Université de Montréal

Boisement d'une ancienne mine de chrysotile du sud du Québec :
Essais de divers types de technosols et d'essences ligneuses

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Boisement d'une ancienne mine de chrysotile du sud du Québec :
Essais de divers types de technosols et d'essences ligneuses

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Résumé

L’industrie minière de l’amiante a profondément transformé les paysages du sud du Québec, laissant environ 800 millions de tonnes de résidus miniers sous la forme de haldes près des villes minières telles Thetford Mines et Black Lake. Certains enjeux environnementaux découlant de ces paysages, notamment l’érosion des haldes par le vent et l’eau, ont été atténués avec succès sur certains sites en les recouvrant de technosols composés d’un mélange de matières résiduelles fertilisantes, puis en les ensemençant avec des graminées. Cependant, aucune tentative de boisement n’avait été faite jusqu’à présent, car ces milieux étaient considérés comme trop contraignants pour l’établissement et la croissance des arbres. Dans cette étude, nous avons développé deux plantations expérimentales dans une mine d’amiante en déclassement dans le sud du Québec, sur des stériles et sur des résidus miniers, dans lesquelles a été comparée la performance de huit espèces d’arbres, clones et provenances plantés sur deux types de technosols, tous deux construits à partir d’un mélange de biosolides municipaux et de boues de désencrage aménagés en andains. Les propriétés physiques et chimiques des technosols, comme l’état hydrique et l’activité ionique du sol, ainsi que la survie et la croissance des arbres, ont été évaluées. La nutrition foliaire et l’utilisation de l’eau ont également été étudiées pour trois clones de peupliers hybrides au cours de leur troisième saison de croissance. Les clones de peupliers hybrides avaient une survie significativement plus élevée (87 à 94 %) après trois ans que toutes les espèces de conifères (10 à 56 %) testées. L'espèce résineuse présentant les taux de survie les plus élevés était l'épinette blanche (46 à 56 %). Les conditions sèches lors de la mise en terre peuvent en partie expliquer les taux de mortalité plus élevés des conifères. Les taux de croissance moyens des peupliers hybrides étaient de 45 cm an^{-1}, alors que certains arbres atteignaient plus de 300 cm de hauteur après trois ans. Le phosphore et le potassium étaient les nutriments qui limitaient le plus la croissance. Le clone DN×M-915508 semble être le plus tolérant à la sécheresse. Bien qu'il y ait peu de différences entre les technosols, les résultats suggèrent que l'ajout de sols contaminés de classe B au mélange de matières résiduelles fertilisantes a des avantages pour la survie et la croissance des arbres, probablement en raison du volume de sol plus élevé. Notre étude a démontré une technologie prometteuse pour la remise en état par le boisement des mines d’amiante du sud du Québec. Certains traitements devraient être ajoutés pour limiter la concurrence des plantes au cours des premières années afin d'augmenter la survie et la croissance des arbres, alors qu'un volume limité de technosol et donc une faible disponibilité en humidité et en nutriments pourraient compromettre la croissance à moyen et long terme des arbres.

Mots-clés : mine d’amiante en déclassement, remise en état, technosols, boisement, matières résiduelles fertilisantes, conifères, peupliers hybrides

Abstract

The asbestos mining industry has deeply transformed the landscapes of southern Quebec, leaving about 800 million tons of tailings and waste rock in large piles near former mining towns such as Thetford Mines and Black Lake. Some of the environmental issues arising from these landscapes (e.g. wind and water erosion) have been successfully addressed at some sites by covering the piles with a mixture of by-products as the technosol and then seeding with grasses. However, no attempt for afforestation had been made thus far because these environments were thought to pose too many constraints on tree establishment and growth. In this study, we developed two experimental plantations at a decommissioned asbestos mine in southern Quebec, on waste rocks and on tailings, as a means to test the performance of eight tree species/clones/provenances planted on two types of technosols, both mainly constructed from the mixing of municipal biosolids and deinking sludge and configured in small windrows. Physical and chemical properties of technosols, including soil water availability and ionic activity, and seedling survival and growth were assessed. Foliar nutrition and water use were also evaluated for three hybrid poplar clones during the third growing season. Hybrid poplar clones had significantly higher survival (87 to 94%) after three years than all conifer species (10 to 56%) tested, although dry conditions at planting could have had adverse effects on conifers. The conifer species exhibiting the highest survival rates was white spruce (46 to 56%). Average growth rates of hybrid poplars were 45 cm y^{-1}, whereas some trees reached over 300 cm in height after three years. Clone DN×M-915508 is believed to be more drought tolerant than the other hybrid poplar clones tested. Phosphorus and potassium were the most growth-limiting nutrients. Although there were few differences between the technosols, results suggest that the addition of Class B contaminated soils to the main mixture of by-products has benefits regarding tree survival and growth, likely because of its larger volume. Our study demonstrated a promising technology for reclamation through afforestation of asbestos mines in southern Quebec. Some treatments should be added to limit plant competition within the first few years as a means to increase survival and growth, whereas limited technosol volume and thus low moisture and nutrient availability could compromise mid- to long-term growth of trees.

Keywords: decommissioned mine, mine reclamation, technosols, afforestation, fertilizing by-products, conifers, hybrid poplars

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A_{net}	Net carbon assimilation rate
ANOVA	Analyse de variance
B	Bore
C	Carbone
C_a	Concentration de dioxyde de carbone dans la chambre
Ca	Calcium
Cd	Cadmium
cm	Centimètre
cm y^{-1}	Centimeter per year
cm^2	Centimètre carré
cm^2g^{-1}	Centimètre carré par gramme
cm^3	Centimètre cube
CO_2	Dioxyde de carbone
Cu	Cuivre
Fe	Fer
g	Gramme
g_{sw}	Stomatal conductance on water vapor
ha	Hectare
HCl	Acide chlorydrique
HP	Hybrid poplar
JP	Jack pine
K	Potassium
kg ha^{-1}	Kilogramme par hectare
kg m^{-2}	Kilogramme par mètre carré
kg m^{-3}	Kilogramme par mètre cube
kPa	Kilopascal
kV	Kilovolt
L ha^{-1}	Litre par hectare
m	Mètre
mg	Milligramme
Mg ha^{-1}	Mégagramme par hectare
$((Mg)_3Si_2O_5(OH)_4)$	Amiante chrysotile
MLM	Mixed linear models
mm	Millimètre
Mn	Manganèse
mol m^{-2} s^{-1}	Mole par mètre carré par seconde
N	Azote
NH_4^+	Ion ammonium
NO_3^-	Ion nitrate

P	Phosphore
Pb	Plomb
PO_4^{3-}	Ion phosphate
PRS	Plant Root Simulator
RP	Red pine
rpm	Revolution per minute
S	Soufre
SLA	Specific leaf area
VBPD	Vienna Pee Dee Belemnite
WS_n	White spruce (north)
WS_s	White spruce (south)
WUE_i	Intrinsic water use efficiency
Zn	Zinc
$\delta^{13}C$	Rapport de l'isotope 13 de carbone sur l'isotope 12 de carbone
µm	Micromètre
µmol CO_2 mol^{-1}	Micromole de CO_2 par mole
µmol m^{-2} s^{-1}	Micromole par mètre carré par seconde
µmol s^{-1}	Micromole par seconde
°	Degré
°C	Degré Celsius
124	M × B - 102216
216	M × N - 103124
508	DN × M - 915508

Don’t let the muggles get you down

- J.K. Rowling

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Introduction générale

L'exploitation des ressources minières, dont les origines remontent aussi loin que le début de nos civilisations, a marqué les paysages, les écosystèmes et l'environnement, avec des conséquences aussi locales que globales (Hong et al., 2018; Otte et Jacob, 2008; Rodrìguez-Pérez et al., 2018). D'un côté, on observe l'altération, voire la destruction, d'habitats naturels, alors que d'un autre côté, les déchets miniers sont peu propices au rétablissement des écosystèmes vu leur contenu en métaux toxiques, ainsi que leur faible capacité de rétention des nutriments et de l'eau (Cooke et Johnson, 2002; Otte et Jacob, 2008). En Amérique du Nord, une large proportion des activités minières et énergétiques prend place en zones forestières (Macdonald et al., 2015). Vu l'importance de ces écosystèmes pour les services rendus ainsi que pour leur rôle dans l'atténuation des changements climatiques, il est pertinent de voir à la restauration écologique de ces sites miniers dans les milieux forestiers (Ciccarese et al., 2012; Griscom et al., 2017). Les points importants du processus de restauration sont le retour d'une productivité, d'une structure, d'une autonomie et de fonctions similaires à celles d'origine ; le retour d'une composition en espèces indigènes et d'une diversité végétale se rapprochant de celles de l'écosystème préminier ; ainsi que la production viable de services écosystémiques (Zipper et al., 2011; Macdonald et al., 2015).

Les impacts des activités minières sur les écosystèmes forestiers varient en superficie, en intensité, en durée, et aussi selon le type d'exploitation, par exemple l'extraction de minerais, de ressources fossiles ou les carrières (Wong, 2003). De manière générale, on observe une perte totale de la végétation, une restructuration du paysage par l'excavation et par l'empilement de résidus miniers de texture grossière (à une échelle spatiotemporelle variable), ainsi qu'une destruction des sols et de la matière organique et des microorganismes qu'ils contiennent (Cooke et Johnson, 2002). La quasi-absence de matière organique et de microorganismes dans les résidus miniers laissés en place après l’exploitation entraine une incapacité de l’écosystème à stocker et recycler les nutriments et l’eau qui sont essentiels au rétablissement de la végétation (Larney et Angers, 2012). La contamination des sols et des cours d’eau par des métaux toxiques ainsi que le débalancement du pH et de la salinité sont également fréquents, surtout lors de l’extraction de minerais et d’énergies fossiles. Ces conditions contraignent la régénération naturelle des écosystèmes forestiers, principalement en inhibant les processus de formation des sols et la croissance de la végétation. De plus, vue l’importance des perturbations environnementales causées par les activités minières, la restauration écologique peut être elle-même un objectif inatteignable, puisqu’il peut être impossible de rétablir l’écosystème d’origine. Le cas échéant, d’autres pratiques, comme la décontamination, la remise en état, le (re)boisement ou la réhabilitation, peuvent être tout aussi pertinents pour limiter les enjeux environnementaux et de santé publiques (Lima et al., 2016).

Dans le cas des mines de chrysotile (minéral communément appelé amiante), les résidus miniers laissés en place peuvent arborer des concentrations en magnésium, en nickel, en aluminium, en chrome, en manganèse ou en cobalt qui atteignent des niveaux toxiques pour les plantes et certains autres organismes (Belanger et al., 1986; Trivedi et al., 2004; O'Dell et Claassen, 2009). Les haldes d’amiante engendrent également des problèmes environnementaux particuliers. Bien que l’extraction et l’utilisation de l’amiante soient interdites au Canada depuis 2011, les 135 années d’exploitation de l’amiante dans le secteur de Thetford Mines ont généré plus de 800 millions de tonnes de résidus miniers amiantés sous forme de haldes (Ministère de l'Environnement et de la Lutte contre les changements climatiques, 2019).

Les fibres d’amiante contenues dans les résidus miniers sont facilement transportables par le vent et par l’eau en raison de leur petite taille (Schreier, 1987). Jacques et Reinhard (2021) ont trouvé des traces de contamination du réseau hydrographique de la rivière Bécancour par les résidus miniers amiantés à plus de 25 km en aval des haldes minières de Thetford Mines, de même que dans un lac en amont du secteur minier. Les résidus amiantés peuvent altérer la qualité de l'eau et des sédiments de lac et de rivière en augmentant le pH ainsi que la concentration d'éléments traces associés au chrysotile comme le magnésium, l'aluminium, le chrome et le nickel (Schreier, 1987; Jacques et Pienitz, 2021). Il est également possible que les fibres d'amiantes contenues dans les sédiments affectent le comportement et le développement de certains animaux aquatiques et benthiques, et puissent aussi causer un stress oxydatif et une phytotoxicité pour certaines algues et macrophytes (Belanger et al., 1986; Trivedi et al., 2004).

Au Québec, en vertu de la Loi sur les mines, les territoires ayant subi des travaux d'exploration ou d'exploitation minière doivent être décontaminés, réhabilités, remis en état ou restaurés afin, entre autres, de réduire au minimum les impacts sur l'environnement et d'éliminer les risques pour la santé et la sécurité des citoyens (Ministère de l'Énergie et des Ressources naturelles, 2016). Outre les raisons légales, la remise en état de sites miniers peut être encouragée par le désir des populations locales de mitiger les impacts sur l'environnement limitrophe (Lévesque et al., 2020). Le rétablissement de la végétation sur les haldes de résidus miniers peut permettre de réduire l'érosion et la contamination des eaux de surface et souterraine ainsi que de stabiliser les résidus miniers (Wong, 2003; Avera et al., 2015; Groupe de concertation des bassins versants de la zone Bécancour, 2021). Toutefois, il est souvent nécessaire de remettre en état, principalement par des traitements de fertilisation, les propriétés physiques, chimiques et biologiques des sols afin que ceux-ci puissent soutenir la végétation (Cooke et Johnson, 2002; Larney et Angers, 2012).

Les sols résultant de la remise en état des sites miniers sont qualifiés de « technosols » en raison de leurs matériaux parentaux anthropiques (organiques ou minéraux) dont l'origine peut être technogénique ou naturelle (Monserie et al., 2009). La construction délibérée d'un sol dans un contexte de remise en état permet d'adapter les propriétés du technosol spécifiquement aux besoins de l'écosystème perturbé (Watkinson et al., 2017). Les matières résiduelles fertilisantes peuvent servir de matériaux de choix lors de la construction de technosol, vu leur capacité à améliorer les propriétés physiques, chimiques et biologiques des sols, à améliorer l'apport en nutriments pour la flore et les microorganismes, et en raison de leur abondance et de leur disponibilité au Canada (Larney et Angers, 2012).

Dans le contexte du boisement d'une mine d'amiante en déclassement du sud du Québec, il apparait nécessaire de mieux comprendre les propriétés et la composition des technosols qui favorisent l'établissement et la survie de la végétation plantée afin de mieux orienter les pratiques et d'assurer le succès de la remise en état du site minier. En effet, la valorisation des matières résiduelles fertilisantes dans des opérations de boisement et de végétalisation minière, malgré sa pertinence, n'est pas encore documenté en ce qui concerne les rejets de mines d'amiante. Dans ce mémoire seront présentés le contexte de l'étude au premier chapitre, les travaux de recherche dans le second chapitre, suivi par un retour et la conclusion de celle-ci.

Chapitre 1. Contexte de recherche

1. Remise en état des sols

1.1. Matières résiduelles fertilisantes

La matière organique contenue dans les sols, dérivée principalement de l'apport de litière et de la décomposition de celle-ci, est responsable d'une bonne partie de la fertilité des sols, en plus de jouer un rôle important dans les fonctions de l'écosystème forestier (Turcotte et al., 2009; Sorenson et al., 2011). Elle constitue également une source d'énergie et de nutriments pour les microorganismes du sol. Le retrait des premiers horizons organiques et minéraux des sols forestiers lors de l'exploitation minière engendre une perte totale de la matière organique, altérant par le fait même la structure physique, la flore microbienne, les stocks de nutriments ainsi que la capacité de rétention et la dynamique de l'eau dans les sols (Larney et Angers, 2012; Avera et al., 2015). L'ajout de matière organique via des amendements organiques peut permettre de démarrer les cycles biogéochimiques, grâce au rétablissement des communautés microbiennes, ainsi que d'accélérer les processus pédogénétiques des substrats miniers, soit les résidus (déchets résultant de l'extraction de minerais) et les stériles (roches retirées lors de l'exploitation minière), souvent caractérisés par des textures grossières (Gardner et al., 2010; Macdonald et al., 2015; Antonelli et al., 2018).

Les matières résiduelles fertilisantes sont fréquemment utilisées pour la construction des technosols et la remise en état de sites perturbés. Elles proviennent notamment de trois sources principales, soit agricoles (résidus de récoltes, fumiers), forestières (résidus de bois, boues de désencrage), et urbaines (composts et autres matières municipales biodégradables, biosolides) (Larney et Angers, 2012). L'utilisation de ces matières résiduelles fertilisantes pour rehausser la qualité des sols perturbés est intéressante puisque la plupart sont habituellement destinées à l'incinération ou à l'enfouissement sanitaire. Il s'agit donc d'une façon d'éviter les émissions de gaz à effets de serre habituellement associées à la gestion de ces matières résiduelles (Majumder et al., 2014; Perreault et al., 2017; Faubert et al., 2019; Bélanger et al., 2021). De plus, leurs coûts sont principalement restreints à leur transport et à leur application.

Les boues de désencrage résultent du traitement des eaux industrielles de l'industrie du recyclage du papier. Elles sont principalement composées de fibres de bois ainsi que d'argiles (kaolin), d'encres et autres composés chimiques provenant du processus de recyclage et de désencrage du papier (Chantigny et al., 1999). Leurs fortes teneurs en matière organique récalcitrante en font un amendement dont les bénéfices sont de longue durée (Camberato et al., 2006; Faubert et al., 2016). Bien qu'elles contiennent peu d'azote et de phosphore (Fierro et al., 1999; Faubert et al., 2016), elles sont des sources considérables de calcium. Elles peuvent servir à rééquilibrer les proportions de calcium dans les sols post-miniers par rapport aux autres macronutriments, comme c'est le cas des résidus de l'exploitation d'amiante (chrysotile), très riches en magnésium (O'Dell et Claassen, 2009; Cantin et Chouinard, 2014). Elles peuvent également améliorer certaines propriétés physiques et chimiques des sols comme la densité apparente, la stabilité des agrégats, la capacité de rétention de l'eau, le taux d'infiltration de l'eau et la capacité d'échange cationique (Chantigny et al., 1999; Camberato et al., 2006).

L'usage des boues de désencrage dans les traitements fertilisants est surtout pertinent lorsqu'il est jumelé avec d'autres traitements, comme des fertilisants inorganiques ou d'autres matières résiduelles fertilisantes comme les biosolides (Fierro et al., 1997; Fierro et al., 1999; Filiatrault et al., 2006; Cantin et Chouinard, 2014). L'usage en combiné avec d'autres matières fertilisantes permettrait notamment d'éviter l'immobilisation de l'azote provoquée par le ratio C:N très élevé des boues de désencrage (Camberato et al., 2006).

