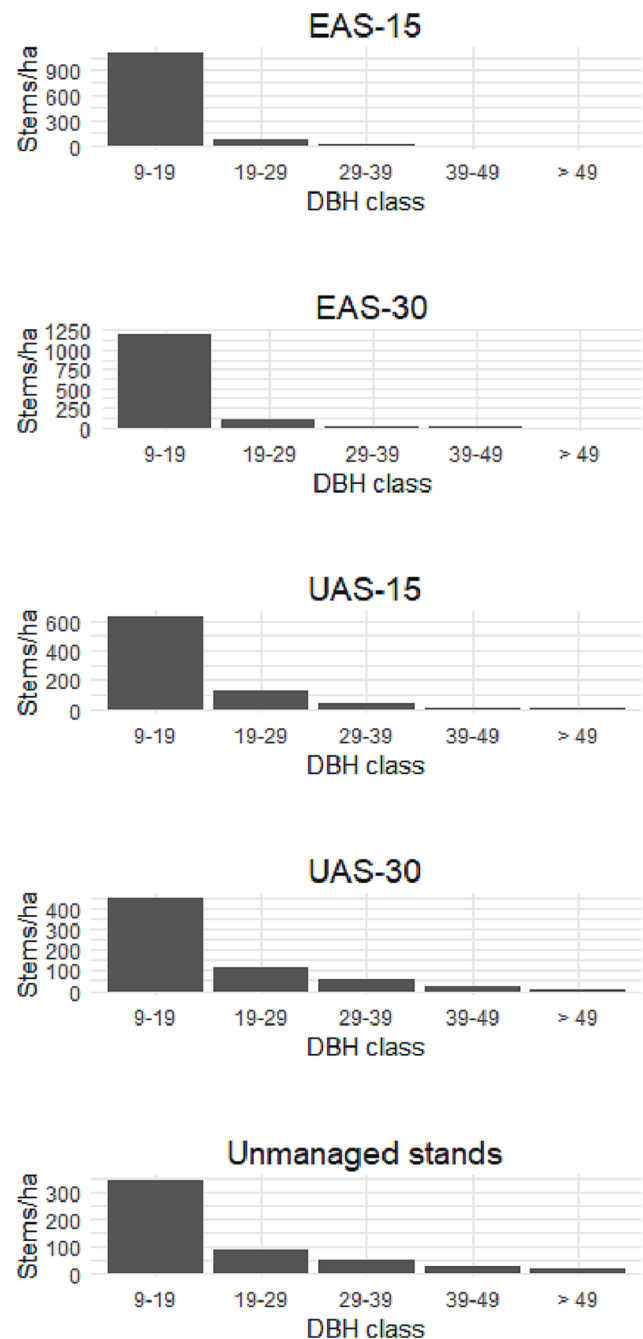


Fig. A.1. Diameter class distribution in Kenauk forest for 37 forest stands divided by type of forest management regime and time since harvesting. Even-aged stands (EAS) and uneven-aged (UAS) stands logged 15 or 30 years ago, and unmanaged stands (Quebec, Canada).

Appendix B

Vigor Code For Standing Dead Trees – Visual Estimate.

Table B1

Visual assessment composed of four categories: the first category appeals to the signs of a very recent death while the fourth category encompasses characteristics of a death that has occurred for several years (decayed tree) (Boulet, 2005 adapted by Hartmann, 2008).

Vigor codes for standing dead trees	Description
1: Recently dead	-small twigs are still visible-bark and top are intact
2: Intermediate	-no twigs, but presence of branches-bark and crown are present but begin to deteriorate
3: Intermediate	-no visible twigs-some main branches are still apparent-crown is broken-bark is missing or detached from the tree
4: Long dead	-no branches (with some exceptions), just the trunk-almost no bark-trunk is decomposing (soft)-tree height is significantly reduced

Appendix C

Schematic Representation of the Two-Fold Forest Inventory Approach.

