**Additional carbon sequestration potential of abandoned agricultural land afforestation in the boreal zone: a modelling approach.**

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Abstract

Agricultural land abandonment is a widespread phenomenon that generally results in C accumulation due to natural establishment of woody vegetation. However, whether afforestation of abandoned agricultural lands (AAL) can sequester more C than AAL naturally reverting to woodlands is unclear. In this study, we used the CBM-CFS3 model to compare the additional C sequestration potential of afforested AAL (AR) with a reference scenario where AAL naturally returns to forest (NR). Simulations were performed for stands located in Quebec's boreal zone (Canada) on podzolic soils. The AR scenario corresponded to stands afforested at a density of 2000 trees $ha^{-1}$ with one of five commonly planted species in the region, namely black spruce (BS), white spruce (WS), eastern white pine (EWP), jack pine (JP) and tamarack (TK). The NR scenario corresponded to stands naturally reverting to natural forests of one of five species naturally occurring in the region, namely BS, WS, balsam fir (BF), trembling aspen (TA) and white birch (WB). The yield tables used for NR were phased out by 5, 10, 15 and 20 years to simulate various dynamics of colonization by woody plants following agricultural abandonment. Net C accumulation in AR stands varies depending on the planted species, from 127 to 255 Mg C $ha^{-1}$ after 100 years with TK and WS, respectively. Net C accumulation in NR stands after 100 years ranges from 82 to 145 Mg C $ha^{-1}$ for BS and TA, respectively, but these values are sensitive to both tree density and colonization time following abandonment. In both scenarios, the soil C pool shrinks by 6 to 12 Mg C $ha^{-1}$ over the first 80-85 years, but the following soil C stock replenishment is faster in AR than in NR stands due to higher litter production (up to 50 Mg C $ha^{-1}$). The additional C sequestration potential of AAL afforestation, i.e. the difference in net C accumulation between AR and NR stands, is the highest in stands afforested with WS and reaches a peak of 121-175 Mg C $ha^{-1}$ 80-87 years after afforestation depending on the NR scenario. Afterwards, additional C sequestration decreases due to a reduction in plantation growth combined with increasing natural regeneration growth. This modelling approach helps predict AAL additional C sequestration potential and suggests that AAL afforestation yields a more rapid C sequestration than natural regeneration, which could contribute to reaching net-zero emissions by 2050.

*1. Introduction*

To limit global warming to a maximum of 1.5 ° C above pre-industrial levels, it is necessary to implement measures to achieve net-zero emission by 2050 (IPCC, 2018). Land use change contributes to climate change, and forest restoration and management are thus increasingly viewed as important mitigation tools (IPCC, 2019). Large scale afforestation is considered to be an inexpensive and easy way to capture a large fraction of anthropogenic $CO_2$ emissions (Humpenoder *et al.*, 2014; Bastin *et al.*, 2019, but see Friedlingstein *et al*. 2019 ). As part of its plan to achieve net-zero emissions by 2050, the Canadian government pledged to plant 2 billion trees by the end of the decade (Environment and Climate Change Canada, 2019). Many factors nevertheless influence plantations' C sequestration potential, such as location and planted species.

According to the good practice guidance for land use, land-use change and forestry (IPCC, 2003), the additional C sequestration of a plantation can be estimated by calculating the difference between: 1) net C accumulation in both the vegetation and the soil following afforestation minus life cycle emissions occurring prior to planting (project scenario); and 2) the C that would have been accumulated on the same land in the absence of a plantation (baseline scenario). As C emissions related to all afforestation operations account for a small fraction of the biological balance (Gaboury et al., 2009), the additional C sequestration potential of plantations is likely to be higher in unproductive lands where tree natural regeneration has stopped or is slow. For instance, the afforestation of open woodlands (OW), a stable unproductive state of the Canadian boreal forest (Jasinski and Payette, 2005; Girard *et al.*, 2008) has the potential to sequester ~77 Mg C $ha^{-1}$ over a period of 70 years (Gaboury *et al.*, 2009) and to offset up

to 8% of all Québec's industrial process GHG emissions after 45 years (Boucher *et al.*, 2012). Recent studies suggest that boreal OW's C sequestration potential could be even higher by selecting appropriate species (Fradette *et al.*, 2021) or by selecting high potential OW (Dufour *et al.*, 2016). The implementation of OW afforestation can however be technically difficult due to their remoteness, relative poor accessibility and the necessary intensive mechanical site preparation prior to planting (i.e., soil scarification).

In contrast, afforestation of abandoned agricultural lands (AAL) may be easier to implement. First, AAL are easily accessible relative to remote boreal forests, reducing GHG emissions associated with the establishment and management of plantations; second, they do not require such intensive site preparation (e.g., scarification), reducing soil C emissions to the atmosphere; and third, tree plantations in AAL provide other services such as crop protection against wind, frost and soil erosion, as well as increased water conservation and biodiversity (Grala, 2004; Grebner *et al.*, 2013; Climate Action Initiative, 2013). Agricultural land abandonment is a worldwide phenomenon (Campbell *et al.*, 2008; Prishchepov *et al.*, 2021). Although it has several social and economical detrimental consequences, AAL also induced significant net C accumulation in several regions of the world (Vuichard *et al.*, 2008; Urbano and Keeton, 2017; Bell *et al.*, 2021). The natural reversion to natural grasslands or woody lands is the most commonly observed pattern, which accumulates significant amounts of C at minimal cost and provides several other positive environmental benefits (Bell *et al.*, 2020; Huang *et al.*, 2020). Some studies have found that a mix of active and passive management practices yield high rates of C sequestration in the soil (Bell *et al.*, 2020) although higher rates can

be reached with plantations, depending on location and planted species (Laganière *et al.*, 2010; Li *et al.*, 2012). Active management such as land restoration by planting is less commonly practiced but can be more efficient depending on the objectives. Studies have found that afforested AAL can accumulate larger amounts of C than naturally regenerated AAL, particularly in the aboveground biomass (Tremblay and Ouimet, 2013; Voicu *et al.*, 2017). Tremblay and Ouimet (2013) have shown that afforestation results in higher C sequestration than naturally regenerated stands in Quebec, Canada, suggesting that the afforestation of abandoned agricultural lands has the potential to create additional C sinks. However, these studies are scarce compared to those documenting soil C gain relative to cultivated lands (Paul *et al.*, 2002; Laganière *et al.*, 2010; Li *et al.*, 2012; Shi *et al.*, 2013; Kampf *et al.*, 2016; Bell *et al.*, 2020).

The goal of this study was to assess additional C sequestration potential of AAL afforestation in the eastern balsam fir-yellow birch domain, of Quebec, Canada, where 4500 ha were classified as abandoned farmland in 2016 (MAPAQ, 2016). We used the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) (Kurz *et al.*, 2009) to simulate the long-term C sequestration dynamics in both afforested AAL (afforestation scenario, AR) and AAL that would naturally revert to woody lands (baseline scenario, NR) to estimate the additional C sink that afforestation may create. Several vegetation dynamic scenarios were used for both AR and NR to assess the potential variability in plantations' additional C sequestration at different time and spatial scales, and thus help management practice decisions in national climate change mitigation plans.