Les biosolides municipaux, ou boues d'épuration, sont issus du traitement des eaux usées municipales, industrielles et des fosses septiques (Cantin et Chouinard, 2014). La matière organique qu'ils contiennent est principalement composée d'hydrates de carbone, donc des glucides labiles facilement et rapidement utilisés et minéralisés par les microorganismes du sol (Lessa et al., 1996). De plus, les biosolides peuvent mobiliser de grandes quantités d'ions nitrate (NO_3^-) dans les deux premières années suivant l'application (Perreault et al., 2017), favorisant ainsi l'établissement rapide de la végétation. La quantité et le taux d'application sont très importants pour cet amendement organique. En effet, si l'application (en quantité ou en fréquence) dépasse les besoins de la végétation, l'azote et le phosphore (sous forme d'ions nitrate et phosphate, PO_4^{3-}) peuvent être lessivés ou pris en charge par le ruissellement, et ainsi parvenir jusqu'aux eaux souterraines et de surface (Samaras et al., 2008). À de fortes teneurs, ces deux éléments sont responsables de l'eutrophisation des eaux de surface. De plus, pour être utilisés en tant qu'agent de fertilisation, les biosolides doivent répondre à des critères de désinfection et de qualité microbiologiques stricts (Ministère du Développement durable, de l'Environnement et des Parcs, 2005; Hébert, 2015). Finalement, la teneur et la biodisponibilité des éléments traces contenus dans les biosolides, par exemple le cadmium, le plomb, le cuivre ou le zinc, sont variables selon la source et le traitement subit, le pH et le type de sol, ainsi que la capacité des plantes ciblées à puiser ces éléments toxiques (Smith, 2009). Dans certains cas, une teneur trop élevée dans les biosolides utilisés à des fins de fertilisation peut engendrer un stress pour les arbres plantés et donc affecter leur croissance, à court et long terme (Cline et al., 2012).

1.2. Préparation de terrain et microrelief des sols

La méso et microtopographie jouent un rôle critique dans la végétalisation du site. En effet, l'hétérogénéité et la rugosité des surfaces engendrent des conditions de surface variées, créant ainsi des variations dans la température et l'humidité du sol, augmentant la capacité de rétention de l'eau et des nutriments et permettant même de favoriser l'établissement de communautés microbiennes (Macdonald et al., 2015). En aménageant une plus grande variété de caractéristiques de surface, on favorise la colonisation et l'établissement d'une plus grande diversité végétale. Cela permet également de protéger les semis et les jeunes plantes dans les premiers stades de colonisation en réduisant l'impact de certaines perturbations de surface. Ces perturbations comprennent par exemple le ruissellement, l'impact physique de la pluie (effet de battance), ou l'érosion et la déposition sédimentaire (O'Dell et Claassen, 2009).

Lors de la remise en état de sites perturbés, il est souvent nécessaire de recourir à la préparation manuelle ou mécanique des sites afin de créer des microsites de plantation qui favoriseront la survie et la croissance juvénile des arbres. En effet, une préparation adéquate des microsites de plantation peut permettre d'éviter d'importantes pertes économiques résultant d'une réduction de la croissance juvénile ou de la survie des jeunes plants (Bilodeau-Gauthier et al., 2011; Löf et al., 2012). Certaines technologies de remise en état du milieu forestier peuvent servir d'inspiration pour l'aménagement de technosol en milieu minier. Par exemple, l'aménagement de monticules et le contrôle de la compétition interspécifique semblent avoir eu des résultats prometteurs sur le peuplier hybride en Amérique du Nord, soit en serre (Kabba et al., 2007), en contexte de boisement en zones agricoles (Pinno et Bélanger, 2009; Desrochers et Sigouin, 2015) ainsi que pour les plantations à croissance rapide pour la production de bois (Bilodeau-Gauthier et al., 2011; Mc Carthy et al., 2017; Thiffault et al., 2020; Bilodeau-Gauthier et al., 2022). De plus, il est possible de combiner plusieurs méthodes de préparation des sites de plantation afin de maximiser les chances de succès du boisement (Löf et al., 2012).

L'aménagement de monticules consiste généralement en la création de zones de plantation surélevées où les risques d'une accumulation trop importante d'eau, de compétition interspécifique et de dommages aux jeunes plants par la faune sont réduits (Löf et al., 2012; Desrochers et Sigouin, 2015). Les monticules peuvent améliorer certaines propriétés physiques du sol comme la température, la compaction et la perméabilité (Bilodeau-Gauthier et al., 2011). Ils peuvent être aménagés de manière intermittente (buttons) ou continue (andains).

Certaines espèces d'arbres peuvent être sensibles à la compétition sous-terraine pour l'eau, les nutriments et l'espace disponible pour les racines, ou à la compétition hors-sol pour la lumière. Les effets de la compétition sur les arbres peuvent se traduire par une réduction de la croissance (diamètre et hauteur de la tige) et de la reproduction ainsi que par une augmentation de la mortalité (Casper et Jackson, 1997; Kabba et al., 2007; Messier et al., 2009). La végétation compétitrice peut être limitée par des moyens mécaniques (aération du sol, aménagement de monticules, coupe de la végétation avec de la machinerie) ou chimiques.

2. Végétalisation des sites miniers

La végétalisation de sites miniers se concentre sur le rétablissement des communautés végétales sur les sites miniers dégradés et en déclassement. Elle promeut entre autres, et selon les conditions de l'écosystème préminier, le développement de la stratification verticale de l'écosystème, la diversité en espèces végétales et l'importance des espèces indigènes. Le retour du couvert végétal joue également un rôle critique dans le processus de remise en état des sols suite à l'exploitation minière. En effet, la quantité de matière organique générée par le couvert de végétation peut être comparable, après quelques années, à celle attribuée à un amendement organique (Bendfeldt et al., 2001).

Le niveau de perturbation de l'écosystème forestier influence quant à lui le processus de végétalisation, qui, dépendamment de la situation, tendra vers la régénération naturelle ou le boisement (Löf et al., 2019). La régénération naturelle, soit la capacité d'un écosystème perturbé à retrouver sa structure ou ses fonctions de manière autonome, entraine des coûts généralement bien moindres que pour l'établissement d'une plantation puisqu'elle requiert moins de manipulations et de ressources, bien que le processus soit généralement plus long et moins prévisible (Macdonald et al., 2015). Il n'est pas toujours possible de se baser sur la régénération naturelle lors du processus de restauration des mines. D'abord, la taille du domaine minier peut créer des environnements isolés, où la dispersion naturelle des propagules et la colonisation végétale sont extrêmement difficiles vu la distance avec la forêt adjacente. De plus, la possibilité d'avoir recours à la régénération naturelle dépend principalement du niveau de perturbation de l'écosystème. Ainsi, un écosystème forestier très fortement perturbé nécessitera plus d'interventions pour sa restauration ou sa remise en état, souvent plus coûteuses en efforts et en capital, et qui demandent un plus grand niveau de gestion et de surveillance à long terme (Macdonald et al., 2015). Dans ces cas-là, le boisement est souvent nécessaire, soit le fait de planter, à la graine ou en semis, des espèces dans les zones ayant longtemps été dépourvues d'un couvert forestier (par exemple les sites miniers) (Zhang et Stanturf, 2008).

Le succès du boisement des sites miniers repose en partie sur la sélection des espèces qui composent les plantations. D'abord, certaines espèces d'arbres peuvent présenter naturellement des traits leur permettant de s'adapter aux conditions hostiles des sites miniers : besoin en éléments nutritifs relativement peu élevés, tolérance à la sécheresse, capacité à fixer l'azote atmosphérique, croissance rapide, etc. (Borišev et al., 2018). Il est toutefois pertinent de tester divers génotypes au sein d'une même espèce qui présente un fort potentiel pour la remise en état par le boisement, afin de comparer leur taux de survie et de croissance sous différentes conditions (Larocque et al., 2013). D'un autre côté, afin d'assurer la qualité et la stabilité des fonctions écosystémiques de l'écosystème forestier restauré, il est souvent nécessaire d'établir des plantations comprenant diverses espèces (Paquette et Messier, 2010; Aerts et Honnay, 2011). Cette pratique peut également être bénéfique à la productivité des plantations d'arbres (Verheyen et al., 2016; Kambach et al., 2019). De plus, le fait de planter à la fois des espèces pionnières à croissance rapide en mélange avec des espèces d'arbres de stade de succession plus tardif favorise le succès à long terme de la végétalisation des sites miniers (Borišev et al., 2018).

Dans des cas de boisement, la manipulation de la morphologie ou physiologie des graines et semis, en serre ou en laboratoire, peut permettre une meilleure adaptation aux conditions stressantes des environnements miniers (Ahanger et al., 2014; Macdonald et al., 2015). En effet, les espèces plantées et les types de plants utilisés (plants à racines nues, plants en récipients, barbatelles, boutures, plançons) ont une forte importance sur leur performance une fois qu'ils sont plantés sur sols miniers (Wang et al., 2000; Grossnickle, 2012; Liu et al., 2012; Borišev et al., 2018). L'adaptation des plants devrait être planifiée en fonction des limitations du site hôte (rusticité, disponibilité de la lumière, espace disponible pour les racines, potentialité de sécheresse, etc.) (Thiffault et al., 2003; Grossnickle, 2012).

En général, des attributs comme un diamètre de la tige ou une taille du réseau racinaire plus élevés favorisent la survie des semis. En effet, la dimension des jeunes plants lors de la mise en terre peut influencer leur potentiel de croissance ainsi que leur compétitivité dans leur nouvel environnement (Thiffault et al., 2003). La manipulation génétique des plants, par exemple pour surexprimer certains gènes bénéfiques à leur survie dans les conditions hostiles des sites miniers, peut également favoriser le succès de la végétalisation (Borišev et al., 2018). Les caractéristiques optimales des plants diffèrent selon les espèces. Par exemple, Landhäusser et al., (2012) ont démontré que les semis de peupliers possédant les meilleurs taux de survie et de croissance étaient aussi ceux ayant les plus grandes réserves en glucides non structuraux dans les racines et les ratios racine – tige (*root-to-shoot ratio*) les plus élevés.

Les espèces de peupliers (genre *Populus*) sont couramment utilisées dans des initiatives de remise en état de sites perturbés en Amérique du Nord, en raison de leur distribution étalée sur le continent, leur capacité à se reproduire de manière végétative (bouture), leur croissance rapide, leur productivité élevée, ainsi que pour leur tolérance à certains stress comme les inondations ou les sécheresses (Réseau ligniculture Québec, 2011). Les peupliers ont généralement des besoins nutritionnels élevés et répondent souvent très bien aux traitements de fertilisation, et ce, même dans le cas d'amendement par des biosolides contenant une certaine quantité de métaux traces (Cavaleri et al., 2004; Sebastiani et al., 2004; Bilodeau-Gauthier et al., 2022).

Il existe au Québec un programme d'amélioration génétique du peuplier qui a pour but de produire des clones adaptés et tolérants aux conditions bioclimatiques et édaphiques du Québec. L'hybridation des peupliers, ou le croisement entre deux individus d'espèces différentes, permet de générer des individus possédant des caractéristiques (productivité, résistance aux maladies, tolérance à la sécheresse, etc.) supérieures à celles des espèces parentales (Réseau ligniculture Québec, 2011). Il est possible de tirer avantage de cette hétérosis lors de la végétalisation de sites miniers, où les nutriments et l'eau sont généralement limités. Plusieurs études se sont attardées aux différences possibles de tolérance et de stratégie de défense contre la sécheresse, ou encore de potentiel de séquestration de carbone entre hybrides (Marron et al., 2003; DesRochers et al., 2007; Fortier et al., 2010; Larcheveque et al., 2011).

D'autres espèces d'arbres, notamment des conifères comme l'épinette blanche (*Picea glauca*), le pin gris (*Pinus banksiana*), le pin rouge (*Pinus resinosa*) et quelques espèces de mélèzes (*Larix* spp.), sont couramment cultivées et utilisées au Québec pour le boisement de terres perturbées ainsi que pour la production de bois (Thiffault et al., 2003). Leur popularité pour le boisement peut s'expliquer par le fait que les conifères ont généralement des stratégies de croissance bien adaptées aux environnements avec peu de nutriments disponibles. La durée de vie plus longue ainsi que la concentration en nutriments plus faibles de leurs tissus font en sorte que les conifères ont un taux de perte de nutriments moindre (Gower et Richards, 1990; Aerts, 1995). Ces stratégies conservatrices expriment un contraste fondamental avec celles d'espèces d'arbres à croissance rapide comme le peuplier hybride, qui favorisent une acquisition rapide des ressources qui se traduit par une plus grande productivité (Díaz et al., 2004). Même parmi les essences résineuses, il existe des différences de productivité notables qui sont plus ou moins prononcées en fonction de la fertilité du sol, de sa texture et de son drainage. Par exemple, le potentiel de croissance de l'épinette blanche est généralement plus faible, dans l'ordre, que celui du pin gris, du pin rouge et du mélèze (Thiffault et al., 2003; Ouimet et al., 2007; Bélanger et al., 2021). Ceci se traduit également par des réponses variées aux traitements de fertilisation et aux amendements organiques (Shepard, 1997; Brais et al., 2015; Emilson et al., 2019; Bélanger et al., 2021; Makar et Markham, 2021) qui peuvent être observées quelques années après la mise en terre des plants et l’application des traitements de fertilisation (Reid et Watmough, 2014).

3. Succès de la remise en état des sites miniers

Il est nécessaire de rester prudent lors de l'analyse du succès de la remise en état d'un site minier (Mahlum et al., 2018). Dans cet ordre d'idée, les technosols, de même que plusieurs autres caractéristiques des écosystèmes forestiers comme le couvert forestier et la diversité végétale, reviennent très rarement, voire jamais, à leurs conditions préperturbations (Rowland et al., 2009). Cela est dû à l'intensité des perturbations, qui surpasse la résistance et la résilience des sols et des écosystèmes forestiers face aux perturbations (Cooke et Johnson, 2002). Sachant cela, il est tout de même pertinent de concentrer des efforts sur la remise en état des sites miniers forestiers. Ces derniers, même s'ils ne fonctionnent ou ne supportent plus la végétation et d'autres formes de vie de la même manière qu'à l'origine, peuvent tout de même permettre d'améliorer les conditions environnementales locales (ex. reconstitution d'habitats fauniques, stabilisation de contaminants miniers, diminution de l'érosion des dépôts miniers et de la pollution, amélioration de la qualité des eaux de drainage, embellissement du paysage, etc.) (Ciccarese et al., 2012; Avera et al., 2015). Des critères de succès sont habituellement inclus à même la planification du processus de restauration écologique ou de remise en état, afin d'encadrer systématiquement celui-ci (Cooke et Johnson, 2002).

Dans la littérature, plusieurs indicateurs de succès, autant structurels que fonctionnels, sont utilisés afin de déterminer le niveau ou le taux de rétablissement des écosystèmes forestiers (Cooke et Johnson, 2002). Par exemple, certaines conditions et fonctions écosystémiques peuvent servir d'évaluation du succès de la remise en état de sites perturbés, comme la biodisponibilité des nutriments, la composition des communautés végétales, le taux de décomposition de la litière ou le développement de l'horizon organique des sols (Rowland et al., 2009). La survie et la croissance des plantes et la biodiversité peuvent également faire office d'indicateurs de succès (Carvalho et al., 2018). De manière générale, les technosols sont comparés avec ceux de l'écosystème d'origine, ou à défaut, d'écosystèmes avoisinants non perturbés, afin de déterminer leur taux de rétablissement ou encore la direction de leur rétablissement (convergente ou divergente aux conditions initiales) (Cooke et Johnson, 2002; Rowland et al., 2009). De plus, l'étude de la nutrition des plantes, notamment par le biais d'analyses de leurs tissus (feuilles, tiges, racines, etc.), peut également agir en tant que proxy de la qualité des sols (Maia, 2012).

Les réserves de carbone dans les sols sont tout aussi reconnues comme étant un proxy de la qualité des technosols ainsi que du rétablissement de l'écosystème post-minier (Frouz et al., 2009; Turcotte et al., 2009). Le potentiel d'accumulation du carbone dans les sols forestiers, ainsi que dans la biomasse végétale, peut d'ailleurs être l'une des principales motivations de restaurer les sites miniers. Effectivement, dans le contexte de la lutte aux changements climatiques, la capacité de séquestration du carbone par les écosystèmes forestiers et les plantations constitue un service écosystémique très recherché (Frouz et al., 2009).

L'usage de biosolides en tant qu'amendement organique dans la remise en état des mines permet d'augmenter la capacité de ces sites perturbés à stocker du carbone dans les sols, même longtemps après l'application (Trlica et Teshima, 2011). Antonelli et al. (2018) ont trouvé que l'importance des puits de carbone augmentait proportionnellement avec le taux d'application (unique) des biosolides utilisés pour la restauration des résidus miniers d'une mine de cuivre et de molybdène de la Colombie-Britannique. Les taux de séquestration de carbone étaient de cinq à neuf fois plus élevés pour les sites miniers restaurés avec des biosolides que pour les sites non amendés aux termes de leur étude étendue sur 13 ans. Alors qu'un gain initial de carbone dans les résidus miniers était initialement dû à l'ajout de biosolides, ce serait plutôt la biomasse végétale qui contribuerait à l'accumulation de carbone organique dans les sols à long terme.

Un autre indicateur de succès de la remise en état de sites miniers peut être le retour d'une composition en espèces végétales ressemblant à celle de l'écosystème préperturbation. Bien que le retour complet de la végétation d'origine (composition, structure et fonction) ne puisse potentiellement se faire sur une échelle de temps humaine, le boisement de sites perturbés peut restaurer de manière satisfaisante certaines fonctions écosystémiques ainsi qu'une partie de la biodiversité d'origine (Chazdon, 2008). Par exemple, la composition en espèces arborées utilisées pour la végétalisation de sites perturbés peut avoir un effet sur la composition et la diversité de la végétation au sol, incluant les pousses d'arbres (Barbier et al., 2008; Collin et al., 2016). Cette influence peut s'expliquer en partie par la litière que génèrent les arbres, qui, selon les espèces, peut être plus ou moins facile à décomposer ou avoir différents effets sur le pH des sols (Carnevale et Montagnini, 2002; Thomaes et al., 2010). Des plantations d'arbres à croissance rapide comme les peupliers peuvent servir de stade de transition entre des milieux perturbés ouverts (friches agricoles laissées à l'abandon, sites miniers, etc.) vers des peuplements forestiers composés d'espèces indigènes (Boothroyd-Roberts et al., 2013). Ainsi, les plantations de peupliers pourraient agir comme des peuplements nourriciers en facilitant, par leur croissance rapide limitant rapidement les ressources en lumière pour la végétation compétitrice, l'établissement d'espèces arborées indigènes ainsi que des espèces de stades de succession plus tardifs (Lust et al., 2001; Carnevale et Montagnini, 2002; Paquette et al., 2008; Boothroyd-Roberts et al., 2013).