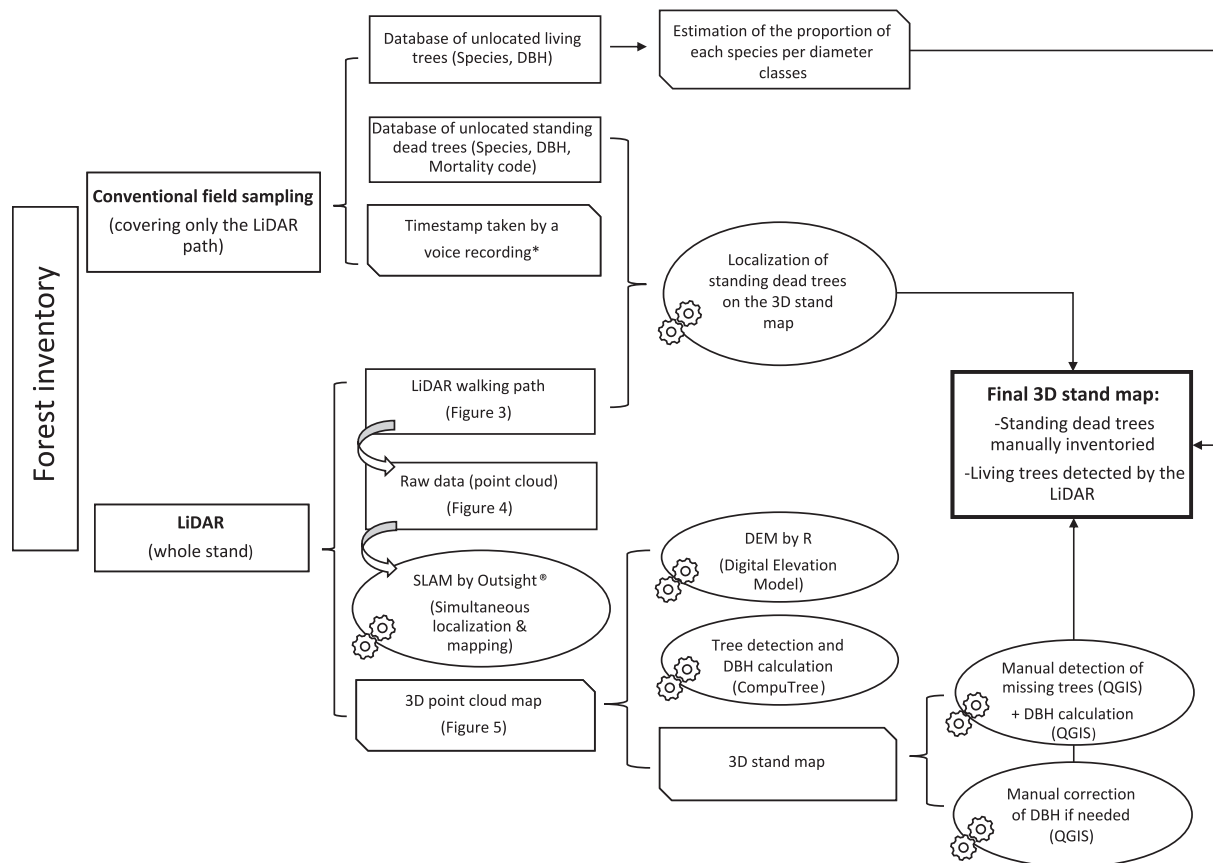


Fig. C.1. Schematic representation of the two-fold forest inventory approach used to map all living and standing dead trees in a forest stand, as well as the calculation of their basic metrics (e.g., DBH, slope position). Simple rectangles represent data collection, rectangles with diagonal corners snipped represent processed data, ovals represent digital processes.

Appendix D

Correction Factor for the Diameters Overestimated By Computree.

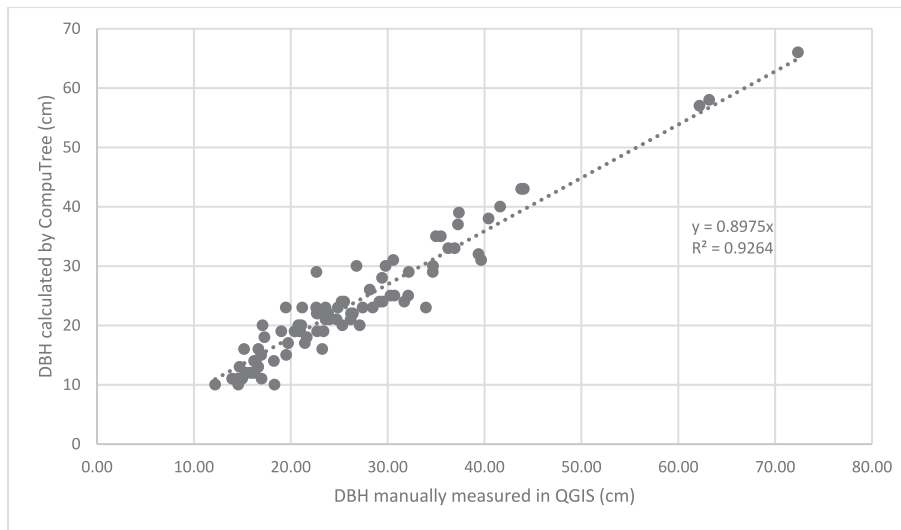


Fig. D.1. Correlation between the diameters manually measured by QGIS from the 3D point cloud map and the ones automatically calculated by CompuTree.