## *2. Materials and Methods*

### *2.1. CBM-CFS3*

We used the Carbon Budget Model of the Canadian Forest Sector (CBM-CFS3) (Kurz *et al.*, 2009) to simulate C stock changes in the five pools of forest ecosystems (IPCC, 2003) − namely aboveground biomass (AGBM), belowground biomass (BGBM), litter, deadwood, and soil organic matter (SOM) − in afforested and naturally regenerated AAL (afforestation and natural regeneration scenarios, respectively) over the course of a period of 100 years. CBM-CFS3 uses merchantable volume yield ($m^3$ $ha^{-1}$) tables to derive stand-level biomass C increments with stand age in each of the above- and belowground tree biomass C pools tracked by the model (Stinson *et al.*, 2011). Annual turnover rates are specified for each of the above- and belowground biomass and dead organic matter pools, and depend on the selected ecological boundary (Kull *et al.*, 2016). Deadwood, litter and SOM C dynamics are explicitly simulated from the creation of snags to the decay of litter and dead wood and the eventual transfer of C into humified and more stable SOM pools (Kull *et al.*, 2016). The size and dynamics of dead organic matter (DOM) pools are related to the stage of stand development and the disturbance and management history of the stand. The approach of the CBM-CFS3 is to simulate the dynamics of DOM pools on the basis of available information for the stand, its history, and its ecological characteristics. The proportion of each biomass and DOM carbon pool that is transferred to another carbon pool, the atmosphere, or the forest product sector (in the case of harvesting) is defined by "disturbance matrices" that are calibrated to specific regions (Kull *et al*., 2016).

*2.2. Simulation scenarios*

Simulations were carried out for stands with characteristics of AAL located in the Saguenay–Lac Saint-Jean region (SLSJ), QC, Canada (48° 25' 59.9" N; 71° 48 '47.4" W), in the eastern balsam fir-yellow birch domain (Saucier *et al.*, 2011) (Fig. 1). According to the Canadian soil classification system (Canadian Society of Soil Science, 2020), soils in the SLSJ region, are mainly represented by Podzolic (57%, 633 000 ha) and Regosolic (25%, 277 000ha) soil orders in well-drained conditions, whereas poorly drained soils are mostly associated with Gleysolic (6%, 67 000 ha) and Organic (11%, 123 000 ha) soil orders (Raymond *et al.*, 1965; Raymond, 1971). In order to better represent soil characteristics at the regional scale, only Podzolic soil Order was selected for the simulation (Table 1).

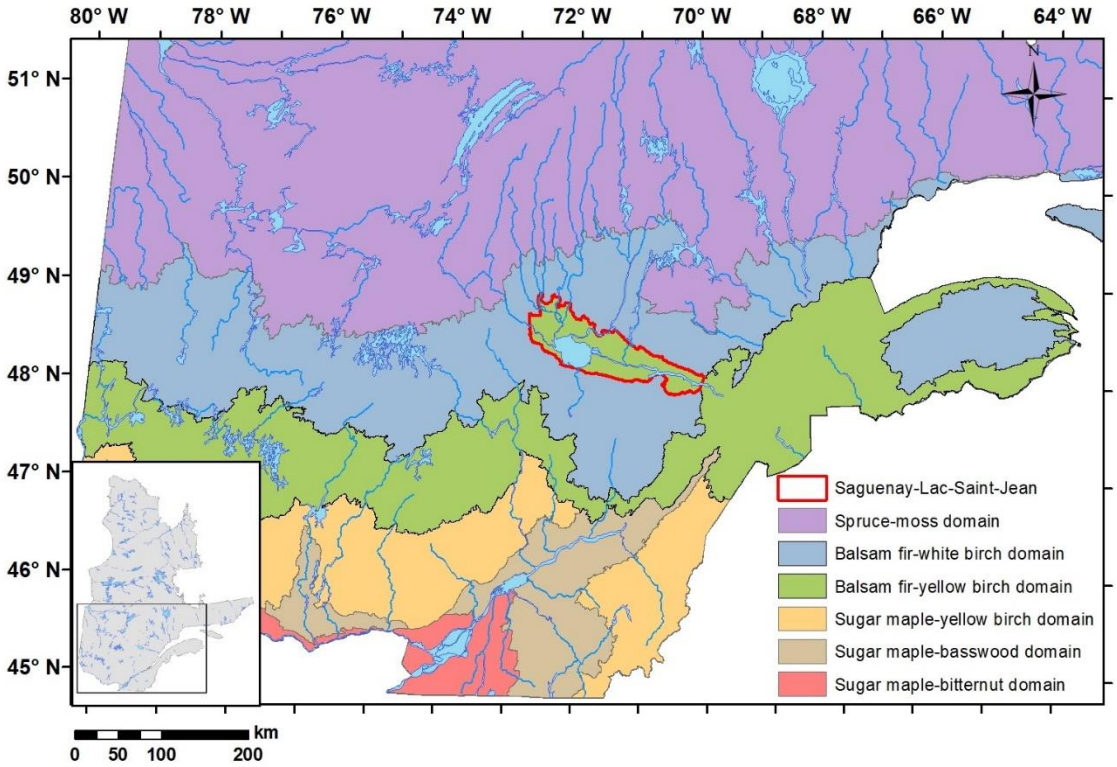


Fig. 1 Bioclimatic domains in the province of Québec, Canada (Saucier *et al.* 2011). The simulations carried out in CBM-CFS3 have the characteristics of stands located in the balsam fir-yellow birch bioclimatic domain.

The afforestation scenario (AR) is a 1-ha non-forest land afforested with no prior site preparation (no till) with one of the five most commonly planted species in the region: black spruce (*Picea mariana* [Mill] BSP; BS), white spruce (*Picea glauca* [Moench] Voss; WS), eastern white pine (*Pinus strobus* [L.]; EWP), jack pine (*Pinus banksiana* [Lamb]; JP) and tamarack (*Larix laricina* [Du Roi] K. Koch; TK). We selected merchantable wood volume yield tables for monospecific plantations with the average site quality index (SQI) of plantations in the eastern balsam fir-yellow birch domain (Poulin, 2013) and a stem density of 2000 trees $ha^{-1}$ (Table 1). The SQI is an index

calculated from the average height of the dominant trees at a given age, which allows to estimate the productivity of a site (the merchantable volume of wood produced) over time (yield tables) (Bolghari and Bertrand 1984, Laflèche *et al.* 2013). The higher the SQI the higher the merchantable volume production per hectare. The afforestation event was applied at year 1 of the 100-year simulation. As tilling or scarification are not required for plantations on AAL, no soil C losses related to soil preparation were accounted for in the simulation. C emissions related to processes preceding planting (AR scenario) were not accounted for as Gaboury et al. (2009) found them to be negligible (<0.5 % of additional C sequestration of plantations).

| | Afforested stand (AR) | Naturally regenerated stand (NR) |
|---|---|---|
| Soil type | Podzolic | Podzolic |
| Ecological boundary | Boreal shield east | Boreal shield east |
| Species | *Picea mariana* (BS)<br>*Picea glauca* (WS)<br>*Pinus strobus* (EWP)<br>*Pinus banksiana* (JP)<br>*Larix laricina* (TK) | *Picea mariana* (BS)<br>*Picea glauca* (WS)<br>*Abies balsamea* (BF)<br>*Betula papyrifera* (WB)<br>*Populus tremuloides* (TA) |
| Site Quality Index (SQI) | Plantation SQI<br>BS – 8<br>WS – 10<br>EWP – 8<br>JP – 6<br>TK – 12 | Natural stand SQI<br>BS – 15<br>WS – 15<br>BF – 15<br>WB – 15<br>TA – 18 |
| Tree density | 2000 stems $ha^{-1}$ | Low-medium-high |
| Disturbance type | Afforestation | Afforestation |
| Time of disturbance (yr) | 1 | 1 |

Table 1. Model entries for simulations of the AR and NR scenarios.