4. Objectifs de recherche

Une meilleure intendance des terres est nécessaire afin d'atténuer l'effet des changements climatiques (Griscom et al., 2017). À ce titre, des actions sont requises pour augmenter les puits de carbone et réduire les émissions de carbone en modifiant l'utilisation des terres. La remise en état de sites miniers offre une telle opportunité. Il est possible d'utiliser les matières résiduelles fertilisantes pour reconstruire les sols de ces sites fortement dégradés et ainsi recréer une couverture végétale durable. Toutefois, pour assurer le succès de la démarche, il est nécessaire de développer des technologies pour chaque situation. D'une part, il faut choisir des matières résiduelles fertilisantes et des préparations de terrain bien adaptées aux objectifs et au contexte et, d'autre part, sélectionner des espèces performantes pour bien s'établir sur le site.

Le présent projet de recherche a pour but principal d'identifier des technologies de remise en état qui favorisent le boisement d'une mine d'amiante en déclassement dans la région Chaudière-Appalaches. De ce fait, l'article scientifique présenté au prochain chapitre traite des résultats de deux dispositifs expérimentaux qui visaient à i) identifier des mélanges de matières résiduelles fertilisantes qui favorisent une survie et une croissance optimales des arbres, et ii) identifier des espèces, clones ou provenances qui sont bien adaptées pour le boisement du site minier. Plus précisément, deux plantations expérimentales, une sur résidus et l'autre sur stériles miniers, comprenant deux types de technosols et huit espèces, clones ou provenances d'arbres, ont été établies au sein de la mine. Afin de répondre au premier objectif, la survie et la croissance des arbres ainsi que les propriétés chimiques et physiques des deux technosols ont été étudiées. La nutrition foliaire ainsi que l'efficacité d'utilisation de l'eau des arbres ont été étudiées, en plus de leur survie et de leur croissance, afin de répondre au second objectif.

Chapitre 2. Novel soil reconstruction leads to successful afforestation of a former asbestos mine in southern Quebec: Emphasis on hybrid poplars.

2.1. Avant-propos

Le présent article portant sur le succès de diverses technologies de remise en état par le boisement d'une mine d'amiante en déclassement du sud du Québec a été rédigé par Laurence Grimond sous la supervision de Nicolas Bélanger et de David Rivest, qui ont apporté des commentaires sur les versions préliminaires de ce travail. Pour leurs contributions à cette étude, Simon Bilodeau-Gauthier, Rim Khlifa et Raed Elferjani seront co-auteurs de cet article.

2.2. Introduction

The asbestos mining industry in Canada was mostly active in the province of Quebec, in the southern parts, where about 75% of operations took place (Kuyek, 2003). The last asbestos mine in southern Quebec shut down in 2012 after more than 100 years of activity (Marier, 2016). The typical open-pit mining resulted in a large transformation of the landscape surrounding workers' towns (e.g. Thetford Mines, Black Lake and Val-des-Sources), leaving several millions of tons (estimated at 800) of tailings and waste rocks in large piles (Bureau d'audiences publiques sur l'environnement, 2020). This unique landscape (Figure 1) also raises multiple environmental and socioeconomical concerns. Wind and water erosion of the waste piles can lead to the contamination and the alkalinization of surface waters and sediments (Schreier, 1987; Jacques and Pienitz, 2021), potential health hazards from the asbestos fibres (Bourgault et al., 2014), as well as visual pollution from the piles (Bureau d'audiences publiques sur l'environnement, 2020).

Figure 1. – The open pit of an asbestos mine and piles in southern Quebec. Credits: N. Bélanger

The establishment of vegetation on such surfaces can limit environmental and socioeconomical impacts (Rey et al., 2004; Martín-Moreno et al., 2016). Revegetation can also rehabilitate some ecosystem services such as habitats for biodiversity, nutrient cycling and carbon sequestration, thus acting as a solution for mitigation of climate change (Ciccarese et al., 2012). Yet, in asbestos mining settings, encroachment of the piles by the nearby forests is very limited, if not inexistent, due to the hostile physicochemical nature of tailings and waste rocks (Figure 2). Like in other post-mining settings, natural and autonomous regeneration of the former ecosystems is nearly impossible due to too many physical and chemical constraints for plant establishment and/or growth (Macdonald et al., 2015). The removal of all vegetation and soils during mineral extraction leads to the complete loss of organic matter pools, thus reducing the water-holding capacity of the whole system, disrupting the cycling and availability of essential plant nutrients and altering the activity of soil microbes, which are crucial for nutrient cycling and adding carbon to the soil (Larney and Angers, 2012).

Figure 2. – The plateau of a tailings pile exhibiting no plant growth at an asbestos mine in southern Quebec. Credits: N. Bélanger

In the context of mine reclamation, industrial by-products such as municipal biosolids and composts have been successfully used for soil fertilization or reconstruction and plant restoration (Gardner et al., 2010; Watkinson et al., 2017; Antonelli et al., 2018; Carabassa et al., 2019). Many of these by-products can help soils retain water and provide the necessary structure for plant root development as they are important organic matter (carbon, C) sources, and they also bring about essential macronutrients (e.g. N, P and K) and micronutrients for plant growth (e.g. Cu, Zn and Mn) (Larney and Angers, 2012). When such by-products can be reused instead of being buried in landfills or incinerated, greenhouse gas emissions can be significantly reduced (Majumder et al., 2014; Faubert et al., 2019).

Mine rehabilitation is compulsory in Quebec since 1995 under the Mining Act (RSQ, c. M-13.1). Costs of ecological reclamation are to be covered by owners, but unfortunately, most of asbestos mining operations had ceased when the legal provisions were implemented and as a result, many mines were not submitted to these legal obligations and they were therefore abandoned (Bureau d'audiences publiques sur l'environnement, 2020; Lévesque et al., 2020). Yet, at some asbestos mine sites where some operations are still ongoing (e.g. granite quarry), revegetation of tailings and waste rock piles has been underway. This is generally done by seeding with grasses on soils reconstructed with a mixture of by-products, namely municipal biosolids and deinking sludge. On the one hand, municipal biosolids are rich in labile organic matter and nitrate (NO_3^-) (Larney and Angers, 2012; Perreault et al., 2017). On the other hand, deinking sludge originating from the paper recycling industry contains recalcitrant organic matter and considerable amounts of calcium (Ca) (Camberato et al., 2006). This mixture was shown to be successful for the establishment of a grass cover with no symptom of toxicity (Figure 3), but no attempts for afforestation have been made on these sites thus far.

Afforestation success in such hostile environments is not only a matter of soil reconstruction but is also dependent on species selection. Some tree species may naturally exhibit traits (drought resistance, low nutritional requirements, fast growth, etc.) that allow them to adapt to the harsh conditions prevailing at the mining sites (Borišev et al., 2018). Such tree species in Quebec, e.g. *Picea glauca*, *Pinus banksiana* and many of the *Populus* genus, are part of genetic improvement programs of the provincial government, and are commonly used for reforestation and afforestation of degraded lands (Pinno and Bélanger, 2009; Larchevêque et al., 2015; Guittonny-Larchevêque and Pednault, 2016; Bélanger et al., 2021). Species survival, growth and nutrition, in addition to soil chemical properties, are robust indicators to assess the early success of ecosystem restoration on mine reclamation sites (Bruno Rocha Martins et al., 2020).

Figure 3. – The plateau of a waste rock pile seeded with grasses on a technosol made of a mixture of by-products at an asbestos mine in southern Quebec. Credits: N. Bélanger

Recycling of industrial by-products for the revegetation of asbestos mine spoils is highly relevant, but not yet documented in the scientific literature. In this study, we developed two experimental plantations at a decommissioned asbestos mine in southern Quebec, one on a tailings pile and another on a waste rock pile, as a means to test the performance of eight tree species/clones/provenances planted on two types of reconstructed soils (technosols) and thus potentially propose a suitable approach for afforestation of asbestos mines in southern Quebec. Both technosols were mainly composed of a main mixture of municipal biosolids and deinking sludge, although one also included Class B metal contaminated soils as a means to have a material that maintains its structure (Beaulieu, 2021). Our first objective was to determine which technosol promotes the greatest rates of survival and growth of the planted trees. To do so, physical and chemical properties of both technosols were compared. We hypothesized that the technosol containing Class B soils would yield greater survival and growth by providing more volume for tree roots to acquire water and nutrients. Our second objective was to identify tree species, clones and provenances that are best suited for afforestation under these conditions by assessing, besides survival and growth, water use efficiency and foliar nutrients. We hypothesized that hybrid poplars, a fast-growing species with genetically enhanced features, would perform best regarding survival and growth under the harsh conditions at the mine. We also hypothesized that the most rustic hybrid poplar clones (i.e. M×B-102216) would have greater survival and water use efficiency, but lower growth than other clones.

2.3. Methods

2.3.1. Study site

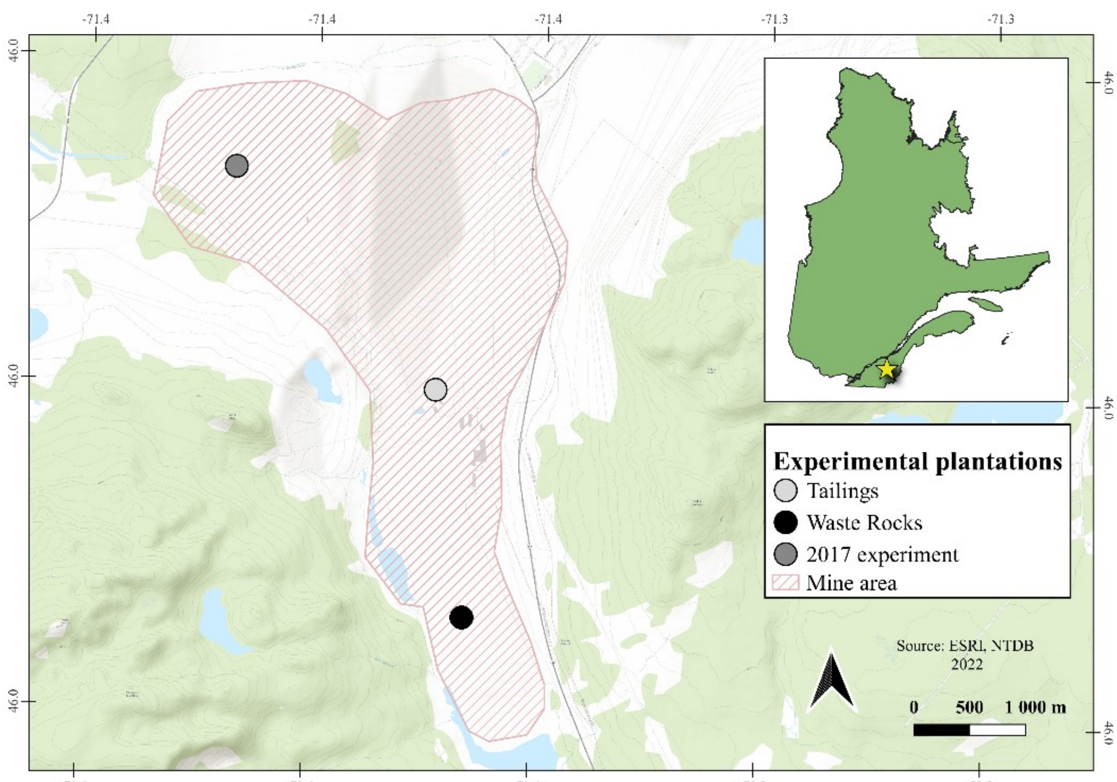

Figure 4. – Location of the experimental plantations within the study site, a decommissioned asbestos mine in southern Quebec

The study was conducted at a decommissioned asbestos mine in Black Lake, southern Quebec, Canada (46°00′48″ North and 71°22′06″ West) (Figure 4). The mine was in operation from 1955 to 2012 for the extraction of serpentine asbestos ($(Mg)_3Si_2O_5(OH)_4$). Since its closure, the mine has been undergoing a reclamation, except for a few areas which were converted into a granite quarry.

The site is part of the Appalachian geological province, more specifically located in the Thetford Mines ophiolite complex of the Lower Ordovician. The study site region includes several rock types, including granite, granodiorite, monzogranite, peridotite and serpentinite (De Souza and Tremblay, 2012). The climate is humid continental. According to the nearest meteorological station (Thetford Mines), the area receives a mean annual rainfall of 945 mm and snowfall of 364 cm, whereas the mean annual temperature is 4.6 °C (1981-2010 period, Environment and Climate Change Canada, 2021). The site is within the sugar maple – yellow birch bioclimatic domain (Government of Quebec, 2019). Soils are typically Eluviated Dystric Brunisols and Orthic ferro-Humic and Humo-Ferric Podzols with a moder type forest floor (Soil Classification Working Group, 1998).

2.3.2. Experimental design

The main experimental design (2018) was developed following an earlier study in 2017, during which we assessed survival rates of five tree species planted in three different microsites. This 2017 experiment consisted of four reclamation sites established on land previously reclaimed between 2012 and 2015 by seeding grasses on soils reconstructed with a mixture of municipal and deinking sludge applied at a rate of 1200 Mg ha^{-1} (300 and 900 Mg ha^{-1}, respectively). This provided an equivalent technosol depth of approximately 20 cm. Technosols were seeded with oats (*Avena sativa*) at a rate of 80 kg ha^{-1} and was used as a nurse plant, whereas a mixture composed by 55 % of Timothy grass (*Phleum pretense*), 30 % of red clover (*Trifolium pretense*), and 15 % of alsike clover (*Trifolium hybridum*) was applied at a rate of 20 kg ha^{-1}. Selected tree species included green alder (*Alnus viridis* subsp. *crispa*), tamarack (*Larix laricina*), white spruce (*Picea glauca*), three cultivars of miyabeana willow (*Salix miyabeana*; SX61, SX64 and SX67) and three hybrid poplar clones (*Populus maximowiczii* × *P. balsamifera*; M×B-915302, M×B-915303 and M×B-915311). These shrubs and trees were planted on three microsites in early June. The first microsite was prepared by excavating 50 × 50 cm mounds with a height of about 30 cm. A second microsite was prepared by applying glyphosate (regulation number 28486, 4.67 L ha^{-1} diluted in 100 L ha^{-1}) to a 30 cm radius around the centre of the planting microsite less than 24 hours before planting. A third microsite consisted of planting cuttings or seedlings without site preparation. In this case, shrubs or trees were planted directly into the grasses and this was considered the control.

After one year, the survival rates of all species on all microsites were greatly unsatisfactory, ranging from 0 to 30% depending on the species, with hybrid poplars, tamaracks and white spruce planted on mounds having the best success. Supplementary material A provides details on survival rates of all species after one and four growing seasons. The previously reclaimed areas are not suitable for planting trees because grasses create too much competition for light, water and nutrients, and roots occupy too much soil space for tree roots to develop (Messier et al., 2009). For this reason, we developed a second experiment in 2018 on mining wastes that were not previously reclaimed. This field experiment is the focus of this study.

The experiment was divided into two 1.5 ha sites, one developed from waste rock and the other from tailings. Soil reconstruction was tested using two mixtures: (1) the same base mixture of municipal and deinking sludge alone (see description above), and (2) the same base mixture with the addition of Class B metal contaminated soils (Beaulieu, 2021). The first technosol is referred to as the B soil, and the second technosol is referred to as the BSC soil. The mixing between the contaminated soil and base mixture was done with an excavator at a volume ratio of 1:1, making it twice as thick as the B technosol. To provide maximum rooting depth and water and nutrient availability, the material was first applied evenly (~20 cm depth for the B technosol, and ~40 cm for the BSC technosol) with a bulldozer. A small excavator was then used to create windrows of ~50 cm (or ~1 m in the case of the BSC technosol) in height and ~75 cm wide with a spacing of 1 m between the rows. Each site included three replication blocks that followed a randomized factorial design as a means to test the effects of the different mixtures on tree survival and growth.

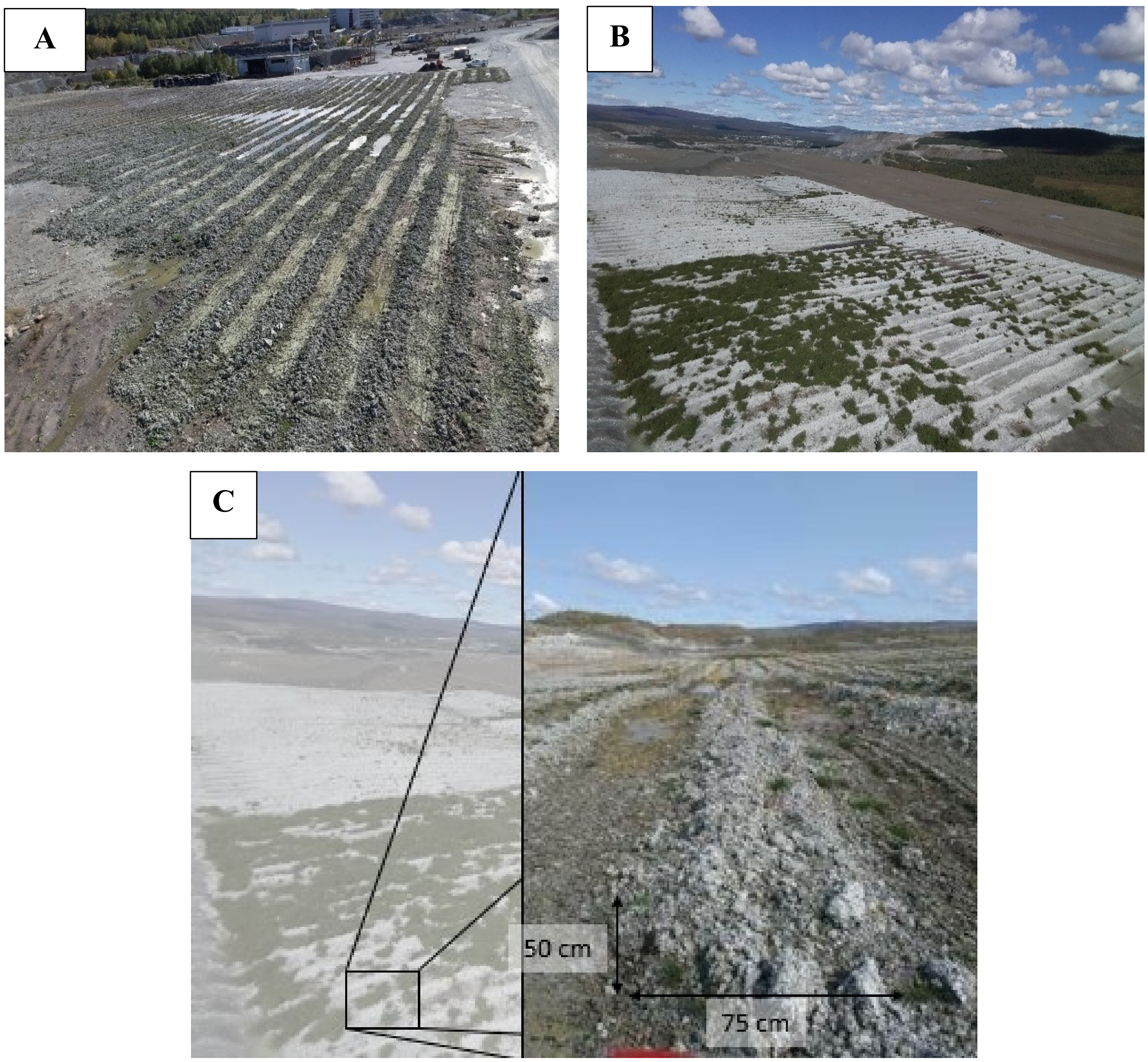

Figure 5. – Aerial view of the technosols constructed on the waste rocks (A) and tailings (B) piles, and close-up view of a fresh windrow on B technosol (C).