Appendix E

Aic Mean Values – Manual Field Sampling Database.

Table E1

AIC mean values, based on manual field sampling, for non-spatially explicit models explaining the variation in the probability of tree mortality in 37 forest stands in Kenauk forest for the whole tree mortality (codes 1 to 4) (Quebec, Canada).

Model	AIC
Sp × FMR × DBH	3712
Sp × FMR/time × DBH	3721
Sp × FMR	3750
Sp × FMR/time	3758
FMR	3785
FMR/time	3787
Sp	3805
DBH	3813

Appendix F

Single Effects of Tree Species Group and DBH on Tree Mortality Probability.

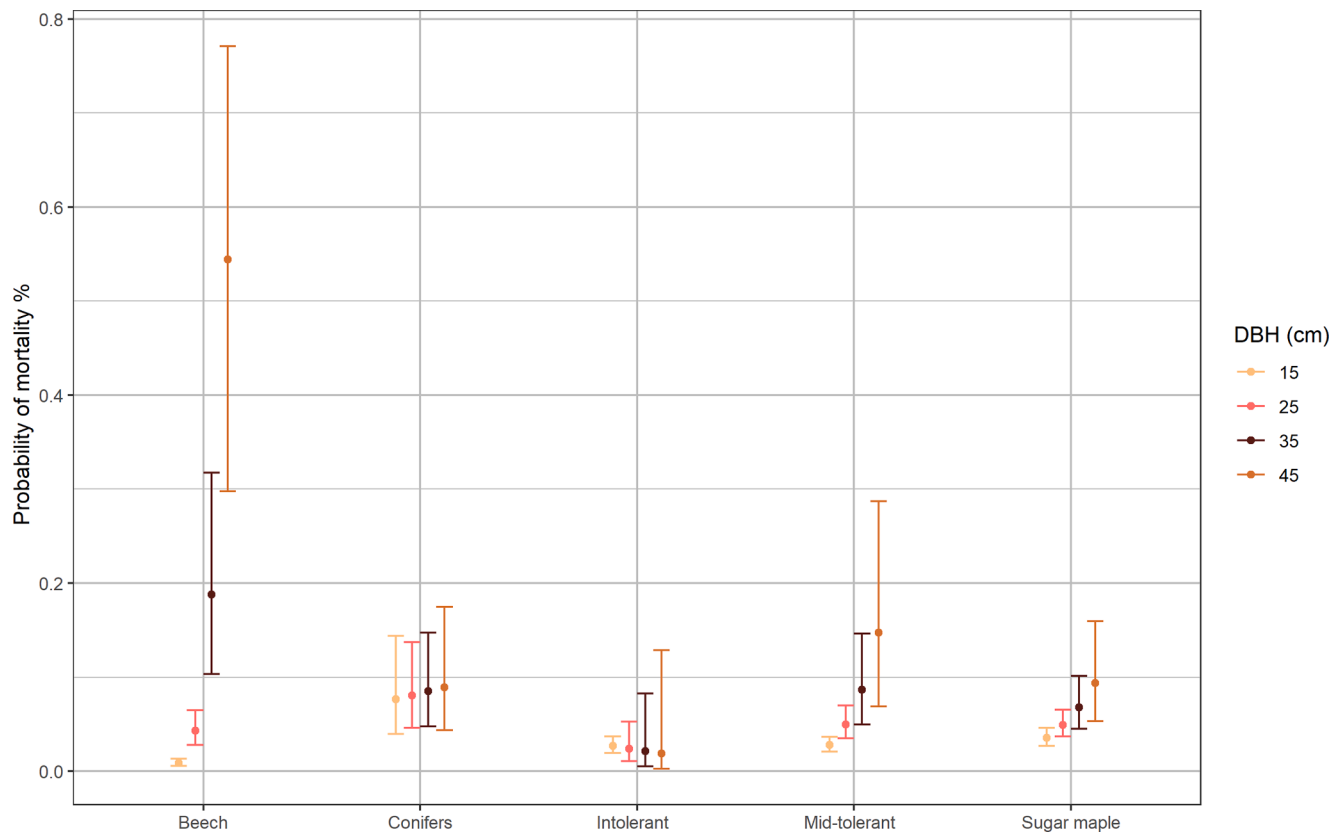


Fig. F.1. Single effects of tree species group and DBH on tree mortality probability in 37 forest stands in Kenauk forest for all tree mortality codes (1 to 4) (Quebec, Canada). The 95% confidence interval are shown around the predicted values. Single effects were computed based on one of the 100 random datasets (see section 2.4 Forest inventory).

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