The reference scenario (NR) is a set of 1-ha agricultural non-forest stands naturally reverting to forested land dominated by one of five species that naturally regenerate in AAL of the region (Tremblay and Ouimet, 2013): BS, WS, balsam firm (*Abies balsamea*; BF), trembling aspen (*Populus tremuloides* [Michx.]; TA) and white birch (*Betula papyrifera* [Marsh]; WB). Monospecific stands were used because 1) yield tables are available for natural monospecific stands only (MRNFP, 2003); and 2) they provide a broad range of C accumulation potentials due to the use of both slow-growing/less productive (e.g., BS) and fast-growing/more productive (e.g., TA) species in the simulations. Natural regeneration was simulated by applying an afforestation disturbance and by using merchantable wood volume yield tables of natural stands for the balsam fir-yellow birch ecoregion (MRNFP, 2003) with a moderate SQI and a medium tree density level. The afforestation event was applied at year 1 but the yield table was phased out by 5, 10, 15 and 20 years to simulate various dynamics of colonization by trees following agricultural abandonment (Flinn and Vellend, 2005). Field surveys in Quebec's abandoned farmlands suggest delays of tree colonization of 15-20 years (Tremblay and Ouimet, 2013). This procedure allows CBM-CFS3 to simulate a slow natural regeneration establishment from year 1, while inducing a delay in merchantable volume production.

Initial soil organic C (SOC) content of both AR and NR stands is set by CBM-CFS3 at 74 Mg C $ha^{-1}$ by default, which corresponds to the average C content in the top 0-30 cm for Canadian cultivated podzolic soils (Janzen *et al.*, 1993; Gregorich *et al.*, 1995). This layer contains most of the dynamic SOC fraction (as opposed to inert SOC fraction), with typical turnover times ranging from a few years to several decades (Janzen et al. 1993).

*2.3. Net C accumulation and additional C sequestration potential*

Net C accumulation in AR and NR stands was calculated as the difference between C accumulation in the five carbons pools of the ecosystem at time *t* and stands' initial C stock (Equation 1). The simulation considers that there is no tree, shrubs or grasses biomass at year 0 for both scenarios; thus, stands' initial C stock equals initial soil C stock.

Net C accumulation (Mg C $ha^{-1}$) = Total C stock $_{(t)}$ – Total C stock $_{(t0)}$ Eq. 1

The additional C sequestration potential of AAL afforestation was calculated as the difference between the net C accumulation in the AR and the NR scenarios (Equation 2):

Additional C sequestration (Mg C $ha^{-1}$) = Net C accumulation [AR] $_{(t)}$ – Net C accumulation [NR] $_{(t)}$ Eq. 2

## *3. Results*

### *3.1. Afforestation scenarios*

The total amount of C in the plantations after 100 years (Fig. 2) varies by 128 Mg C $ha^{-1}$ between the stand afforested with WS (329 Mg C $ha^{-1}$) and the one afforested with TK (201 Mg C $ha^{-1}$). Stands afforested with EWP (265 Mg C $ha^{-1}$), JP (250 Mg C $ha^{-1}$) and BS (229 Mg C $ha^{-1}$) have intermediate values. AGBM becomes the main C stock after 45-50 years and accumulates a maximum amount of 147, 112, 103, 90 and 72 Mg C $ha^{-1}$ in WS, EWP, JP, BS and TK stands, respectively. This C pool starts to decline after ~85 years in both WS and JP stands (Fig. 2). BGBM contains a maximum of 32, 25, 23, 20 and 16 Mg C $ha^{-1}$ in WS, EWP, JP, BS and TK stands, respectively, accounting for 18% of total biomass C stock for all species. The initial C stock of podzolic soils in the study region is set to 74 Mg C $ha^{-1}$ and declines following afforestation in all stands, reaching minimum values at 91 yr for TK, 78 yr for BS, 73 yr for JP and EWP and 58 yr for WS. After 100 years soil C stock is about 67-71 Mg C $ha^{-1}$ (Fig. 2).

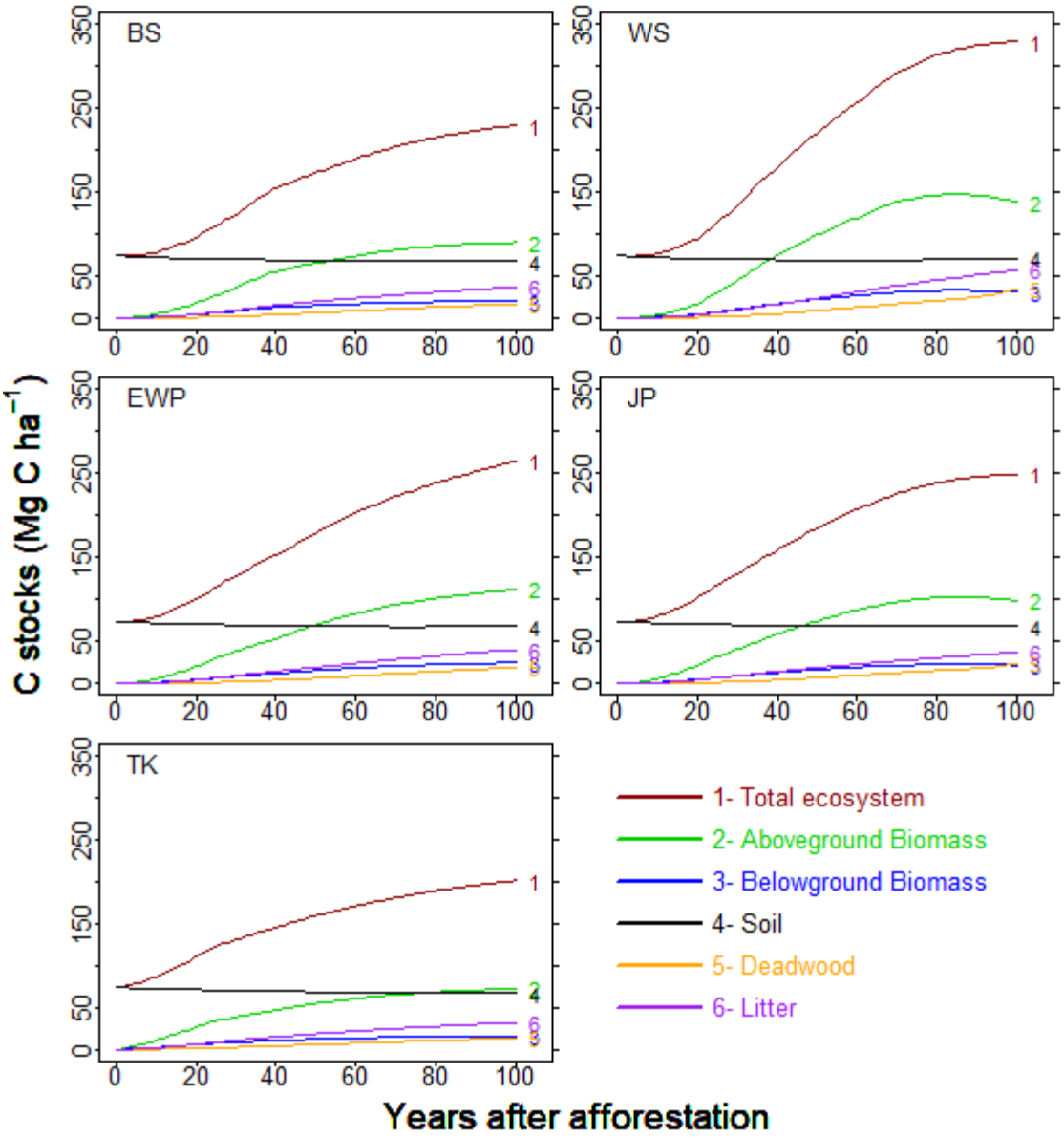


Fig. 2 Carbon stock changes in the five pools and total ecosystem of the managed stands (AR) afforested with black spruce (BS), white spruce (WS), eastern white pine (EWP), jack pine (WP) and Tamarack (TK).