Five species were selected for the 2018 experimental plantation, i.e. jack pine (*Pinus banksiana*; JP), red pine (*Pinus resinosa*; RP), two cultivars of white spruce (WSn and WSs, one more rustic than the other), tamarack (TAM), and three clones of hybrid poplar (*Populus maximowiczii* × *P. balsamifera,* M×B-102216; [*P. deltoides* × *P. nigra]* × *P. maximowiczii*, DN×M-915508; and *P. maximowiczii* × *P. nigra*, M×N-103124). Seedling characteristics are provided in Table 1. Seedlings of all species, provenances and clones were planted in monocultures on the top of the windrows from early (waste rock site) to mid (tailings site) June 2018 at a constant spacing of 1 m. Each plot consisted of 49 trees.

Tableau 1. – Nursery stock type of hybrid poplar clones and conifers selected for the 2018 experimental plantation.

Clone or species	Stock type	Container size (cm^3)	Height (cm)
M×N-103124	Bare roots	-	100 - 150
M×B-102216	Bare roots	-	100 - 150
DN×M-915508	Bare roots	-	100 - 150
TAM	Bare roots	-	20 - 30
JP	Container	110	10 - 15
RP	Container	310	20 - 30
WSn	Container	310	30 - 40
WSs	Container	310	30 - 40

M×B-102216 is *Populus maximowiczii* × *P. balsamifera*; DN×M-915508 is *P. deltoides* × *P. nigra]* × *P. maximowiczii*; M×N-103124 is *P. maximowiczii* × *P. nigra*; JP is jack pine; RP is red pine; WSn and WSs are two cultivars of white spruce, the first being more rustic than the second; and TAM tamarack.

The randomized design included three blocks per site. Each block contained 16 plots of trees planted on the B soil and 8 plots of trees planted on the BSC soil (Figure 2). Eight species/provenances/clones were planted per block. Each plot contained 49 trees (i.e. seven rows with seven seedlings each). For soil sampling and analysis, we also added a control plot (where no tree was planted) for each soil mixture per block. Buffers were applied to separate technosols (2 m) and plots (1 m). A total of 144 plots and 7056 trees were planted for the experiment. Competition was minimal in the first year of plantation, but it became significant in 2019 and thus, all trees were brush cleared across a radius of 50 cm around the tree in August of that year. No other means of weed control were used during the study.

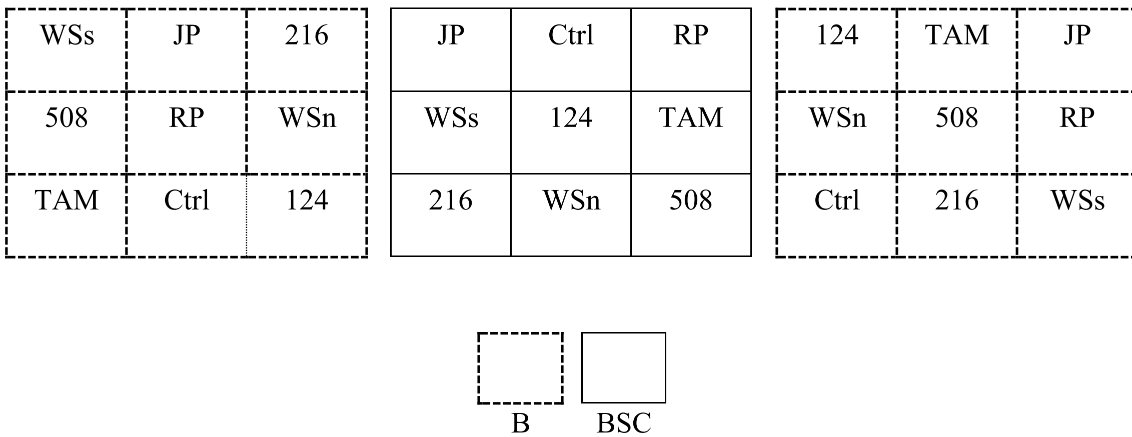

Figure 6. – Example of an experimental plantation block with a randomized factorial design. B is a plot with the basic mixture of biosolids and deinking sludge, and BSC is a plot with the basic mixture with the addition of Class B contaminated soils, 216 is for the M×B-102216 hybrid poplar clone, 508 is for the DN×M-915508 clone, 124 is for the M×N-103124 clone, JP is for jack pine, RP is for red pine; WSn and WSs are for two cultivars of white spruce, the first being more rustic than the second; and TAM is for tamarack.

2.3.3. Soil sampling and monitoring

Soil samples were collected at the end of September 2019. In each control plot of the experimental plantation sites, samples were collected in five random windrows up to a depth of 15 cm using a soil corer with an 8 cm diameter. This procedure yielded 90 samples.

In June 2021, we inserted Plant Root Simulator (PRS) probes (Western AG Innovations) in nine randomly selected windrows of the two experimental plantation sites in order to determine solution ionic activity in the two technosols. A set of probes composed of four anodes and four cathodes was inserted vertically on the uppermost part of each selected windrow at a depth of 0 to 15 cm. We inserted probes in pairs (one anode and one cathode) along the windrow at approximately every 2 m to capture as much variability as possible. The probes were left in the technosols to equilibrate with the soil solution for five weeks. Probes were then extracted and thoroughly cleaned with deionized water, stored in the refrigerator in sealed plastic bags before analysis.

Soil temperature, volumetric water content and water potential were recorded from June to August 2021 with respectively six external temperature sensors, six soil WaterScout SM 100 moisture sensors and six Watermark soil moisture sensors (Spectrum Technologies) placed evenly in three control plots in each plantation site. Sensors were inserted at depths of 10 cm and were connected to a data logger (WatchDog 1000 Micro Station, Spectrum Technologies) that recorded soil temperature and moisture conditions every hour.

2.3.4. Survival and growth

All trees were monitored for their survival. Survival rates were calculated based on the ratio of the number of trees alive and the total number of trees. An annual survey was done in October from 2018 to 2020, but only the results from the 2020 survey (i.e., at the end of the third growing season) will be presented. Considering the slow growth of most conifers planted and the high potential for reclamation of hybrid poplars, early growth was monitored only on the fast-growing hybrid poplar clones. Stem diameter at the collar as well as total height were measured for all living hybrid poplar trees in October of 2019 and 2020, i.e., two and three growing seasons after planting. Only the results taken in 2020 will be presented. Total height was measured using a telescopic measuring pole following the method of Pérez-Harguindeguy et al. (2013) for plant height measurements, and stem diameter was measured using an electronic caliper.

2.3.5. Leaf sampling and monitoring

Due to their fast-growing potential, leaf sampling also focused on hybrid poplar clones. Five trees were randomly selected in each hybrid poplar plot for foliar sampling in mid-August 2020, which is equivalent to the end of the third growing season. To avoid edge effect, sampled trees were selected in the middle of the plots. Ten fully expanded leaves with petiole from the upper third of the crown (i.e., full light) were collected on each of the five trees. This procedure yielded 270 composite samples.

Gas exchange measurements were done using the LI-6800 portable photosynthesis system equipped with the LI-6800-01A fluorometer, a combined light source and a chamber set for a 6 cm^2 leaf area (Licor). Variables of interest, measured or computed by the instrument, were net carbon assimilation rate (A_{net}), stomatal conductance to water vapour (g_{sw}), and intrinsic water use efficiency (WUE_i). CO_2 concentration in the chamber (C_a) was set to 400 µmol CO_2 mol^{-1} using CO_2 cartridges. Leaf temperature was set at 25 °C, relative humidity at 55 %, airflow at 300 µmol s^{-1}, and light intensity was saturated at 1500 µmol m^{-2} s^{-1}. The gas analyzers were calibrated before each measurement. The leaf chamber was clamped onto the leaf so as not to change the orientation of the leaf. Measurements were registered only when A and C_a reached a steady state, which could take between 5 and 15 minutes.

More specifically, gas exchange measurements were taken three times from late June to early July 2021, between 8:30 AM and 12:00 on generally dry and sunny days. Measurements were always made on recently matured leaves that were well exposed to sunlight and that did not present signs of senescence or disease. The order of measurements (clones and treatments) was randomized for each sampling period to reduce possible bias on photosynthesis parameters associated with time of day. Due to time restrictions, measurements were focused in one replication block during each sampling period. To limit edge effect, we did not sample the trees from the border of the plots. As a whole, one leaf of 18 trees (i.e., two per hybrid poplar per plot) could be measured with the LI-6800 each time. As gas exchange measurements require strict meteorological conditions, only the tailings plantation site could be sampled during the summer of 2021.

2.3.6. Laboratory analysis

The PRS probes were eluted in 0.5 M HCl for 1 hour. Inorganic N supply rate (NO_3^--N and NH_4^+-N) was determined colorimetrically by automated flow injection analysis (San Series analyzer, Skalar), while other ion supplies (i.e. Ca, Mg, K, P, Fe, Mn, Cu, Zn, B, S, Pb, Al and Cd) were determined by coupled plasma spectrometry (Optima ICP-OES 8300, PerkinElmer).

Soil samples were first oven dried at 65 °C for 48 hours. The samples were then weighed, and apparent bulk density was measured using the water displacement approach proposed by Boivin et al. (1990) and following Archimedes' principle. Samples were then sieved through a 2 mm stainless steel mesh to measure soil pH in water using a 1:5 soil to water ratio. Remaining samples were finely ground in a planetary ball mill (PM 400 Planetary Ball Mill, Retsch) for 5 minutes at 400 revolutions per minute (rpm) to achieve < 60 µm fineness. Organic C and total N were assessed by combustion at 1040 °C and respectively infrared and thermal conductivity detection with an EA 1108 CHNS-O analyzer (Thermo Fisons).

We used a high-pressure hydraulic press (Reflex Instruments) to produce homogenous sample pellets (10 mm thickness, 26 mm diameter) with 8 to 10 g of ground material for each sample. Soil bulk chemical composition, with concentrations (%) of P, K, Ca, Mg and Al being of the most interest, were measured using a handheld X-ray fluorescence spectrometer (XRF) equipped with a rhodium tube of 50 kV and a silica drift detector (Vanta M series, Olympus). The Geochem calibration mode was used, set with two built-in beam filters used for heavier elements and light elements separately (40 kV and 10 kV, respectively). Each beam filter scanned the sample pellet for 30 seconds, whereas each sample pellet was scanned three times to obtain a more representative average of the chemical composition.

Leaf area of hybrid poplar leaves was determined by scanning leaves one by one using a CI-202 Portable Leaf Area Meter (CID Bio Science). Fresh samples were weighed, then oven-dried and ground as described above. Between 1.7 and 1.8 g of dried leaf material was necessary to produce a sample pellet (10 mm thickness, 13 mm diameter). These pellets were used to determine leaf elemental concentrations (%) of P, K, Ca, and Mg with the Vanta XRF analyzer. The same analytical conditions described above were used, except that the resulting data were corrected following data from five customized P, K, Ca, and Mg internal standards made of cellulose, lignin, xylose and gallic acid to simulate the proportions of cellulose (32%), lignin (28%), hemicellulose (25%) and tannins (15%) of common plant materials or organic soils. The linear regressions yield strong to very strong R^2 (0.91) for all elements of interest. While the use of X-ray spectrometry has become an official method for the chemical characterization of soils and rocks, the approach has recently increased in popularity for plant tissues (Arantes de Carvalho et al., 2018; Borges et al., 2020).

Leaf total C and total N were determined on 2.3 mg of pooled samples (all samples from one plot) with a NC2500elemental analyzer (Carlo-Erba) by combustion, separation of elements by gas chromatography and detection by thermal conductivity. For $\delta^{13}C$ analysis, 1.4 mg of each pooled leaf sample was weighed in individual tin capsules for all samples and reference material to have the same amount of CO_2. A Micromass model Isoprime 100 isotope ratio mass spectrometer coupled to a Vario MicroCube elemental analyzer (Elementar) in continuous flow mode was used to conduct the $\delta^{13}C$ analysis. Three internal reference materials ($\delta^{13}C$ = -28.73 ± 0.06 ‰; $\delta^{13}C$ = -11.85 ± 0.04 ‰; $\delta^{13}C$ = -17,04 ± 0,11‰) were used to normalize the results on the NBS19-LSVEC scale. Results are expressed in delta (δ) unit as a part per mil (‰) of VBPD (Vienna Pee Dee Belemnite), and the overall analytical uncertainty is better than ± 0.1‰.

2.3.7. Data analysis

A meta-analysis of PRS data to assess the bioavailability of chemical elements in soils worldwide was used to compare the studied technosols to forest soils worldwide (Ochoa-Hueso et. al submitted). In this case, we compared the PRS data in the two technosols at the mine with the average soil solution ionic activities in forest soils. In addition, a meta-analysis conducted by Camiré and Brazeau (1998) on the nutritional requirements of *Populus* spp. was used to assess if the hybrid poplar trees from our study were above or below minimal nutritional requirements. Foliar nutrient thresholds developed by Hansen (1994) and Coleman et al. (2006) for hybrid poplar plantations in north-central United States were also used as proxies for optimal nutrition.

All statistical analyses were conducted in the software R version 4.0.4 (R Core Team, 2021). Descriptive statistics were used to characterize and compare soil properties and nutrients. Two-way ANOVA (analysis of variance) was conducted to check for significant differences in soil variables between the two technosols and among sites and replication blocks, as well as their interaction. Site and the blocking variable were added to our model to consider any effect of the location of control plots where soil samples were taken. When the conditions of independence of the observations, normality of the residuals and homoscedasticity were not met, data were transformed appropriately. Results (p-value) from the Shapiro-Wilk are reported to show which variables followed a normal distribution after transformations.

Mixed effect linear models (MLM) were conducted to test the effect of technosol type and clone or species on survival rates, growth variables (including specific leaf surface area), foliar nutrients and gas exchange variables. Blocks and sites (tailings or waste rocks) were added as a random effect in the models. The function *lmer* from the *lme4* R was used package (Bates et al., 2015). Again, data was transformed when conditions of normality of residuals, independence and homoscedasticity were not met. Tukey's HSD test (*glht* function in the *multcomp* R package) was used to assess differences between clones (Hothorn et al., 2008).

2.4. Results

As the site variable (tailings and waste rock) did not add any significance to the mixed linear models or analysis of variance conducted for this experiment, it is not presented or discussed here.

2.4.1. Physicochemical properties of technosols

Physical and chemical properties of the two technosols are presented in Table 2, whereas two-way ANOVA test results are shown in Table 3. As a whole, the technosols were not very different from each other. For example, pH levels of B and BSC soils were identical at 8.00, Cu and Pb levels were identical at 0.1 % and 0.002 %, respectively, bulk densities were respectively 612 and 628 kg m^{-3}, C levels were 17.4 % and 15 %, Ca levels were 23.6 and 22.7 %, and Al were 2.68 and 2.92 %. Nitrogen levels, C:N ratios, P, K, M and Zn levels exhibited greater divergence between the technosols, but only K and Zn differed significantly between the technosols, with the BSC soil exhibiting a greater K concentration and a lower Zn concentration. The significant interaction term for Ca concentrations suggests that there is a significant effect of the block as a function of technosol type, whereas Zn concentrations also seemed to vary based on the blocking structure, not just the technosol type.

Tableau 2. – Physical and chemical soil properties (means and standard deviations) of the two technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils). Relative differences (variation rate) between the two technosols are negative when BSC soil values are lower than B soil values, and vice versa.

Soil property	Technosol		
	B	**BSC**	**Variation rate (%)**
pH in water	8.00 ± 0.33	8.00 ± 0.15	0
Bulk density (kg m^{-3})	612 ± 110	628 ± 96.9	2.61
Carbon (%)	17.4 ± 5.16	15.0 ± 1.83	-13.8
Nitrogen (%)	0.67 ± 0.20	0.52 ± 0.10	-22.4
C:N [1]	26.3 ± 2.99	29.4 ± 6.09	11.8
Phosphorus (%)	0.33 ± 0.07	0.26 ± 0.1	-26.92
Potassium (%)	0.04 ± 0.07	0.24 ± 0.23	500
Calcium (%)	23.6 ± 2.93	22.7 ± 3.77	-3.81
Magnesium (%)	1.49 ± 0.47	1.86 ± 0.78	24.8
Copper (%)	0.01 ± 0.003	0.01 ± 0.002	0
Lead (%)	0.002 ± 0.0007	0.002 ± 0.001	0
Zinc (%)	0.03 ± 0.005	0.02 ± 0.005	-33.3
Aluminum (%)	2.68 ± 0.39	2.92 ± 0.65	8.96

[1] C:N is the ratio of total carbon on total nitrogen.

Tableau 3. – Two-way ANOVA test results for physical and chemical soil properties as a function of technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils) and replication blocks within the experimental plantations. Values with a significant effect ($p < 0.05$) are indicated by the * symbol.

Soil property	**ANOVA**				
	Transf.[1]	**Shapiro**[2]	**Technosol**	**Block**	**Interaction**
pH	transposed	0.001 *	0.914	0.695	0.917
Bulk density	-	0.729	0.758	0.257	0.780
Carbon	√	0.013 *	0.447	0.511	0.877
Nitrogen	-	0.922	0.118	0.319	0.379
C : N	-	0.134	0.099	0.363	0.094
Phosphorus	-	0.101	0.113	0.615	0.321
Potassium	³√	0.210	0.01 *	0.104	0.062
Calcium	-	0.088	0.231	0.018 *	0.014 *
Magnesium	log	0.087	0.150	0.558	0.068
Copper	-	0.131	0.449	0.120	0.382
Lead	-	0.226	0.543	0.087	0.161
Zinc	-	0.210	0.028 *	0.003 *	0.407
Aluminum	-	0.084	0.148	0.052	0.081

[1] Transformation used on data to help meet the criteria of a normal distribution

[2] Results (p-value) from the Shapiro-Wilk test are to show which variables followed a normal distribution after transformations. Variables with a significant effect ($p < 0.05$) are not normally distributed.