The dynamics of net C accumulation is similar for all species through the first 25 years following afforestation, except for TK, whose initial growth is faster compared to the other species (Fig. 3). After 40 years, however, the amount of C accumulated in the WS stand is up to 47% higher (105 Mg C $ha^{-1}$) than in the other stands (71-85 Mg C $ha^{-1}$). After 100 years, BS, JP and EWP stands contain 155, 175 and 191 Mg C $ha^{-1}$, respectively, vs. 255 Mg C $ha^{-1}$ for the WS stand. The TK stand has the smallest net C accumulation after 100 years of growth (127 Mg C $ha^{-1}$).

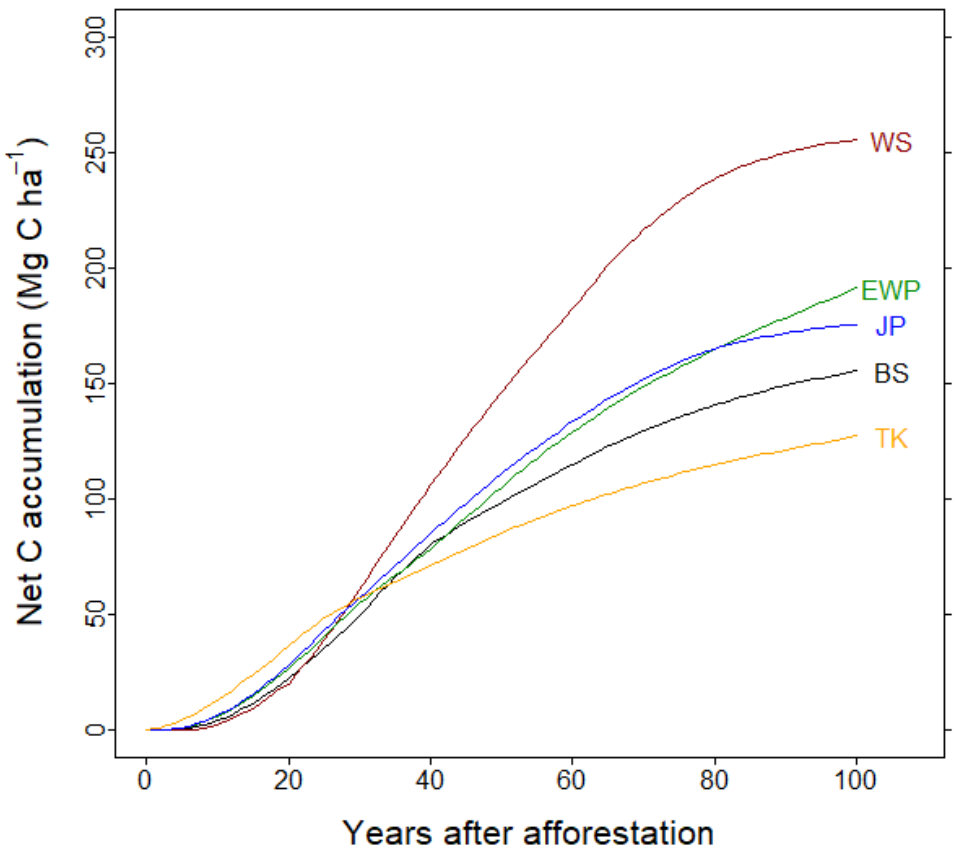


Fig. 3 Net C accumulation (Mg C $ha^{-1}$) in stands afforested with black spruce (BS), white spruce (WS), eastern white pine (EWP), jack pine (WP) and Tamarack (TK).

*3.2. Natural regeneration scenarios*

After 100 years, total C stock is higher in NR stands dominated by hardwoods (219 and 201 Mg C $ha^{-1}$ for TA and WB, respectively) than in those dominated by softwoods (195, 181 and 156 Mg C $ha^{-1}$ for WS, BF and BS, respectively) (Fig. 4). Tree biomass slowly starts to increase from year 1 and reaches 1 Mg C $ha^{-1}$ between 10 to 15 years after abandonment. At the end of the simulation period, AGBM contains 72, 71, 57, 65 and 52 Mg C $ha^{-1}$ in WS, TA, WB, BF, and BS stands, respectively vs. only 12-16 Mg C $ha^{-1}$ for BGBM. Litter represents a larger C pool than BGBM for all stands, accounting for about 57, 56, 30, 28 and 24 Mg C $ha^{-1}$ in TA, WB, WS, BF and BS stands, respectively. The initial soil C stock is set at 74 Mg C $ha^{-1}$ for all stands and declines after abandonment to reach 62, 65, 64, 68 and 67 Mg C $ha^{-1}$ in BS, WS, BF, TA and WB stands after 100 years (Fig. 4). The decrease in soil C stock occurs throughout the whole 100-year period for BS and BF, whereas minimum values of 65 Mg C $ha^{-1}$ are reached at 95 yr for WS, of 66 Mg C $ha^{-1}$ at 76 yr for WB and 67 Mg C $ha^{-1}$ at 71 yr for TA.

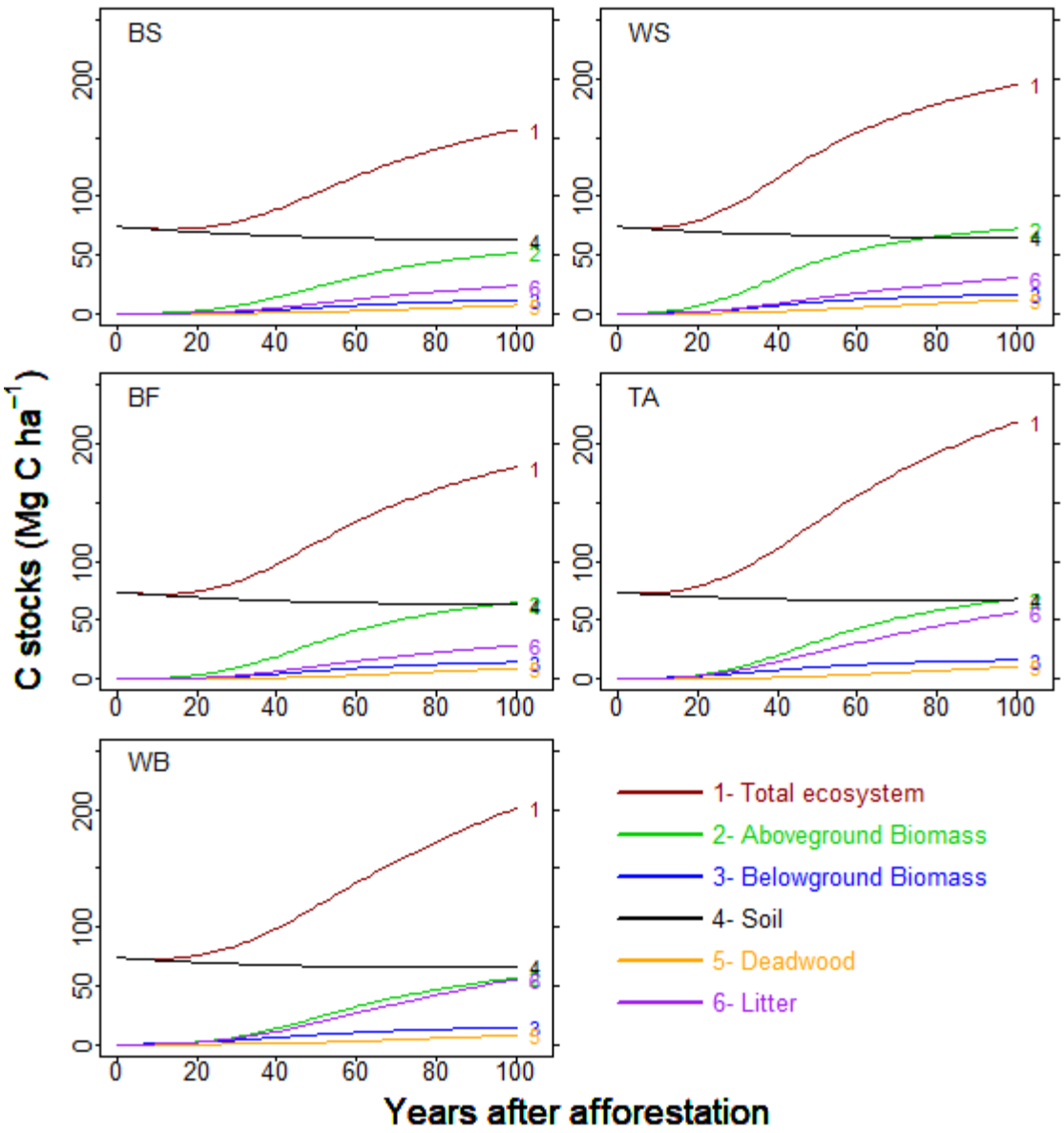


Fig. 4 Carbon stock changes in the five pools and total ecosystem of the reference stands naturally reverting to medium density monospecific forests of black spruce (BS), white spruce (WS), balsam fir (BF), trembling aspen (TA) and white birch (WB).

Net C accumulation in NR stands is negative during the first years and then turns positive at 13, 14, 17, 20 and 24 yr in WS, TA, WB, BF and BS stands, respectively. The net C accumulation at 100 years nevertheless varies strongly depending on the selected species (Fig. 5), decreasing in the following order: TA > WB > WS > BF > BS stands. With a 15-year delay for tree establishment, net C accumulation at year 100 for these species amounts 145, 127, 121, 106 and 82 Mg C $ha^{-1}$, respectively. For each species, tree establishment delay impacts the dynamics of C accumulation, with values up to 55% higher with a 5-year than with a 15-year delay after 50 years. However, this relative difference declines to 8-10% after 100 years (Fig. 5; Table S1). Net C accumulation also varies with the density of tree cover in the natural regeneration with values about 35% higher in high- versus low-density stands for BS, WS, BF and TA, and 65% in WB stands after 100 years (Fig. S1).

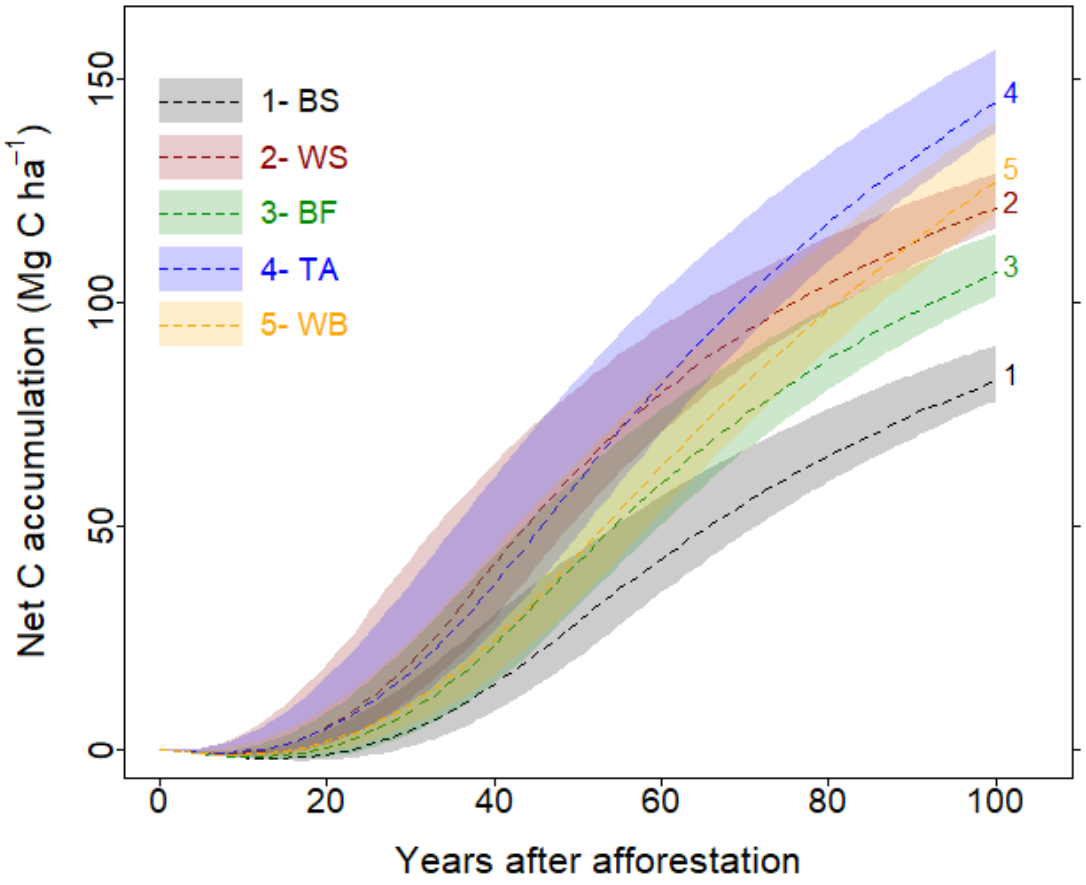


Fig. 5 Net C accumulation (Mg C ha$^{-1}$) in naturally regenerated stands (NR scenario) with black spruce (BS), white spruce (WS), balsam fir (BF), trembling aspen (TA) and white birch (WB) as dominant species. The colored strips show the range of net C accumulation with tree establishment delays of 5 and 20 years (high- and low-end values of the range, respectively). The dashed lines show values for a natural regeneration delay of 15 years.

*3.3. Additional C sequestration potential*

The additional C sequestration potential of AAL afforestation varies widely depending on both the planted species in AR stands and the NR scenario (i.e., the dominant species in the natural regeneration) (Fig. 6). The maximum additional C sequestration potential is reached for stands afforested with WS combined with a natural regeneration of BS at year 87 (175 Mg C $ha^{-1}$). The lowest potential occurs with a combination of stand afforestation with TK and a natural regeneration of WS at year 28 (37 Mg C $ha^{-1}$) (Table 2). The additional C sequestration potential of WS plantations after 30 years (i.e., by 2050) does not strongly vary with the NR scenario, ranging from 41 to 56 Mg C $ha^{-1}$. Additional C sequestration potential starts to stabilize or even to decline 28 years after planting for TK plantations vs. after ~80 years for EWP, JP and WS plantations (Fig 6). Changing the delay of tree establishment on NR stands (from 5 to 20 years) impacts additional C sequestration predictions. For instance, shortening this delay to 5 years reduces WS plantations' additional C sequestration potential by 13-27% and by 5-11% after 50 and 100 years, respectively, depending on the dominant species in the NR stand (Table S1). Conversely, lengthening this delay to 20 years increases WS plantations' additional C sequestration potential by 7-14% and by 3-6% after 50 and 100 years, respectively.