The cationic and anionic activities of the soil solutions of the two technosols are shown in Table 4 along with one-way ANOVA test results. No ionic activity significantly differed between technosols. Nitrogen availability in both technosols was driven by nitrifying bacteria as almost 100% of ionic N was in the form of NO_3^-. Nitrate-N, K, Mg and Cu activities were higher in the BSC soil, but the high variability and the small number of probe samples likely prevented detecting significant differences. For most of the soil elements presented in Figure 7, ionic activities of technosol soil solutions were below forest soil values provided by the meta-analysis of Ochoa-Hueso et al. (submitted). Only P, Cu and Pb soil solution activities were above the mean activities of forest soils worldwide.

Tableau 4. – Anionic and cationic activities of the soil solution (means and standard deviations) measured in the two technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils) using Plant Root Simulator (PRS) probes incubated for five weeks in June 2021. One-way ANOVA shows no significant differences between technosols. Relative differences (variation rate) between the two technosols are negative when BSC soil values are lower than B soil values, and vice versa.

Ion	**Technosol ion supply (µg 10 cm^{-2} 35 days^{-1})**		
	B	**BSC**	**Variation rate (%)**
Total nitrogen	232 ± 93	264 ± 105	13.8
NO_3^--N	235 ± 93.5	263 ± 106	11.9
NH_4^+-N	0.36 ± 0.24	0.47 ± 0.42	30.6
PO_4^{3-}	19.6 ± 17.7	16.9 ± 10.3	-13.8
K^+	26.2 ± 28.6	66.1 ± 101	152.3
Ca^{2+}	2619 ± 339	2584 ± 312	-1.33
Mg^{2+}	251 ± 119	297 ± 84.3	18.3
Cu^{2+}	6.28 ± 4.33	9.42 ± 7.37	50
Zn^{2+}	0.67 ± 0.33	0.81 ± 0.67	20.9
Pb^{2+}	16.4 ± 9.79	17.0 ± 9.02	3.66
Al^{3+}	30.4 ± 10.3	30.4 ± 7.03	0

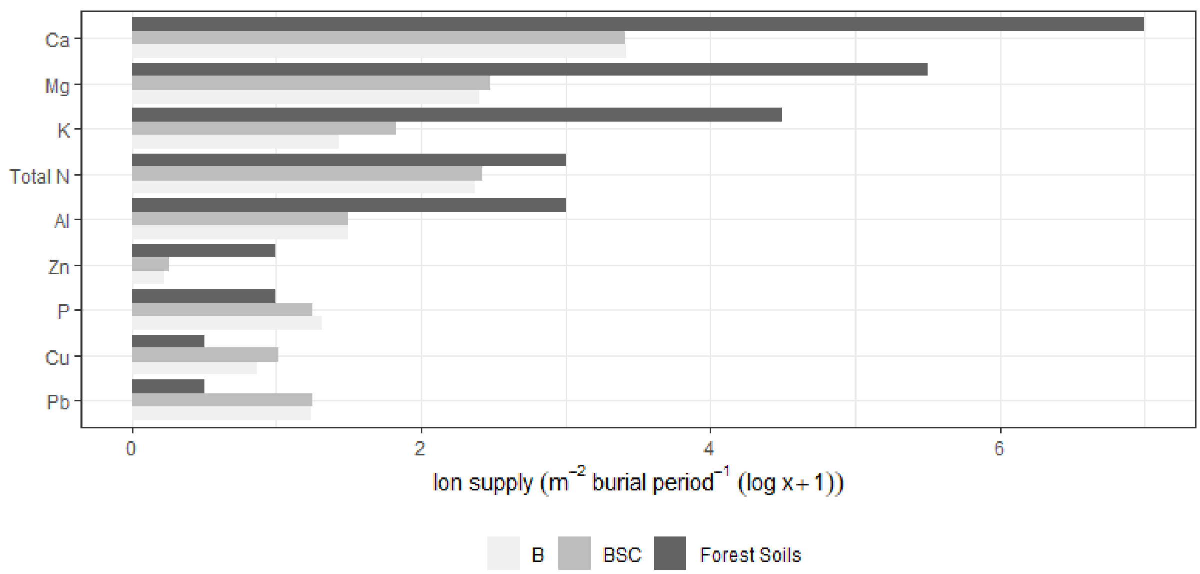

Figure 7. – Bar plot of the anionic and cationic activities of the soil solution measured in the two technosols (B is for the basic mixture of biosolids and deinking sludges; BSC is for the basic mixture with the addition of Class B contaminated soils) in comparison to the mean activities of forest soils worldwide reported in a meta-analysis by Ochoa-Hueso et al. (submitted).

2.4.2. Survival and growth

Survival rates of conifers and hybrid poplar clones are presented in Table 5 and Figure 8. We can observe two main patterns. The first pattern is a large difference in survival rates between conifers and hybrid poplar clones. Survival rates were much greater for hybrid poplars, independently of clones (86.9% to 94.4%), than conifers (10.2 to 56.1%). The large variability in conifer response can be explained by a very low survival rate of jack pine (10.2 to 18.5%) and a higher survival rate of white spruce (46.3 to 56.1%). Survival rates of red pine and tamarack were between those of jack pine and white spruce (25.2 to 37.6%). The difference between conifers and hybrid poplar clones was statistically significant (Table 6). There was also a significant difference of jack pine survival rates when compared to white spruce provenances ($p < 0.001$) and tamarack ($p < 0.05$), and of survival rates between red pine and the northern white spruce provenance ($p < 0.05$) (results not shown). No difference in survival rate was detected between hybrid poplar clones. The second pattern is higher survival rates under the BSC soil for many of the tree species.

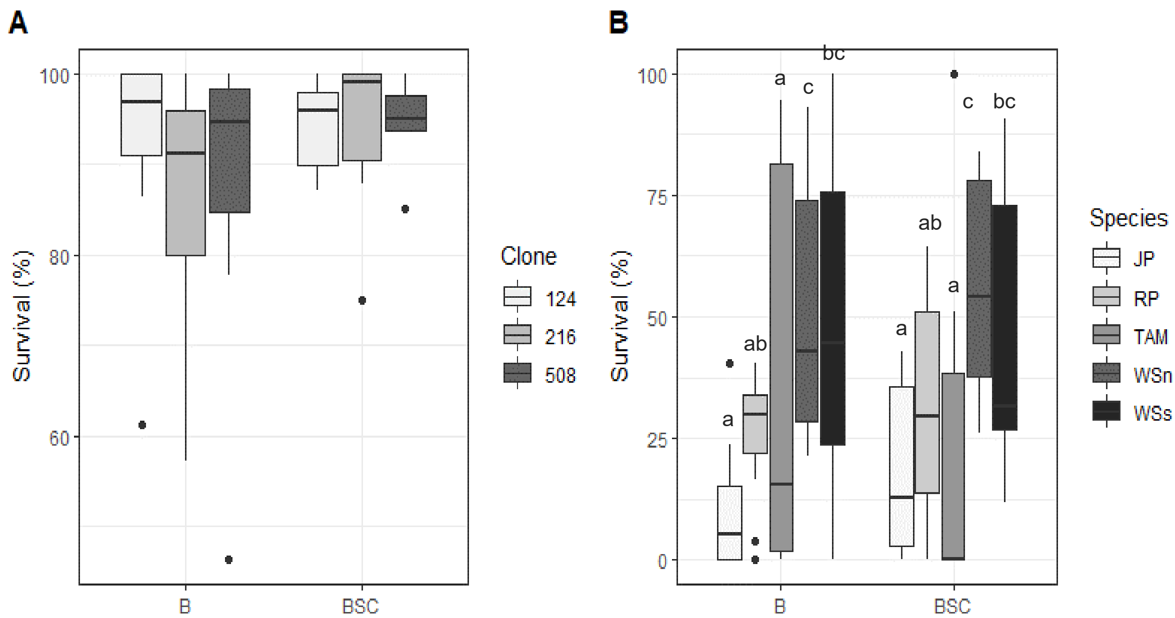

Figure 8. – Box and whisker plots of survival rates of the three hybrid poplar clones (A) and five conifer species/provenances (B) growing on the two technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils). Clone 124 is M×N-103124, clone 216 is M×B-102216 and clone 508 is DN×M-915508. WSn and WSs are the northern and southern white spruce provenances, TAM is tamarack, JP is jack pine and RP is red pine. The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots. Within each technosol, different letters indicate significant differences among tree species.

Although some variations are small, survival rates of jack pine (+ 81.4 %), red pine (+ 22.8 %), white spruce (northern provenance) (+ 12.2 %) and all hybrid poplar clones (+ 1.64 %, + 6.24 % and + 7.59 % for the 124, 508 and 216 clones, respectively) were all higher under the BSC soil. Survival rates for tamarack were lower under the BSC soil (- 33.3 % of variation), whereas they were similar between the two technosols for the southern white spruce provenance (- 1.11 % of variation).

Tableau 5. – Survival rates three growing seasons after planting of conifers and hybrid poplars planted on two technosol types (B, basic mixture of biosolids and deinking sludge; BSC, basic mixture with the addition of Class B contaminated soils). Values are calculated based on the ratio of the number of trees alive on the number of trees monitored. Relative differences (variation rate) between the two technosols are negative when BSC soil values are lower than B soil values, and vice versa.

Species	Technosol		
	B	**BSC**	**Variation rate (%)**
Pinus banksiana	10.22 %	18.54 %	81.4
Pinus resinosa	25.80 %	31.69 %	22.8
Larix laricina	37.56 %	25.17 %	-33.0
Picea glauca (S)	46.78 %	46.26 %	-1.11
Picea glauca (N)	49.98 %	56.07 %	12.2
M×N-103124	92.71 %	94.23 %	1.64
M×B-102216	86.86 %	93.45 %	7.59
DN×M-915508	88.84 %	94.39 %	6.24

Stem height and diameter at collar of hybrid poplar clones are shown in Figure 8. All clones reached a median value of at least 2 m in height after three full growing seasons. Median total height values were above 225 cm for most clones and technosols, except for clone 216 under the B soil. The highest total height growth was with clone 124 under the BSC soil, reaching a median value of 270 cm. Under all scenarios (clones and technosols), some trees reached well above 300 cm after three growing seasons. The BSC soil produced significantly greater height growth than the B soil (Table 6). However, there was no significant clonal effect detected.

After three growing seasons, median values of diameter at collar were near 40 mm for all clones under the B soil and for clone 216 under the BSC soil, whereas median diameters were 48 to 50 mm for clones 124 and 508 under the BSC soil (Figure 9). Again, the BSC soil produced significantly greater diameters than the B soil, but there was no significant clonal effect detected (Table 6).

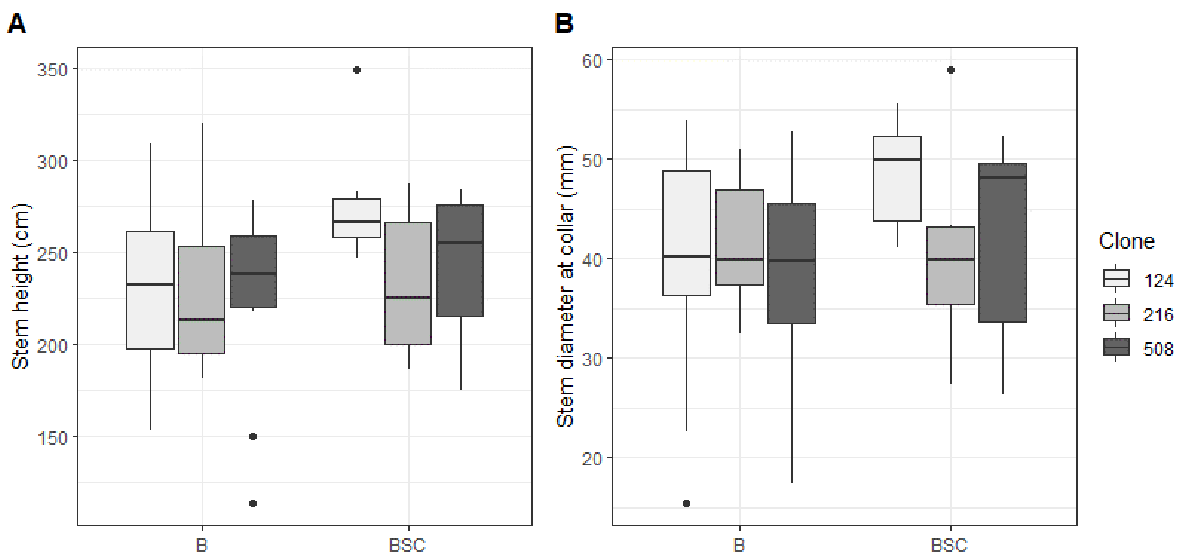

Figure 9. – Box and whisker plots of stem height (A) and stem diameter at the collar (B) of the three hybrid poplar clones growing on the two technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils). Clone 124 is M×N-103124, clone 216 is M×B-102216 and clone 508 is DN×M-915508. The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots.

2.4.3. Foliar nutrients

Foliar nutrients are presented in Figure 10, whereas mixed linear model results are shown in Table 6. Foliar N, Ca and Mg concentrations were generally above minimum nutritional requirements proposed by Camiré and Brazeau (1998), whereas foliar K concentrations were well below the minimum requirement. Median foliar P concentrations were also below the minimum requirement, but many trees of all clones and technosols were above that threshold. All nutrients except Ca exhibited foliar concentration medians that were below the optimal nutrition thresholds developed by Hansen (1994) and Coleman et al. (2006). There was a clonal effect for foliar N concentrations but no technosol effect (Table 6). This was due to significantly higher foliar N concentrations for clone 508. There was a significant technosol effect for foliar Ca, Mg and K concentrations. Foliar Ca and K concentrations were higher under the BSC soil, whereas foliar Mg concentrations were higher under the B soil. For foliar Ca concentrations, there was also a significant clonal effect, with clone 216 showing the greatest concentrations and clone 508 showing the lowest concentrations under both technosols. There was a marginal clonal effect for foliar Mg concentrations, with clone 124 showing greater concentrations than the other clones under both technosols (Table 6).

Tableau 6. – Mixed linear model results for survival rates, growth variables (height, diameter and leaf area) and foliar nutrients as a function of technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils) and clone or species within the experimental plantations. Values with a significant effect ($p < 0.05$) are indicated by the * symbol.

Response variable	Mixed linear models			
	Transf.[1]	Shapiro[2]	Technosol	Clone or species
Survival rate (conifers)	√	0.0001	0.963	< 0.001 *
Survival rate (hybrid poplars)	exp. 3	0.000007	0.112	0.338
Survival rate (all)	-	< 0.001	0.491	< 0.001 *
Stem height	-	0.936	0.023 *	0.167
Stem diameter at collar	-	0.079	0.033 *	0.488
Nitrogen	-	0.058	0.849	< 0.001 *
Phosphorus	transposed	0.003	0.967	0.103
Calcium	log	0.480	0.007 *	< 0.001 *
Magnesium	transposed	0.056	0.023 *	0.085
Potassium	-	0.271	< 0.001 *	0.570
Specific leaf area	-	0.575	0.221	< 0.001 *
Water use efficiency ($\delta^{13}C$)	-	0.898	0.387	< 0.001 *
$_{i}$WUE[3] (normal conditions)	-	0.082	0.278	0.008 *
$_{i}$WUE[3] (drier conditions)	-	0.166	0.608	0.322

[1] Transformation used on data to help meet the criteria of a normal distribution

[2] Results (p-value) from the Shapiro-Wilk test are to show which variables followed a normal distribution after transformations.

[3] $_{i}$WUE is for intrinsic water use efficiency (A_{net} : g_{sw} ratio)

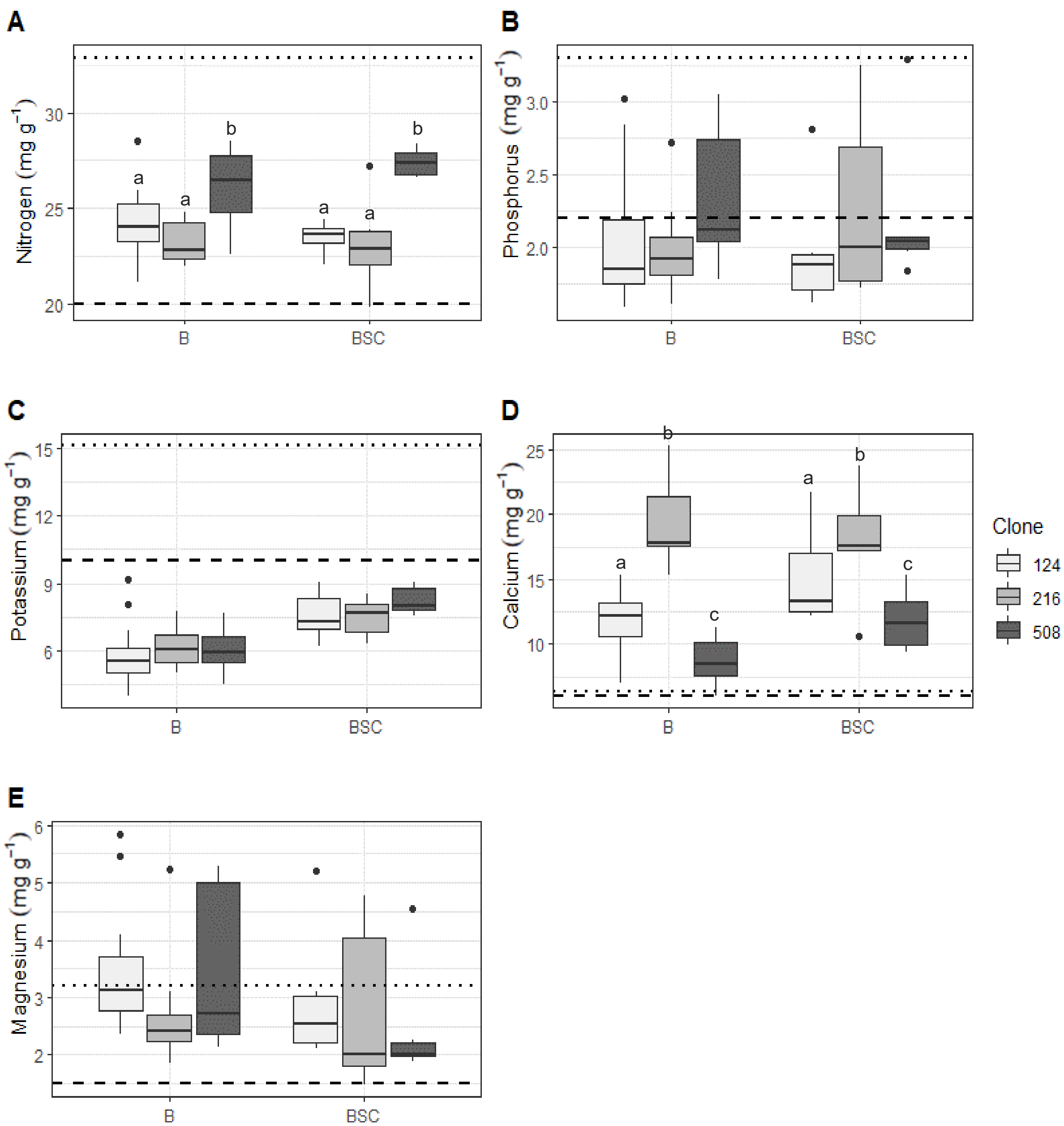

Figure 10. – Box and whisker plots of foliar nitrogen (A), phosphorus (B), potassium (C), calcium (D) and magnesium (E) levels of the three hybrid poplar clones growing on the two technosols (B is the basic mixture of biosolids and deinking sludge; BSC is the basic mixture with the addition of Class B contaminated soils). Clone 124 is M×N-103124, clone 216 is M×B-102216 and clone 508 is DN×M-915508. The horizontal dashed line is the minimal nutritional requirement level for Populus spp. (Camiré and Brazeau 1998), whereas the horizontal dotted line is the optimal nutritional level for hybrid poplars (Hansen, 1994; Coleman et al., 2006). The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots. Within each technosol, different letters indicate significant differences among hybrid poplar clones.