| AR species | NR species | Maximum additional C sequestration (Mg C ha$^{-1}$) | Additional C sequestration (Mg C ha$^{-1}$) | | | Additional C sequestration rate (Mg C ha$^{-1}$ yr$^{-1}$) | | |
|---|---|---|---|---|---|---|---|---|
| | | | Year 30 | Year 50 | Year 100 | Year 30 | Year 50 | Year 100 |
| BS | BS | 75 (80yr) | 45 | 70 | 73 | 1.50 | 1.40 | 0.73 |
| | WS | 38 (40yr) | 30 | 36 | 34 | 0.99 | 0.72 | 0.34 |
| | BF | 57 (45yr) | 41 | 56 | 49 | 1.36 | 1.13 | 0.49 |
| | WB | 56 (45yr) | 39 | 55 | 28 | 1.30 | 1.10 | 0.28 |
| | TA | 43 (40yr) | 32 | 38 | 11 | 1.06 | 0.77 | 0.11 |
| EWP | BS | 109 (100yr) | 50 | 77 | 109 | 1.68 | 1.53 | 1.09 |
| | WS | 70 (100yr) | 35 | 43 | 70 | 1.17 | 0.85 | 0.70 |
| | BF | 84 (100yr) | 46 | 63 | 85 | 1.53 | 1.26 | 0.85 |
| | WB | 67 (70yr) | 44 | 62 | 64 | 1.48 | 1.23 | 0.64 |
| | TA | 47 (65yr) | 37 | 45 | 46 | 1.24 | 0.90 | 0.46 |
| JP | BS | 99 (83yr) | 53 | 82 | 93 | 1.75 | 1.65 | 0.93 |
| | WS | 61 (80yr) | 37 | 48 | 54 | 1.24 | 0.97 | 0.54 |
| | BF | 78 (75yr) | 48 | 69 | 69 | 1.60 | 1.38 | 0.69 |
| | WB | 70 (65yr) | 47 | 67 | 48 | 1.55 | 1.35 | 0.48 |
| | TA | 51 (60yr) | 39 | 51 | 31 | 1.31 | 1.02 | 0.31 |
| TK | BS | 57 (45yr) | 52 | 57 | 45 | 1.75 | 1.13 | 0.45 |
| | WS | 37 (28yr) | 37 | 23 | 6 | 1.24 | 0.45 | 0.06 |
| | BF | 49 (34yr) | 48 | 43 | 20 | 1.60 | 0.87 | 0.20 |
| | WB | 47 (35yr) | 47 | 42 | 0 | 1.55 | 0.83 | 0.00 |
| | TA | 39 (30yr) | 39 | 25 | -18 | 1.31 | 0.51 | -0.18 |
| WS | BS | 175 (87yr) | 56 | 118 | 173 | 1.88 | 2.35 | 1.73 |
| | WS | 136 (88yr) | 41 | 84 | 134 | 1.37 | 1.67 | 1.34 |
| | BF | 152 (85yr) | 52 | 104 | 149 | 1.73 | 2.09 | 1.49 |
| | WB | 140 (80yr) | 50 | 103 | 128 | 1.68 | 2.05 | 1.28 |
| | TA | 121 (80yr) | 43 | 86 | 111 | 1.43 | 1.73 | 1.11 |

Table 2. Additional C sequestration potential and additional C sequestration rates of abandoned agricultural lands (AAL) afforestation for different afforestation (AR) and natural regeneration (NR) scenarios, and at different times after afforestation (30, 50 and 100 years). The time at which maximum additional C sequestration is reached is shown in parentheses. Planted species in AR stands are: black spruce [BS], white spruce [WS], eastern white pine [EWP], jack pine [JP] and tamarack [TK]; and dominant species in NR stands are: black spruce [BS], white spruce [WS], balsam fir [BF], trembling aspen [TA] and white birch [WB]. Values are shown for a tree establishment delay after land abandonment of 15 years.

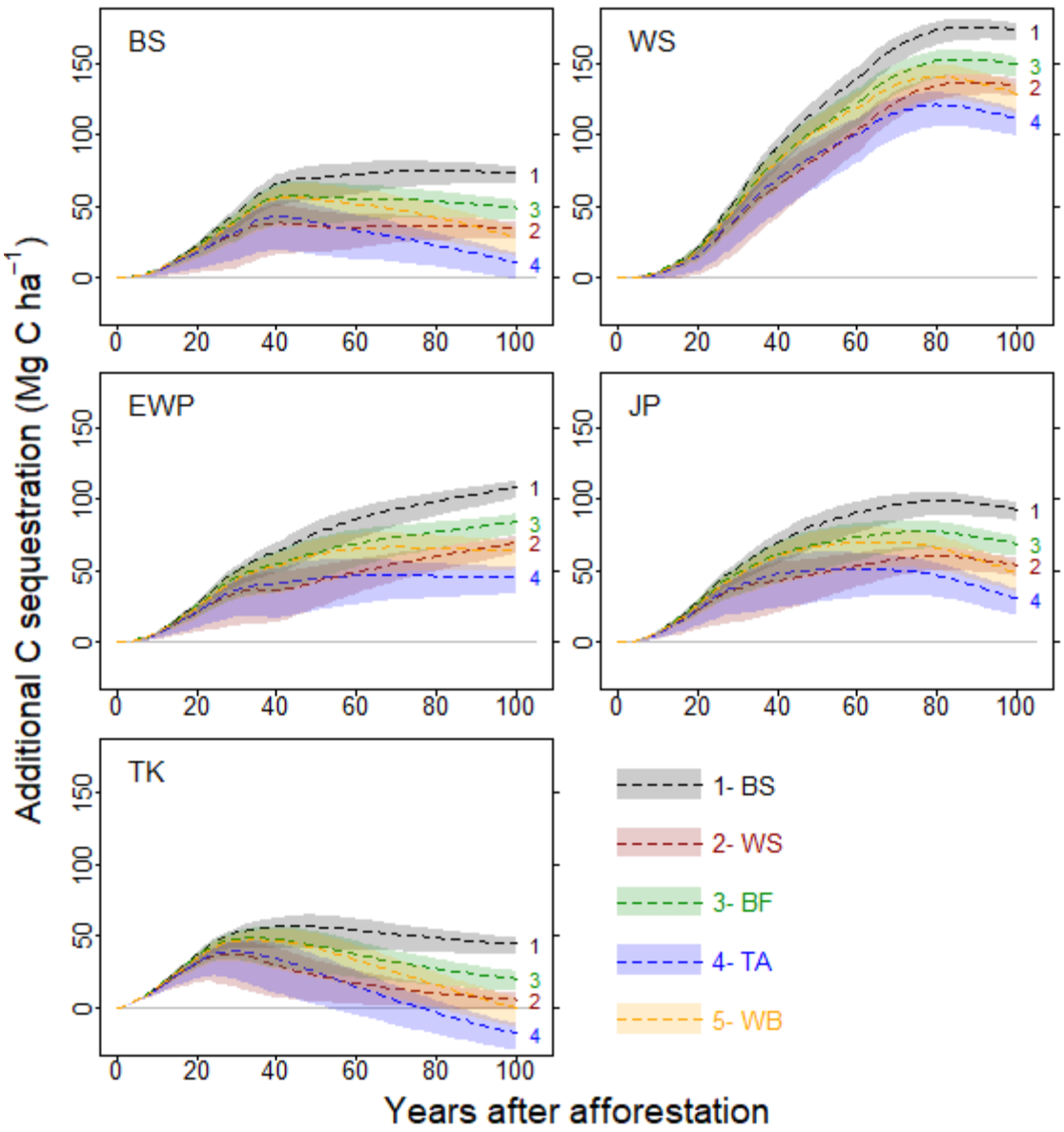


Fig. 6. Additional C sequestration potential of agricultural lands following abandonment as a function of planted tree species (black spruce [BS]; white spruce [WS]; eastern white pine [EWP]; jack pine [BS]; and tamarack [TK]) and tree species dominating the natural regeneration following abandonment (black spruce [BS], white spruce [WS], balsam fir [BF], trembling aspen [TA] and white birch [WB]). The colored strips show the range of additional C sequestration with tree establishment delays of 5 and 20 years (high- and

low-end values of the range, respectively). The dashed lines show values for a tree establishment delay after abandonment of 15 years.

## *4. Discussion*

The abandonment of agricultural lands generally leads to the establishment of woody species, which in the long-term results in net C accumulation in both the vegetation and the soil (Bell *et al.*, 2020; Huang *et al.*, 2020). Depending on site characteristics and species, afforestation of AAL may accelerate and increase C sequestration relative to natural regeneration although this hypothesis has rarely been tested. Most studies have indeed assessed net C accumulation due to natural regeneration or afforestation of AAL relative to their previous agricultural use (Laganière *et al.*, 2010) but few of them have tried to quantify the potential additional C sink that afforestation may create relative to natural regeneration (Tremblay and Ouimet, 2013). To assess this additional C sink, it is necessary to account for the C that would accumulate in the natural regeneration in the absence of AAL management (reference scenario) (IPCC, 2003). In this study, we simulated several reference and afforestation scenarios to assess the long-term C sequestration dynamics and potential of AAL afforestation. Our modelling approach contrasts with other studies that have used chronosequences (Tremblay and Ouimet, 2013). This approach has the advantage to circumvent potential issues related to poorly selected chronosequences, such as variability in site history and ecological characteristics (Johnson and Miyanishi, 2008), which can lead to biased results (Vesterdal *et al.*, 2002; Laganière *et al.*, 2010).

*4.1 Biomass C stocks in AR stands*

Simulated live biomass (AGBM + BGBM) C stocks in afforested AAL varies depending on species selection (Fig. 2), but is in the range of what was found in other studies in the same region (Tremblay *et al.*, 2006; Stinson *et al.*, 2011). However, Tremblay and Ouimet (2013) reported a much faster C accumulation and higher C stocks in the AGBM of WS plantations with 97 Mg C $ha^{-1}$ after only 35 years and ~200 Mg C $ha^{-1}$ after 50 years vs. ~50 Mg C $ha^{-1}$ and ~100 Mg C $ha^{-1}$ in our simulation (Fig. 3). This difference may result from higher plantation density in Tremblay and Ouimet (2013), but also from higher SQI, which can vary greatly from one site to another depending on previous agricultural use which affects soil properties and fertility (Wall and Hytönen, 2005). Medium SQI were chosen for our simulations to provide conservative C sequestration estimates. Preliminary field observations on afforested AAL in the region point towards higher SQI than the ones used in the present simulations, especially for WS stands. Field measurements at 5 years suggest that WS stands may reach SQI of 14, which is markedly higher than the SQI of 10 used in the simulation (Table S2).

*4.2 Net C accumulation in AR vs. NR stands*

The results of our simulations of C accumulation are in good agreement with field data from other studies. For instance, Tremblay and Ouimet (2013) found that live biomass C stock was on average 150% higher in AR than in NR stands after 50 years. In our simulations, the live biomass stocks after 50 years is ~130% higher in the stand afforested with WS than in the stand naturally regenerated (NR) with the same species. The

difference was slightly higher for BS, the biomass stocks in the AR stand after 50 years being ~180% higher than that in NR stand with BS as the dominant species. Using a global database of 91 studies of secondary successions around the world, Anderson et al. (2006) found that the rate of accumulation of biomass in plantations was on average 122% higher than in naturally regenerated sites, with no significant effects of initial disturbance type. Similarly, our simulations show that net C accumulation rate in tree biomass in the first 50 years is on average 131% higher for the simulated afforestation scenarios than for the reference scenarios (1.77 Mg C $ha^{-1}$ $yr^{-1}$ vs. 0.77 Mg C $ha^{-1}$ $yr^{-1}$). This difference is however of only 61% after 100 years due to the full establishment of the natural regeneration (1.25 Mg C $ha^{-1}$ $yr^{-1}$ vs. 0.77 Mg C $ha^{-1}$ $yr^{-1}$).

Higher live biomass in afforested AAL throughout the first decades following afforestation is mainly attributable to the delay before succession establishment in NR stands. A ~20-year delay was observed in Québec's AAL resulting in very low C accumulation in AGBM and BGBM during the 30 years following abandonment (Tremblay and Ouimet, 2013). Such delays were also observed in landscapes throughout Europe and eastern North America recovering from past agricultural use (Flinn and Vellend, 2005) and in other abandoned lands worldwide (Harmer *et al.*, 2001; Bonet and G. Pausas, 2004; Sirami *et al.*, 2006; Ruskule *et al.*, 2012). Modelling studies of C sequestration dynamics on AAL also showed that C sequestration was generally slow in the early years after abandonment, although C accumulation increased significantly after (Schierhorn *et al.*, 2013; Voicu *et al.*, 2017). In the present study, we used yield tables phased out by 15 years as this matches the biomass data found in Tremblay et Ouimet (2013) on naturally regenerated AAL after 50 years. There is however a lack of

knowledge regarding the dynamics of the natural regeneration in AAL and how it is influenced by previous agricultural use and practices, topography or pedoclimatic conditions. We have tackled this source of uncertainty by using several establishment delay values. The results show an effect of the delay period on the dynamics of C accumulation in the natural regeneration (NR stands) especially in the first 60 years after abandonment (Fig 5). This uncertainty, however, declines with time and does not affect the conclusions regarding the C accumulation potential of the different species (low or no overlaps of their ranges of values after 100 years). By impacting NR scenario's C accumulation potential, the delay in natural regeneration establishment necessarily affects additional C sequestration estimates. Shortening this delay to 5 years reduces plantations' additional C sequestration potential but this effect diminishes with time with differences amounting less than 14 Mg C $ha^{-1}$ after 100 years (Table S1). Conversely, lengthening this delay to 20 years as reported by Tremblay and Ouimet (2013) increases plantations' additional C sequestration potential, but the difference is low after 100 years (< 7 Mg C $ha^{-1}$).

*4.3 Soil C stock*

The initial soil C stock of 74 Mg $ha^{-1}$ compares to the 74–100 Mg C $ha^{-1}$ reported for AAL with similar pedo-climatic conditions (Tremblay *et al.*, 2006; Ouimet *et al.*, 2007; Tremblay and Ouimet, 2013). The model predicts a net decrease in soil C content of 5–7 Mg C $ha^{-1}$ over the 100 year-period. This is less than the ~28 Mg C $ha^{-1}$ decrease (from 82 to 53.9 Mg· $ha^{-1}$) reported by Tremblay and Ouimet (2013) for the 0–50 cm soil layer over the 50 years following afforestation. In contrast, other studies have shown that the

natural reversion to woody lands or plantations in similar pedo-climatic contexts can induce an accumulation of C in the soil due to the development of a forest floor, especially in the case of conifer afforestation (Laganière *et al.*, 2010), and the accumulation of organic C in the subsoil (Vesterdal *et al.*, 2002; Ouimet *et al.*, 2007), especially under broadleaved species (Laganière *et al.*, 2010).