2.4.4. Soil water and temperature

Variations in soil temperature and water from June 15th to July 10, 2021, are presented in Figure 11. Soil temperature varied from about 14°C to 22°C. Patterns and values of soil temperature were very similar between soil types and sites (i.e. waste rocks or tailings). There was greater divergence in patterns and values of soil water potential and volumetric water content between soil types and sites, especially for soil water potential. Overall, data suggest that two gas exchange measurements with the LI-6800 were done under dry conditions (June 17th and June 25th) and under wetter conditions (a few days only after a large rain event on July 5th). A general decrease in soil water potential and an increase in soil water content can be observed due to the rain event. For example, the B soil at the waste rock site moved from a water potential of 200 kPa (very dry) to about 20 kPa after the July 5th rain event. A similar pattern was observed for soil volumetric water content, moving from 2% to 12 % in the B soil at the tailing site. Changes in water potential and volumetric water content were not as large for other soils – yet they followed a similar pattern. On July 5th when gas exchange measurements were done, soil water potential varied from 25 kPa to 75 kPa, and soil volumetric water content ranged from 2.5 % to 6%. Since some measuring probes and loggers were damaged and some batteries failed, not all result curves are shown in Figure 11.

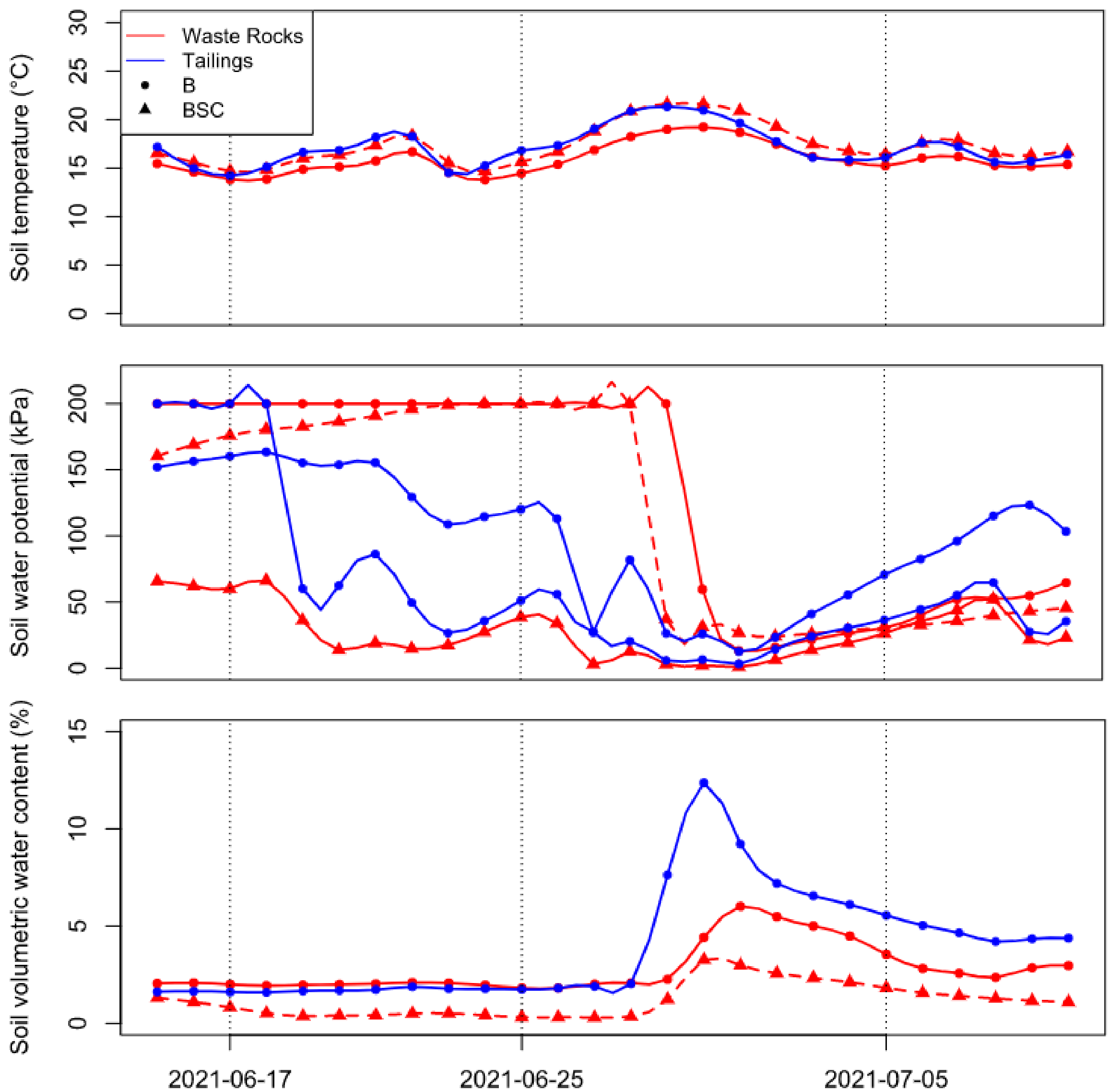

Figure 11. – Daily soil temperature, water potential and water content measured in the technosols (B is for the basic mixture of biosolids and deinking sludge; BSC is for the basic mixture with the addition of Class B contaminated soils) at waste rocks (red lines) and tailings (blue lines) from June 15th to July 10, 2021. Dotted grey lines are days of gas exchange measurements. Since some measuring probes and loggers were damaged and some batteries failed, not all result are shown.

2.4.5. Specific leaf area, foliar $\delta^{13}C$ and water use efficiency

Specific leaf area and foliar $\delta^{13}C$ are presented in Figure 12 and mixed linear model results are shown in Table 7. Mean specific leaf area ranged between 81 and 147 cm^2 g^{-1}, whereas $\delta^{13}C$ ranged between -29.16 and -26.27. Clone 508 had significantly higher median specific leaf area, reaching just under 120 $cm^2\ g^{-1}$. Clone 216 showed a greater variation of specific leaf area values, ranging from 95 $cm^2\ g^{-1}$ to 110 $cm^2\ g^{-1}$ under the BSC and the B technosol, respectively. However, we did not detect a significant technosol effect on specific leaf area. All clones had significantly different foliar $\delta^{13}C$, with clone 216 exhibiting the less negative values. Both technosols presented similar values and clonal patterns regarding foliar $\delta^{13}C$ and thus, no significant difference was found between technosols.

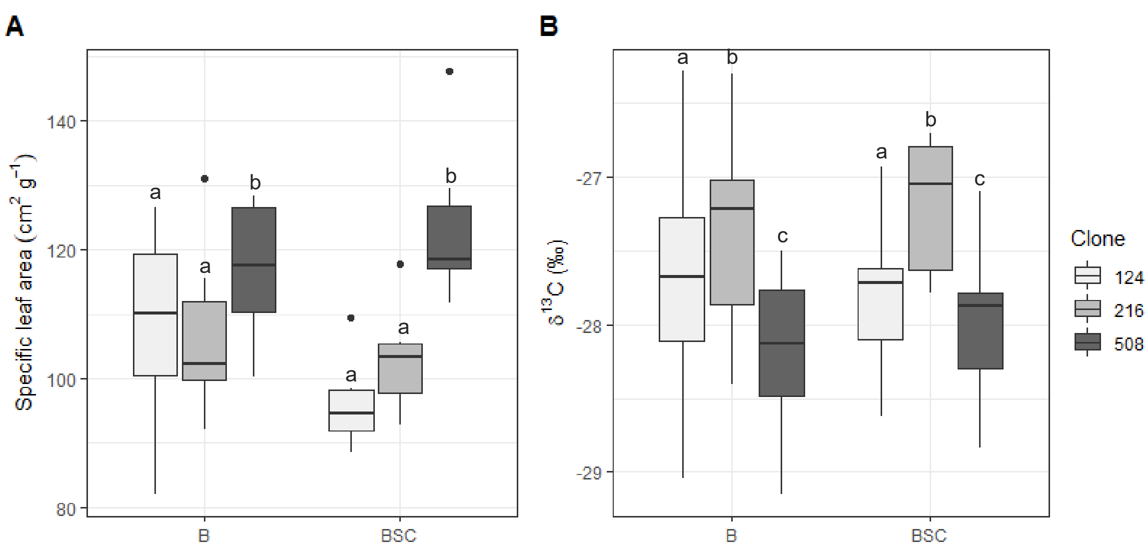

Figure 12. – Box and whisker plots of specific leaf area (A) and foliar $\delta^{13}C$ of the three hybrid poplar clones growing on the two technosols (B is the basic mixture of biosolids and deinking sludge; BSC is the basic mixture with the addition of Class B contaminated soils). Clone 124 is M×N-103124, clone 216 is M×B-102216 and clone 508 is DN×M-915508. The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots. Within each technosol, different letters indicate significant differences among hybrid poplar clones.

Intrinsic water use efficiency (WUE_i) expressed as net carbon assimilation (A_{net}) on stomatal conductance to water vapour (g_{sw}) was contrasted between dry and wetter conditions for all soil types and sites (Figure 13). Under dry conditions, A_{net} had a greater range of values (5 to 20 mol $m^{-2}\ s^{-1}$), whereas g_{sw} had a limited range of values (a little over 0.0 to 0.2 mol $m^{-2}\ s^{-1}$). Under wetter conditions, A_{net} varied from 16 to 26 mol m^{-2} s^{-1} and g_{sw} reached values from 0.1 to over 0.5 mol m^{-2} s^{-1}. A clonal effect on WUE_i was apparent only under wetter conditions (Figure 14). Clone 508 had a significantly higher WUE_i than the other clones (Table 6). Although no technosol effect was detected, clone 124 had a lower WUE_i under the B soil compared to the BSC soil. Clones 216 and 508 showed similar trends for both technosols.

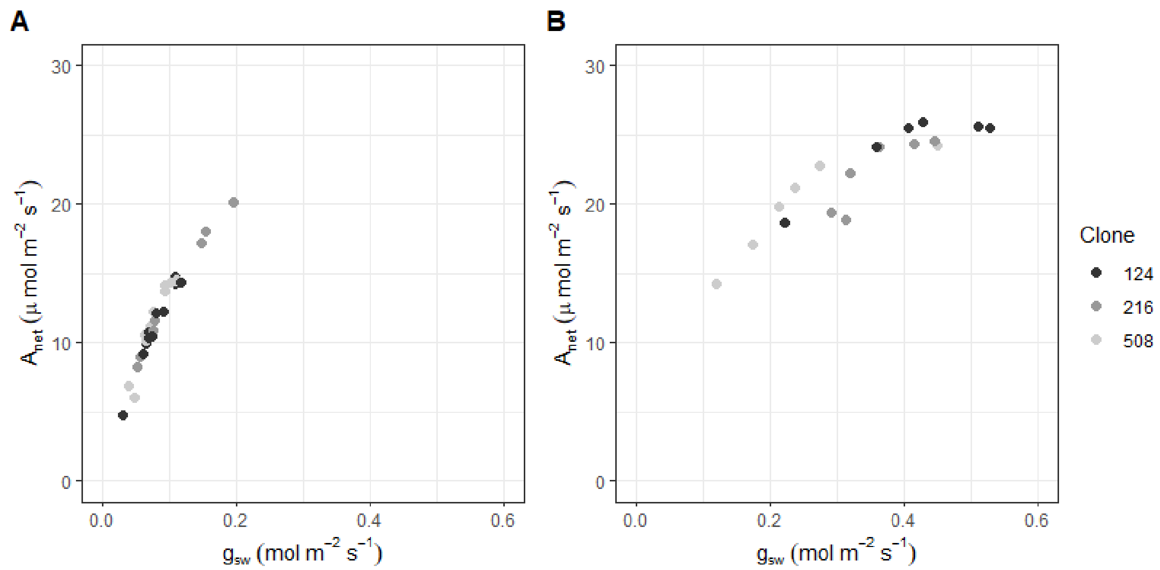

Figure 13. – Intrinsic water use efficiency (WUE_i) of the three hybrid poplar clones shown as the relation between net carbon assimilation (A_{net}) and stomatal conductance to water vapour (g_{sw}) under dry (A) and wetter (B) conditions. Clone 124 is M×N-103124, clone 216 is M×B-102216 and clone 508 is DN×M-915508.

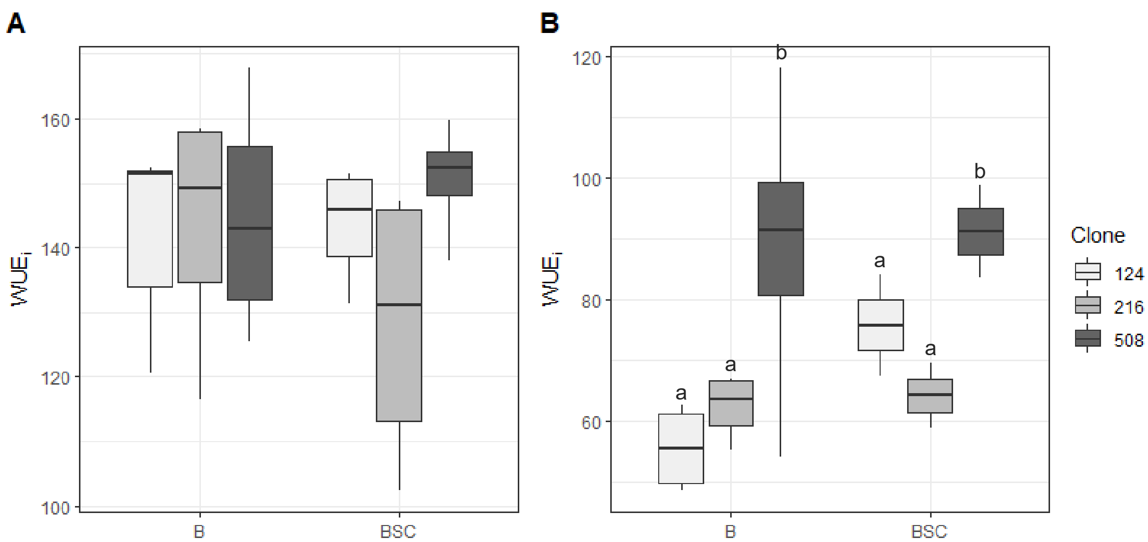

Figure 14. – Box and whiskers plots of intrinsic water use efficiency (A_{net} : g_{sw} ratio) of the three hybrid poplar clones in dry (A) and wetter (B) conditions. Clone 124 is M×N-103124, clone 216 is M×B-102216 and clone 508 is DN×M-915508. The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots. Within each technosol, different letters indicate significant differences among hybrid poplar clones

2.5. Discussion

Results in this study suggest that the two technosols constructed on mining waste of an old asbestos mine have shown mitigated successes in supporting survival and growth of tree seedlings over three growing seasons. Survival and growth of hybrid poplars were clearly superior to all conifer species tested.

2.5.1. Survival rates

There were significant contrasts in survival rates between conifer species (10 – 55 %) and hybrid poplars (90 – 98 %). Some variability was also observed in survival rates among conifer species, with spruces showing higher survival rates than pines and tamarack. In the context of mine reclamation, survival rates of conifer species largely vary depending on site conditions and substrates/technosols. Compared to this study, Onwuchekwa et al. (2014) reported similar survival rates for jack pine and white spruce, whereas other studies reported greater survival rates for tamarack, jack pine and white spruce (Larchevêque et al., 2015; Gagnon et al., 2020).

In general, *Populus* species and hybrids show potential for afforestation of old mines and quarries (Felix et al., 2008; Quinkenstein and Jochheim, 2016; Tremblay and al., 2019; Bai et al., 2022). Some studies have reported lower survival rates of conifer species in comparison to hybrid poplars on mine sites (Casselman et al., 2006), whereas others found that hybrid poplars had lower or similar survival rates compared to conifers (i.e. jack pine and tamarack) depending of the substrate used (Larchevêque et al., 2015; Guittonny-Larchevêque and Pednault, 2016). Survival rates > 90 % in this study for hybrid poplars are considered high compared to the literature, namely for reclaimed sites but also for abandoned farmland. For example, Truax and al. (2012), who studied the yield of hybrid poplar plantations across a gradient of fertility and elevation on abandoned farmland in Quebec, observed survival rates > 80 % after eight years for clones with the same parents (DN × M, N × M and M × B) as in this study, except at the lowest fertility site where survival rates were > 65 %.

Transplanting stress, mostly related to drought sensitivity, could explain the lower survival of the conifer seedlings. Environmental conditions in the month after planting were particularly warm and dry. These conditions could have created warm soils and low water availability, which can be critical for newly planted seedlings, as their root system is shocked and dysfunctional, preventing the plant to get sufficient water to compensate with transpiration (Rietveld, 1989; Grossnickle, 2005). Additionally, colonization of technosols by competing vegetation was very limited during the first growing season but increased substantially in the second and third years of the experiment. Competition for light was likely more severe for conifers due to their smaller initial dimensions and slower growth compared to hybrid poplars (Drake, 1986; Balandier et al., 2006; Tremblay et al., 2019).