The direction and the magnitude of the changes in soil C stocks following afforestation of AAL depends on various factors among which previous land-use, climate, management practices, planted species and soil texture (Mayer *et al.* 2020). For instance, afforestation of former croplands generally leads to C accumulation in the soil whereas mean soil C stocks increases less, remain unchanged or even decreases following afforestation of former grasslands (Mayer et al. 2020 and references therein). Some studies suggest that the soil C decrease that follows afforestation may be predominantly attributable to the lack of plant growth, leading to low C inputs into the soil, rather than to soil disturbance due to site preparation (Paul *et al.*, 2002; Tremblay *et al.*, 2006). On the other hand, other studies have shown a significant effect of pre-planting disturbances associated with tillage operations, which are believed, along with the low net primary productivity of the newly established plantation, to be responsible for SOC losses in the first few years following afforestation (Turner and Lambert, 2000; Laganière *et al.*, 2010, Mayer *et al.* 2020). As a consequence, minimizing pre-planting disturbances can increase SOC stocks by 15% (Laganière *et al.*, 2010). Soil C emissions due to tilling disturbance are not implemented by default in CBM-CFS3 simulations (Kull *et al.*, 2016), which is not an issue in the present study as no intensive tillage or scarification operation is required before planting in AAL. The observed decrease in SOC stock in our simulations is

therefore entirely due to unbalanced C inputs and outputs to/from the soil. The establishment of the forest, whether planted or naturally regenerated, is a slow process, and several years are necessary until litter's C is incorporated within the soil component of the ecosystem (Kull *et al.* 2016) . In our simulations, the soil C pool starts to slowly replenish after a few decades due to litter input (Fig. 2). These results are in accordance with several review studies concluding that the initial loss of soil organic carbon (SOC) after afforestation is followed by a gradual increase in C stocks which subsequently generates a net gain (Paul *et al.* 2002; Laganière *et al.* 2010; Li *et al.* 2012). Litter production gradually produces a significant C pool with 20 Mg C $ha^{-1}$ for all species after 50 years and 31 (TK) to 48 (WS) Mg C $ha^{-1}$ after 100 years. This is much higher than what was reported in studies carried out in the same ecological region whether in plantations or naturally regenerated stands (Tremblay *et al.*, 2006; Tremblay and Ouimet, 2013) but in the same range as values reported by Vesterdal et al. (2002) in Scandinavia and Richter et al. (1999).

*4.4 Additional C sequestration potential*

The dynamics and the potential of additional C sequestration depends on both AR and NR scenarios as well as the considered time frame (Table 2, Fig. 6). The additional C sequestration rate is generally higher during the first decades (Fig. 6) owing to tree growth in AR stands coupled with the quasi-absence of vegetation in the NR scenario due to the delay in tree establishment. The additional C sequestration potential then declines due to a stabilization of biomass production in AR stands combined with an increasing biomass production in the NR stands. It is difficult to know what the additional C

sequestration would be beyond 100 years, but it would likely be positive as the yield is generally higher in plantations than in natural stands at maturity (MRNFP, 2003). Nevertheless, our results clearly show that afforestation induces a faster C accumulation than natural regeneration. The additional C sequestration potential of ~50 Mg C $ha^{-1}$ of most tested scenarios over a reference period of 30 years would help decrease net C emissions and thus contribute to achieve Canada's net-zero emissions target by 2050.

There is a large amount of C accumulating in the live biomass of AR stands, but the additional C sequestration is limited by the establishment of a natural regeneration in the reference scenario, which is not the case when afforestation is conducted in boreal open woodlands (OW), where natural regeneration is very low (Jasinski and Payette, 2005; Girard *et al.*, 2008). The study of Gaboury et al. (2009) showed that the afforestation of Quebec's boreal OWs with BS has an additional C sequestration potential of ~77 Mg C $ha^{-1}$ over a period of 70 years. In comparison, our simulations show that after the same period of time AAL afforestation with BS has an additional sequestration potential of 74 Mg C $ha^{-1}$ considering a natural regeneration dominated by BS (Fig. 6). However, the rate of C accumulation is initially higher in AAL with a net C accumulation of 65 Mg C $ha^{-1}$ in the first 40 years vs. ~25 Mg C $ha^{-1}$ in boreal OWs (Gaboury *et al.*, 2009). The choice of fast-growing and more productive species may however enhance additional C sequestration. Our simulations show that among the tested species, WS offers the greatest additional C sequestration potential (121-175 Mg C $ha^{-1}$) due to its higher biomass production. The additional C sequestration rate is 1.67 Mg C $ha^{-1}$ $yr^{-1}$ with WS in both AR and NR scenarios, which is consistent with Tremblay and Ouimet (2013) who found a 1.7 Mg C $ha^{-1}$ $yr^{-1}$ greater C accumulation rate in plantations than in naturally

regenerated sites dominated by WS over 50 years. JP and EWP offer approximatively similar additional C sequestration potentials (Fig. 6, Table 2), although the maximum value for EWP is not yet reached at 100 years, suggesting that EWP plantations could sequester higher amounts of C in the very long-term (>100 years) (Urbano and Keeton, 2017). Although TK plantations have the lowest additional C sequestration potential in the long-term, our simulations show they have the fastest one over the first 25 years. They accumulate as much C in the first 20 years as red pine plantations in southern Quebec, one of the species with the highest organic C storage potential in the region (Ouimet *et al.*, 2007). As such, TK plantations may offer an option for rapid C sequestration. Furthermore, these plantations likely have a lower impact on albedo relative to other conifers due to TK's deciduous foliage, which would improve their climate mitigation potential (Breuer *et al.*, 2003; Kuusinen *et al.*, 2014; Kalliokoski *et al.*, 2020). The use of more productive species, such as hybrid larch or hybrid poplar, may induce an even more rapid and higher C sequestration (Fig. S2) while having a lower impact on the albedo due to their deciduous foliage.

*4.5 Limitations of the study*

There are three main limitations to the present study. First, the simulated monospecific NR scenarios are unrepresentative of natural regeneration establishment on AAL, which does not occur in a single event and is generally a mix of several shrub and tree species (Tremblay and Ouimet, 2013). Similarly, it is likely that an understory of shrubs and other trees would develop along planted trees in the afforested stands in the absence of management practices. Thus, it is reasonable to assume that our simulations similarly

underestimate live biomass and C accumulation in both the afforestation and the reference scenarios.

There is also uncertainty regarding the natural regeneration establishment dynamics, particularly in the tree establishment delay after abandonment. We chose a 15-year delay in accordance with the literature but as shown by our sensitivity analysis, a shorter delay would reduce additional C sequestration potential of plantations, especially in the first 50 years following abandonment. Further field research is needed to accumulate data on natural regeneration dynamics in AAL, which would be useful to improve model simulations. Also, NR scenario simulations were run with a medium tree density natural regeneration. Using low- or high-density yield tables would respectively affect net C accumulation in the NR scenario by up to -25% and +25% for certain species after 100 years and therefore additional C sequestration potential of plantations (Fig. S1). Finally, our study does not assess the effect of soil type, soil preparation prior to planting, previous land use and management history on C accumulation dynamics although these variables likely have an effect on tree growth and especially soil C dynamics (Mayer *et al.* 2020). However, CBM-CFS3 does not allow to study these effects in a straightforward manner. For instance, soil C dynamics are similar regardless of soil types and management practices. Available data regarding the effect of these variables should be gathered and integrated to the model.

*5. Conclusion*

The goal of this study was to assess the long-term additional C sequestration potential of AAL afforestation relative to natural regeneration in the eastern balsam fir-yellow birch domain by using a modelling approach. The results of our simulations are consistent with results from chronosequence approaches found in the literature, demonstrating that the methodology we used can correctly estimate the dynamics of natural successions in AAL and help predict the additional C sequestration resulting from afforestation. The simulations showed that WS plantations have the highest additional C sequestration potential, ranging from 111 to 173 Mg C $ha^{-1}$ over 100 years depending on the natural regeneration scenario. Although the AAL additional C sequestration potential is similar to that of afforested boreal OW of eastern Québec, C accumulation in afforested AAL is much faster due to higher tree growth rate in the first 40 years, making AAL afforestation advantageous with respect to the net-zero emissions target by 2050. Therefore, AAL represent an option for Canadian government's large-scale afforestation program although the additional C sequestration potential of AAL remains modest in the long term, and is uncertain due to various possible natural regeneration scenarios. Some key parameters used for modelling (e.g., plant growth, natural regeneration establishment dynamics, soil fertility, initial soil carbon, etc.) may indeed strongly vary locally making extrapolations of our results limited. Conducting replicated and controlled surveys in the field at local scales remains essential in order to adequately monitor C accumulation over time and validate the additional C sequestration potential of AAL.

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