The alkaline pH of both technosols (8.00) also may have negatively affected some of the conifers planted. This includes, for example, *Larix laricina* for which it is not recommended to plant in soils with a pH higher than 7 (Camiré and Brazeau, 1998; Zhang and al., 2013). Jack pine seedlings were also found to be highly sensitive to calcareous and alkaline soils, with reactions to high pH including inhibition of root cell elongation and significant decrease in net photosynthesis (Zhang, Xu, and Zwiazek, 2015). Although white spruce was found to be more tolerant on higher pH than tamarack and jack pine, Zhang et al. (2013) and Xu and al. (2020) showed nevertheless a negative impact of neutral to alkaline pH value on plant physiology such as foliar nutrition, water transport and transpiration and thus on long-term growth and survival. The higher survival of white spruce in this study could be explained by more resistance to higher soil pH as well as the larger seedling size at the time of planting, as well as stock types (Table 1). Indeed, larger containerized seedlings can show better growth and competitive potential compared to smaller (i.e. jack pine, red pine and tamarack) or bareroot (tamarack) seedlings (Burdett, 1990; Thiffault, 2004; Grossnickle, 2012; Thiffault et al., 2012). Other factors that could have contributed to the poor success of conifer survival could be associated to N availability and metal stress. These are discussed later.

No significant difference was found between hybrid poplar clones and between technosols when all species were considered. Yet, apart for tamarack and white spruce, survival rates were generally higher under the BSC technosol for all species, including hybrid poplar clones. There was no difference in bulk density or soil water content between the two technosols. The greater survival rates under the BSC technosol could be due to its greater thickness (see method section for details on reconstruction steps). It is possible that the contaminated (mineral) soil contributed to the maintenance of the BSC technosol structure, whereas the B technosol, composed only of biosolids, tended to collapse more easily during rain events and when water resided at the soil surface (Drake, 1986). As mine tailings and waste rocks do not promote root expansion, a deeper technosol allows more rooting space and thereby can reduce water stress (Grossnickle, 2005; Guittonny-Larcheveque et al., 2016).

2.5.2. Hybrid poplars

Growth, foliar nutrition and water use efficiency of hybrid poplars were evaluated because of their high potential for reclamation at the asbestos mine and as a means to identify the most suited clones for the prevailing conditions. Casselman et al. (2006) also found hybrid poplars to grow faster than conifers and other deciduous species. Hybrid poplars were on average 100 cm tall when planted and reached an overall average stem height of 236 cm after three growing seasons, which corresponds to a growing rate of 45 cm y^{-1}. This growth rate is comparable to growth rates reported in other mine reclamation studies (Casselman et al., 2006; Larchevêque et al., 2015), in forest environments characterized by acidic/poor soils (Bilodeau-Gauthier et al., 2011), and in plantations for biomass production growing at a similar latitude (Larocque, 1999).

Clone 124 (M × N) had the highest median stem height and diameter, with some individuals reaching a stem height of 350 cm and a diameter at collar of 60 mm. Similarly, Truax et al. (2012) reported a clone from the same parents (N × M) to be the most productive on farmland in southern Quebec. Yields reported here for clone 124 are, however, inferior to those measured for the same clone under more suited growing conditions. For example, clone 124 easily reached a stem height of 6 m and a diameter at collar of 50 mm after three growing seasons on fertile farmland soils in southern Quebec (Labrecque and Teodorescu, 2005). Clone 216 (M × B) had the lowest median stem height and diameter. It is one of the most rustic hybrid poplar clones cultivated in Québec, with both his parents in the *P. Tacamahaca* taxonomic section. Poplars of that taxonomic section are usually more resistant to colder conditions and can prosper on low fertility soils (Réseau ligniculture Québec, 2011), but can be outperformed on more fertile sites (Truax et al., 2012).

No significant difference in growth was found between hybrid poplar clones, but hybrid poplars reached significantly greater height and diameter under the BSC technosol. Again, more rooting space and thereby water availability could explain such findings (Guittonny-Larcheveque et al., 2016). Furthermore, wind lodging is a common phenomenon in crops and can lead to significant losses in yields (Kong et al., 2022). This effect was not measured *per se* at the mine, but high winds resulted in the inclination of hybrid poplar stems by as much as 30° and caused the lower leaf surface to be exposed to the sky during daytime (N. Bélanger, personal observation), which leads to less light assimilation. It is speculated that the deeper BSC technosol possibly allowed a deeper rooting depth, which not only exposed roots to other sources of water, but also provided a more robust anchoring that limited lodging in these high elevation and windy environments.

2.5.3. Soil nutrient availability and foliar nutrition

Results indicate that soil N availability was not optimal in the technosols studied, but that foliar N nutrition exceeded the minimal nutritional requirement for hybrid poplar clones (Camiré and Brazeau 1998). However, foliar N concentrations were below the optimal nutritional requirements (Hansen, 1994; Coleman et al., 2006). Inorganic N was almost exclusively available in the soil solution in the form of nitrate (NO_3^-). The late-successional conifers, like white spruce, have been found to proceed to root discrimination against NO_3^- as ammonium (NH_4^+) is preferred (Kronzucker et al., 1997). This could in part explain the higher mortality rates of conifers in our study. Conversely, hybrid poplars likely benefited from the availability of NO_3^- as it was shown to be a limiting factor (Fortier et al., 2010). It must be noted, however, that some *Populus* species and hybrids are, like conifers, better adapted to N uptake from NH_4^+ instead of NO_3^-, especially in alkaline soils like those of the studied technosols with a typical pH of 8.0 (DesRochers et al., 2007). Moreover, the total N supply in the B and BSC technosols was about 20% lower than the average N supply measured in forest soils using PRS probes (Oacha-Hueso et al., submitted), which could imply that the technosols are not optimal for N nutrition of hybrid poplars. Despite all soil N variables indicating sub-optimal conditions for N nutrition, foliar N levels exceeded the minimal requirement of Camiré and Brazeau (1998) for hybrid poplars. We suspect this is not a conservative threshold and that another minimal requirement should be considered for the clones studied (e.g., Hansen (1994), Coleman et al. (2006)).

Truax et al. (2012) found that P availability in hybrid poplar plantations on abandoned farmlands was most influential on the productivity of *P. maximowiczii* hybrids. This also appears to the case for all hybrid poplar clones tested at the asbestos mine. Phosphorus activity in the form of PO_4^{3-} in the two technosols was slightly above the average PO_4^{3-} activity measured in forest soils around the world using PRS probes (Oacha-Hueso et al. submitted). Yet, foliar P levels were consistently found to be under minimal requirements (Camiré and Brazeau, 1998), despite no obvious signal of low soil P availability. This was the case for clone 508, for example (Bilodeau-Gauthier et al. 2022). Ammonium activity improves P uptake of hybrid poplars and thus, the low availability of NH_4^+ could explain in part the low foliar P levels of the three clones studied (DesRochers et al., 2007). Precipitation of PO_4^{3-} with Ca is also a very active process in soils with pH of approximately 8.0 such as those of the two technosols (Havlin et al. 2005) and is perhaps a sink of PO_4^{3-} in the rhizosphere (Hinsinger, 2001; Hawkesford et al., 2012).

Foliar K concentrations, an essential macronutrient for growth and drought resistance (Ahanger et al., 2014; Ahmad et al., 2018), was well under the minimal nutritional requirements, even more so for the B technosol containing only biosolids. The studied technosols had less than half of the K supply compared to forest soils worldwide (Oacha-Hueso et al. submitted). The mixing of contaminated soils with biosolids (i.e. technosol BSC) appears to have mitigated some of that problem (as seen by higher K activity measured by PRS), thus rendering more K in foliage. This is somewhat similar to findings in forest environments where the application of paper mill by-products (ex. paper biosolids and lime mud) to soils increased foliar K levels of clone 508 but never reached the minimal requirements (Bilodeau-Gauthier et al. 2022). The low foliar K levels could also be explained by uptake interferences due to N (Kelty et al., 2004) and Ca (Arnold and Van Diest, 1993). In this study, deinking sludge, which is particularly high in Ca, was used in high amounts in the mixture to create the technosols and thus, we suspect that Ca is likely the main interference for K. In the case of foliar Ca and Mg levels, they respectively exceeded or were similar to the minimal and/or optimal requirements despite showing that technosol Ca and Mg supplies were low compared to forest soils worldwide (Oacha-Hueso et al. submitted). High Ca concentrations in leaves may result in an antagonism between K and Ca, which causes K compensation by Ca in vacuoles (Diem and Godbold, 1993). Fertilization has resulted in decreases in foliar Mg due to Ca antagonism on soil exchange site and during root uptake by trees (Loide, 2004; Bilodeau-Gauthier et al., 2022). The deinking sludge was used in the technosol mixture as a means to mitigate the imbalances that could be created by Mg due to roots exploring parts of the asbestos-rich tailings ($Mg_3Si_2O_5(OH)_4$). After three years of growth, either: (1) roots have not reached the tailings and are not actively absorbing Mg released for asbestos minerals, or (2) roots have reached the tailings and are actively absorbing Mg but Ca-rich technosols are concomitantly and efficiently releasing Ca and thus balancing Ca and Mg uptake. Moreover, foliar Ca levels varied substantially depending on hybrid poplar clones, with clone 216 exhibiting the highest levels but the lowest yields. These findings point to the conclusion that Ca and Mg are not the nutrients limiting hybrid poplars in this system.

Foliar concentrations of metals like Al, Zn, Cu and Pb (not presented) were under the detection limit of the analytical method used. However, total Cu, Pb and Zn concentrations of the technosols were comparable to concentrations in acidic forest soils where biosolids had been applied 25 years earlier in a stand of Douglas-fir and where growth was reduced and foliage levels of phytochelatins (a bioindicator of intracellular metal stress) were increased compared to a control forest site (Cline et al. (2012). In addition, when compared to forest soils worldwide (Oacha-Hueso et al. submitted), both technosols showed greater ionic activity of Cu and Pb by at least two-fold. While pH is a key factor influencing bioavailability and mobility of metals and contaminants (Krol et al., 2020) and that a pH of 8.0 appears to be restricting a phytotoxic response in the mine site, future acidification of the technosols (e.g. by natural and anthropogenic factors such as plant growth and precipitations) could affect metal availability in the mid to long term and as such, monitoring of soils and trees will continue at the site.

2.5.4. Water stress and water use efficiency

A rapid colonization of the technosols by competing vegetation started in the second year of the experiment, making water resources likely more limiting for tree seedlings. The *Populus* genus is particularly susceptible to drought and requires significant amounts of water to maximize its growth rate (Pinno et al., 2009; Théroux Rancourt et al., 2015). Differences in drought tolerance between different *Populus* species and clones have been reported. Poplars of the *Tacamahaca* taxonomic section tend to show drought-tolerant characteristics and are suggested as being more suitable in environments frequently subjected to water limitations than other taxonomic sections (Silim et al., 2009). For example, *P. balsamifera* was found to have high water-use efficiency conferred by its water-conservative strategy (or isohydric behaviour) at the cost of biomass production when compared to some hybrid poplar clones (Larcheveque et al., 2011; Attia et al., 2015).

We thus compared intrinsic water-use efficiency (WUE_i as a function of $A_{net} : g_{sw}$) and $\delta^{13}C$ (as a proxy of integrated water-use efficiency) under two general soil water content scenarios, i.e. dry and wetter conditions. The leaf samples used to assess carbon isotope discrimination and specific leaf area were taken during a dry period in late summer 2020 based on soil water potential data (65 kPa). All clones had higher WUE_i under water deficit conditions, which can be explained by a substantial decrease in stomatal conductance and a moderate decrease of net carbon assimilation. No significant differences in WUE_i under dry conditions were detected between clones, but SLA and $\delta^{13}C$ results suggest a greater drought tolerance for clone 508. This clone had higher SLA, which indicates a greater drought tolerance (Marron et al., 2003). It also had the lowest $\delta^{13}C$, reinforcing the idea that clone 508 is more drought tolerant. Théroux Rancourt et al. (2015) also described clone 508 as the most drought tolerant clone in their greenhouse study. Conversely, they classified a clone from the same parents as clone 216 (M × B) as being drought sensitive. Similarly, clone 216 trees at the asbestos mine exhibited lower SLA values and the highest $\delta^{13}C$ values, indicating a less pronounced carbon isotope discrimination. Since clone 124 presented SLA and $\delta^{13}C$ values between those of clones 508 and 216, its drought tolerance is suggested as intermediate relative to the two other clones.

Drought-tolerant clones can maintain a relatively high A_{net} and a moderate g_{sw} rates when soil moisture is limited, compared to more drought sensitive clones (Silim et al., 2009). This was not observed in this study, however. Under wetter conditions, clone 508 had significantly higher WUE_i, probably because of a prolonged conservative state even after a rainfall event. Indeed, drought response over time might be different between clones as well as recovery time needed to attain homeostasis. In addition, this response can vary over the course of days or even weeks after being exposed to water-limited conditions (Théroux Rancourt et al., 2015).

2.5.5. Implications, sustainability and conclusions

Study results suggest that planting trees on windrowed technosols with a thickness of 1 m and composed of a mixture of municipal biosolids, deinking sludge and contaminated soils is a reliable reclamation technology for tailings and rock piles following asbestos mining, with carefully selected planting species. Hybrid poplars have a particularly high potential for afforestation on windrow technosols compared to all other species tested. Although there were only few differences between the technosols tested, results suggest that the addition of Class B contaminated soils to a mixture of municipal biosolids and deinking sludge can be advantageous regarding tree survival and growth, likely because of its greater thickness/volume. As biosolids and other fertilizing materials can differ in chemical composition from one source of production to another, the use of such mixtures on a large scale should be done with care (Larney and Angers, 2012).

Some conifers also indicated potential for afforestation of asbestos mines (e.g. tamarack), but adjustments in cultural practices are needed to optimize seedling survival and growth. For example, most species would benefit from a more thorough weed control. Chemical and mechanical weed control are both possible in the mining context, but future research is needed to test different low-growing ground cover that may act as a living mulch to decrease weeds, prevent soil erosion, enhance soil porosity and add nutrients, notably N (Franklin et al., 2012). Some conifers could also benefit from mycorrhizae inoculation of technosols prior to planting. Inoculation of forest soils with mycorrhizae was shown to increase the growth of jack pine and the survival and root development of white spruce (Onwuchekwa et al., 2014; Nadeau et al., 2018).

However, responses of trees to mycorrhizae inoculation of technosols in the context of mine reclamation may vary depending on tree and fungal species involved (Quoreshi et al., 2007).

Plantation success at the mine site, i.e., a high survival rate and satisfactory juvenile growth of hybrid poplar clones, could be compromised in the medium or long term. As trees enter the second stage of development, water and nutrient demand will increase and thus, they may become growth limiting. Canopy closure of the hybrid poplars will likely occur within a few years (5 or 6), thus suppressing most of the competing vegetation. However, in similar settings, roots of planted trees develop in the technosols but not in the substrates below (Guittonny-Larcheveque et al., 2016), suggesting that intraspecific competition for rooting space will increase in time and could slow yields substantially. Monitoring of tree growth, along with other variables such as plant and faunal diversity, should be done to assess long-term success of the mine reclamation.

Encroachment of native trees on the technosols was also relatively rapid and active (see Supplementary material B) and consequently, monitoring of this phenomenon should be monitored as another measure of reclamation success. In light of these observations, natural tree regeneration on technosols could be another possible avenue for reclamation of asbestos mines. There are likely socio-economical and environmental benefits of favouring natural regeneration of the nearby forest trees (Macdonald et al., 2015). Planting trees only in areas where natural regeneration has failed to establish on technosols could substantially reduce the costs of mine reclamation. Further research is needed to confirm the technical feasibility and benefits of this alternative approach to asbestos mine reclamation.

Conclusion

L’étude présentée au chapitre précédent a montré que la technologie de remise en état utilisée a permis, à un succès relativement élevé pour certaines espèces, d’établir une plantation d’arbres sur des résidus et des stériles d’une ancienne mine d’amiante dans la région de Chaudière-Appalaches. Les résultats recueillis lors de l’étude ont démontré que, pour les trois premières années de la plantation, les technosols issus de mélanges de matières résiduelles fertilisantes ont favorisé la survie et la croissance de quelques espèces d’arbres. Les peupliers hybrides plantés sur les technosols comprenant l’ajout de sols contaminés au mélange de matières résiduelles fertilisantes de base, composé de biosolides municipaux et de boues de désencrage, avaient les hauteurs et les diamètres les plus élevés. Puisque les propriétés physicochimiques et la biodisponibilité des nutriments des deux technosols étaient similaires, nous pensons que l’avantage de croissance procuré par le technosol contenant les sols contaminés est dû son épaisseur supérieure qui, probablement, confère aux arbres plus d’espace pour le déploiement de leurs racines et un plus grand réservoir d’eau et de nutriments.

Le taux de survie des peupliers hybrides (90 – 98%) était nettement supérieur à celui des conifères (10 – 55%) après trois saisons de croissance. La croissance rapide des peupliers hybrides est un trait qui semble favoriser le succès d’établissement de ces espèces d’arbres dans le contexte du boisement des haldes minières de la région. Au moment de la prise de données, il n’y avait pas de différences significatives dans la survie et la croissance des trois clones étudiés. Par contre, au fil du développement de la canopée des peupliers hybrides, nous prédisons une baisse de la végétation herbacée et de la compétition entre celle-ci et les arbres et une hausse de la compétition entre les arbres pour les ressources souterraines. Cette compétition accrue pourrait exacerber les stress hydriques, mais aussi les stratégies de conservation des ressources et de croissance des arbres. Ainsi, puisque les andains offrent un volume limité de sol à exploiter et que les racines des arbres ne devraient pas explorer les résidus et stériles (Guittonny-Larcheveque et al., 2016), il sera sans doute possible de distinguer des différences dans les tendances de survie et de croissance des clones de peupliers à mesure que les plantations gagneront en maturité.

Les retombées de cette étude sont multiples. D'abord, la restauration des forêts est un enjeu d'actualité, en raison de l'importance de ces écosystèmes pour les services rendus ainsi que pour leur rôle dans l'atténuation des changements climatiques, autant à l'échelle locale que globale. En effet, la promotion des stocks ligneux offre un très grand potentiel de séquestration de carbone (Griscom et al., 2017), mais les conditions hostiles qui prévalent dans les anciens sites miniers nécessitent qu'on s'attarde spécifiquement à la construction de sols qui possèdent les propriétés requises pour répondre aux besoins des arbres et maximiser les puits de carbone. La présente étude a donc permis de tester et de valider avec succès une nouvelle technologie pour le boisement de ces environnements, et peut maintenant servir d'exemple pour les propriétaires et les gouvernements en charge de restaurer les mines d'amiante du Québec. L'usage du peuplier hybride sur les andains contenant le mélange de sols contaminés, de biosolides municipaux et de boues de désencrage est l'approche recommandée dans le contexte des mines d'amiante. Il est toutefois nécessaire de mentionner que la composition chimique des matières résiduelles fertilisantes peut varier d'une source de production à l'autre (Larney et Angers, 2012). La reproduction du dispositif expérimental à plus grande échelle devrait être fait avec précaution, en prenant soin de choisir des intrants de bonne qualité. De plus, puisqu'un des intérêts principaux à utiliser des matières résiduelles fertilisantes est que celles-ci ont un coût relatif lié uniquement à leur transport et à leur épandage, le dispositif expérimental pourrait être adapté en fonction de la disponibilité des matières fertilisantes agricoles, industrielles ou forestières à proximité du site minier (Larney et Angers, 2012).

Les résultats de l'étude ainsi que les recommandations qui en découlent semblent également conformes à la position du public en matière de remise en état des mines d'amiante. Effectivement, une étude récente de Lévesque et al. (2020) témoigne du désir de la population locale de voir l'élargissement des milieux naturels de la région (85% en accord), de la promotion des puits de carbone (75%), de la valorisation des matières résiduelles fertilisantes (79%) et de la diminution de l'érosion et de la contamination du système hydrographique (84%).

Plusieurs nouvelles pistes de recherches ont été soulevées pendant cette étude. Dans un premier temps, il apparait essentiel d'évaluer à moyen et à long terme la survie, la croissance, la nutrition foliaire et le comportement des arbres face au stress hydrique. Une attention particulière devrait être mise sur le peuplier hybride compte tenu de son fort potentiel pour le boisement de ces milieux. Depuis la publication des résultats préliminaires de cette étude, un partenariat a été mis en place dans le but de boiser la plupart des haldes de la mine où l'étude a été menée. En effet, dans le cadre d'un projet financé par le programme 2 milliards d'arbres de Ressources naturelles Canada, des centaines de milliers d'arbres seront plantés au cours des dix prochaines années sur les plateaux des haldes en utilisant principalement la technologie proposée ici ou des approches culturales légèrement adaptées. En vue de garantir le succès de cette initiative à grand déploiement, il sera nécessaire de suivre l'évolution des arbres de l'étude, incluant la nutrition foliaire, afin de modifier, le cas échéant, les taux d'application des matières fertilisantes ou encore les stratégies de contrôle de la végétation compétitrice.

Dans l'optique où le site minier serait trop vaste et les forêts avoisinantes trop éloignées pour permettre à des propagules d'espèces indigènes d'être dispersées naturellement sur le site à restaurer, des espèces indigènes du genre *Populus* pourraient être plantées au lieu de peupliers hybrides. Un avantage pouvant être soulevé est que, contrairement aux peupliers hybrides, les peupliers indigènes peuvent se reproduire de manière sexuée et ainsi assurer une certaine viabilité à moyen terme de la plantation. De plus, des études ont montré que certaines espèces de peupliers indigènes, notamment *Populus balsamifera* et *Populus tremuloides*, peuvent tolérer des stress associés à l'état hydrique et nutritionnel des sols, parfois de manière comparable à des peupliers hybrides (Larchevêque et al., 2011; Pinno et al., 2011). Finalement, l'utilisation d'arbres indigènes est généralement mieux acceptée que l'usage d'hybrides et d'arbres génétiquement modifiés (Hall, 2007; Martín et al., 2018; Brennan et al., 2021). L'essai de d'autres matières résiduelles fertilisantes ainsi que de peupliers indigènes pourraient permettre de développer un dispositif expérimental adapté à un plus grand éventail de conditions.

Enfin, à la lumière de nos observations sur la colonisation rapide et active des technosols par les peupliers des forêts limitrophes (voir le matériel supplémentaire B), la régénération naturelle des technosols doit être considérée comme une avenue possible de remise en état des écosystèmes forestiers sur les sites d'anciennes mines d'amiante du Québec. En effet, il pourrait y avoir des avantages socioéconomiques et environnementaux importants découlant de la régénération naturelle sur des technosols par les forêts avoisinantes (Macdonald et al., 2015). Des recherches supplémentaires seraient nécessaires pour confirmer les avantages de cette approche. Par exemple, faudrait-il uniquement planter des arbres là où les arbres de la forêt avoisinante n'ont pu efficacement s'établir ? Aussi, la question de la stabilisation et du devenir des talus des haldes demeure toute présente. L'érosion active des talus est un enjeu environnemental important dans la région (Groupe de concertation des bassins versants de la zone Bécancour, 2021). Pour le bien des écosystèmes et des communautés limitrophes, d'importants travaux d'ingénierie et de génie végétal doivent être réalisés. Ces travaux seront coûteux et devront être accompagnés d'une volonté politique importante pour se réaliser.

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Annexes.

Supplementary material A: Survival and growth of shrubs and trees planted on technosols after establishment of grasses

Context

Ecological restoration has been ongoing since 2012 at the asbestos mine. Revegetation of tailings and waste rock piles is generally done by seeding grasses on soils reconstructed with the same mixture of by-products, i.e. municipal biosolids and deinking sludge. Although this practice was successful, no attempt for afforestation was made to restore a tree cover on these already revegetated sites. The objectives of this experiment were to evaluate the best suitable (1) planting microsites and (2) tree species for the afforestation of revegetated sites.

Method

The experimental design consisted of four restoration sites established on land previously reclaimed between 2012 and 2015 by seeding grasses on soils reconstructed with a mixture of municipal and deinking sludge applied at a rate of 1200 Mg ha^{-1} (300 and 900 Mg ha^{-1}, respectively). This provided an equivalent technosol depth of approximately 20 cm. Technosols were seeded with oats (*Avena sativa*) at a rate of 80 kg ha^{-1} and was used as a nurse plant, whereas a mixture composed by 55 % of Timothy grass (*Phleum pretense*), 30 % of red clover (*Trifolium pretense*), and 15 % of alsike clover (*Trifolium hybridum*) was applied at a rate of 20 kg ha^{-1}. Selected tree species included green alder (*Alnus viridis* subsp. *crispa*), tamarack (*Larix laricina*), white spruce (*Picea glauca*), three cultivars of miyabeana willow (*Salix miyabeana*; SX61, SX64 and SX67) and three hybrid poplar clones (*Populus maximowiczii* × *P. balsamifera*; M×B-915302, M×B-915303 and M×B-915311). These shrubs and trees were planted on three microsites in early June. The first microsite was prepared by excavating 50 × 50 cm mounds with a height of about 30 cm. A second microsite was prepared by applying glyphosate (regulation number 28486, 4.67 L ha^{-1} diluted in 100 L ha^{-1}) to a 30 cm radius around the centre of the planting microsite less than 24 hours before planting. A third microsite consisted of planting cuttings or seedlings without site preparation. In this case, shrubs or trees were planted directly into the grasses and this was considered the control.

Two survival surveys (October 2017 and October 2020) were conducted for all planted species. In addition, stem diameter at the collar as well as total height were measured for all living hybrid poplar trees in October 2020, i.e. four growing seasons after planting. Total height was measured using a telescopic measuring pole following Pérez-Harguindeguy et al. (2013)'s method for plant height measurements, and stem diameter was measured using an electronic caliper.

Two-way ANOVA (analysis of variance) was conducted to check for significant differences in survival and growth variables between the three planting microsites and among tree species, as well as their interaction (microsites × species). Data were transformed when conditions of normality of residuals, independence and homoscedasticity were not met. Tukey's HSD test (*glht* function in the *multcomp* R package) was used to assess differences between species or clones. All statistical analyses were conducted with R, version 4.0.4 (R Core Team, 2021).

Results and discussion

Survival rates of all shrub and tree species, including hybrid poplar clones as a group and individually, are presented in Table 1 and Figure 1, as well as in Table 2 for statistics.

Results highlight generally low survival rates, but very low survival for shrubs. Green alder and *miyabeana* willow exhibited very low survival rates under the control and herbicide treatment after one growing season (0% to 24%) and on all microsites after four growing seasons (0% to 9%). These survival rates were statistically lower than all tree species tested. During the 2020 survey, although the original willow shoots were dead, we observed shoots (regrowth) on some of the willow stools (Figure 3a). We also observed significant browsing on those shoots, and evidence (feces) of deer and hare activity in the study plots. *Miyabeana* willow was reported as being targeted by herbivores (Larsson, 1998; Tanaka and Nakamura, 2015). Moreover, grasses likely provided a suitable environment for these herbivores and as such, their increased presence is bound to be associated with greater browsing on the planted shrubs and trees (Parsons et al., 2007). Green alder is also quite sensitive to belowground competition, low soil moisture, and herbivory (Matthews, 1992; Asmara et al., 2022), and thus, we suggest that these factors, acting individually or in combination, led to quasi-total mortality of the species in the study plots. All tree species, i.e. tamarack, white spruce, and hybrid poplars, had statistically similar survival rates that differed from green alder and *miyabeana* willow, except for hybrid poplars and willows. There were no significant differences in survival between the three hybrid poplar clones.

Survival rates were significantly greater on mounds for all shrub and tree species after one growing season (69% to 99%) and for tamarack, white spruce and hybrid poplar clones after four growing seasons (50% to 53%). In a forestry context, mounding was shown effective in enhancing seedling survival and growth by suppressing competing plants and permitting good root development, namely for hybrid poplars (Bilodeau-Gauthier et al., 2011; Mc Carthy et al., 2017; Thiffault et al., 2020). However, mounds have also been found to promote low soil moisture (Löf and Birkedal, 2009). Although mounds in the study plots have likely reduced competition for light and increased soil volume for root development (and thus nutrient uptake), water stress likely remained an issue and explains, at least in part, the overall decrease in survival between the first and fourth growing season (Figure 3b). Since no significant effect was detected between the control and herbicide treatment, we suggest that the application of one dose of glyphosate was ineffective in limiting competition stress from grasses. Perhaps a more sustained control of competing vegetation, chemical or mechanical, during the first few years after planting would have been more successful (Desrochers and Sigouin, 2015).

Tableau 7. – Survival rates of all shrubs and tree species after one and four growing seasons on the three planting microsites. Values are calculated based on the ratio of the number of trees alive on the number of trees monitored. In the case of hybrid poplar clones, results are shown for all clones combined (i.e. P. maximowiczii × P. balsamifera) and individual clones (i.e. M×B-915302, -915303, -915311).

Species	Planting microsites					
	One growing season			Four growing seasons		
	Control	Herbicide	Mounds	Control	Herbicide	Mounds
Alnus viridis subsp. *crispa*	0%	10%	77%	0%	0%	4%
Salix miyabeana	24%	6%	69%	1%	5%	9%
Larix laricina	40%	64%	95%	12%	29%	51%
Picea glauca	25%	77%	96%	13%	12%	52%
P. maximowiczii × P. balsamifera	87%	60%	99%	22%	0%	50%
M×B-915302	-	-	-	10%	0%	54%
M×B-915303	-	-	-	13%	0%	33%
M×B-915311	-	-	-	13%	0%	63%

Tableau 8. – Two-way ANOVA test results for survival and growth as a function of species or clone and planting microsite. Values with a significant effect ($p < 0.05$) are indicated by the * symbol.

Response variable	ANOVA				
	Transf.[1]	Shapiro[2]	Species or clone	Microsite	Interaction
Survival rate (all)[3]	exp. 2	< 0.001 *	< 0.001 *	< 0.001 *	0.012 *
Survival rate (hybrid poplars)[3]	-	0.005 *	0.380	< 0.001 *	0.353
Stem height	√	0.047 *	0.845	0.112	0.675
Stem diameter at collar	-	0.060	0.418	0.029 *	0.320

[1] Transformation used on data to help meet the criteria of a normal distribution
[2] Results (p-value) from the Shapiro-Wilk test are to show which variables followed a normal distribution after transformations. Variables with a significant effect ($p < 0.05$) are not normally distributed.
[3] Survival rates after four growing seasons

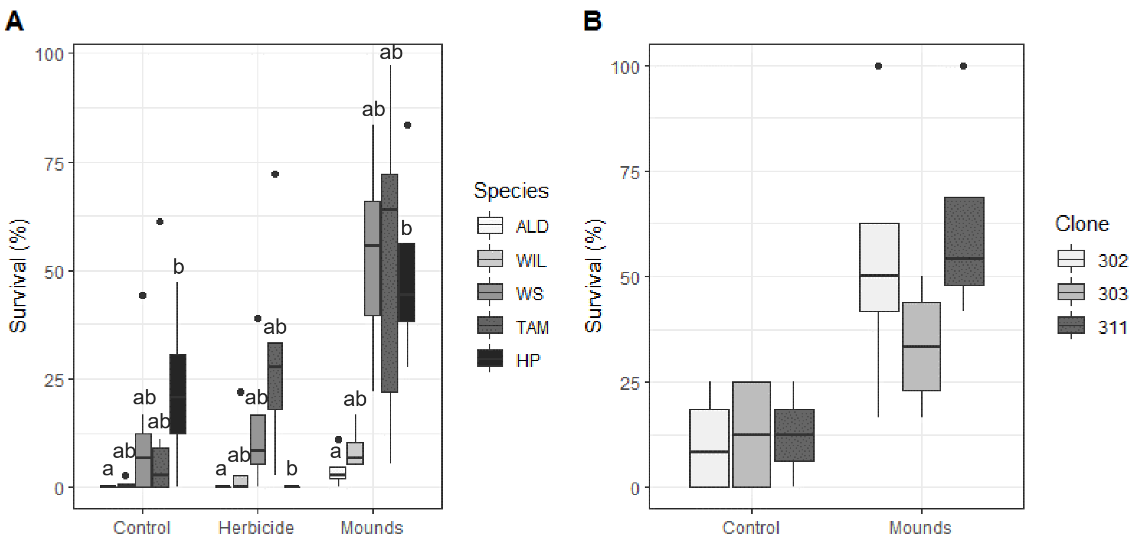

Figure 15. – Box and whisker plots of survival rates of all shrubs and tree species, including hybrid poplar clones as a single group (A) and the three hybrid poplar clones individually (B) on the three microsites. ALD is green alder, WIL is miyabeana willow, WS is white spruce, TAM is tamarack and HP is hybrid poplar (M×B-915302, -915303, -915311)). No data are shown for hybrid poplars under the herbicide treatment in B because no tree survived. The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots. Within each treatment, different letters indicate significant differences among tree species.

Stem height and diameter at the collar of hybrid poplar clones are presented in Figure 2. There was no effect of microsite or clone on stem height, but all clones showed larger diameters on mounds than in the control. The effect of weeding on hybrid poplar diameter was demonstrated before in other cultural contexts (Pinno and Bélanger, 2009).

Hybrid poplar clones were on average 100 cm when planted, and the ones that survived reached an overall mean height of 312 cm after four years of growth. This corresponds to a growing rate of 53 cm yr^{-1}, which is greater than the average 45 cm yr^{-1} of hybrid poplar growing on windrows (see main paper). Some individuals even reached a stem height of more than 500 cm, which corresponds to a growing rate of 100 cm yr^{-1} (see Figure 3c). This growth rate is comparable or superior to growth rates reported in other mine reclamation studies (Casselman et al., 2006;

Larchevêque et al., 2015), in forest environments characterized by acidic/poor soils (Bilodeau-Gauthier et al., 2011), and in plantations for biomass production growing at a similar latitude (Larocque, 1999), but slightly lower than hybrid poplars grown in a forest setting with support of liming and fertilization (Bilodeau-Gauthier et al. 2022) or in rich agricultural soils (Labrecque and Teodorescu, 2005). The clones tested in this experiment all had the same parents (i.e. *Populus maximowiczii* × *P. balsamifera*) and in turn, they did not exhibit a significant difference in survival and growth. Finally, although we did not monitor tamarack growth, the ones that survived seemingly had the second most substantial growth over the four years (Figure 3d).

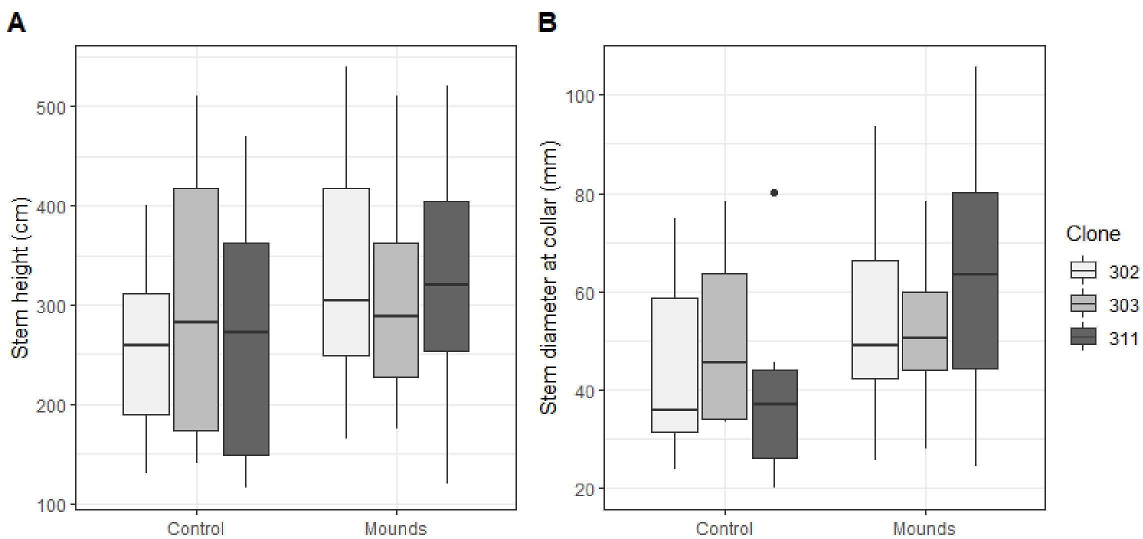

Figure 16. – Box and whisker plots of stem height (A) and stem diameter at the collar (B) of the three hybrid poplar clones (M×B-915302, -915303, -915311) growing on the two planting microsites (no hybrid poplar survived on the phytocide microsite). The coloured boxes represent the values between the 25th and the 75th percentiles, the horizontal line indicates the median and the vertical lines represent minimum and maximum values. Outliers (outside the percentile range) are indicated by black dots.

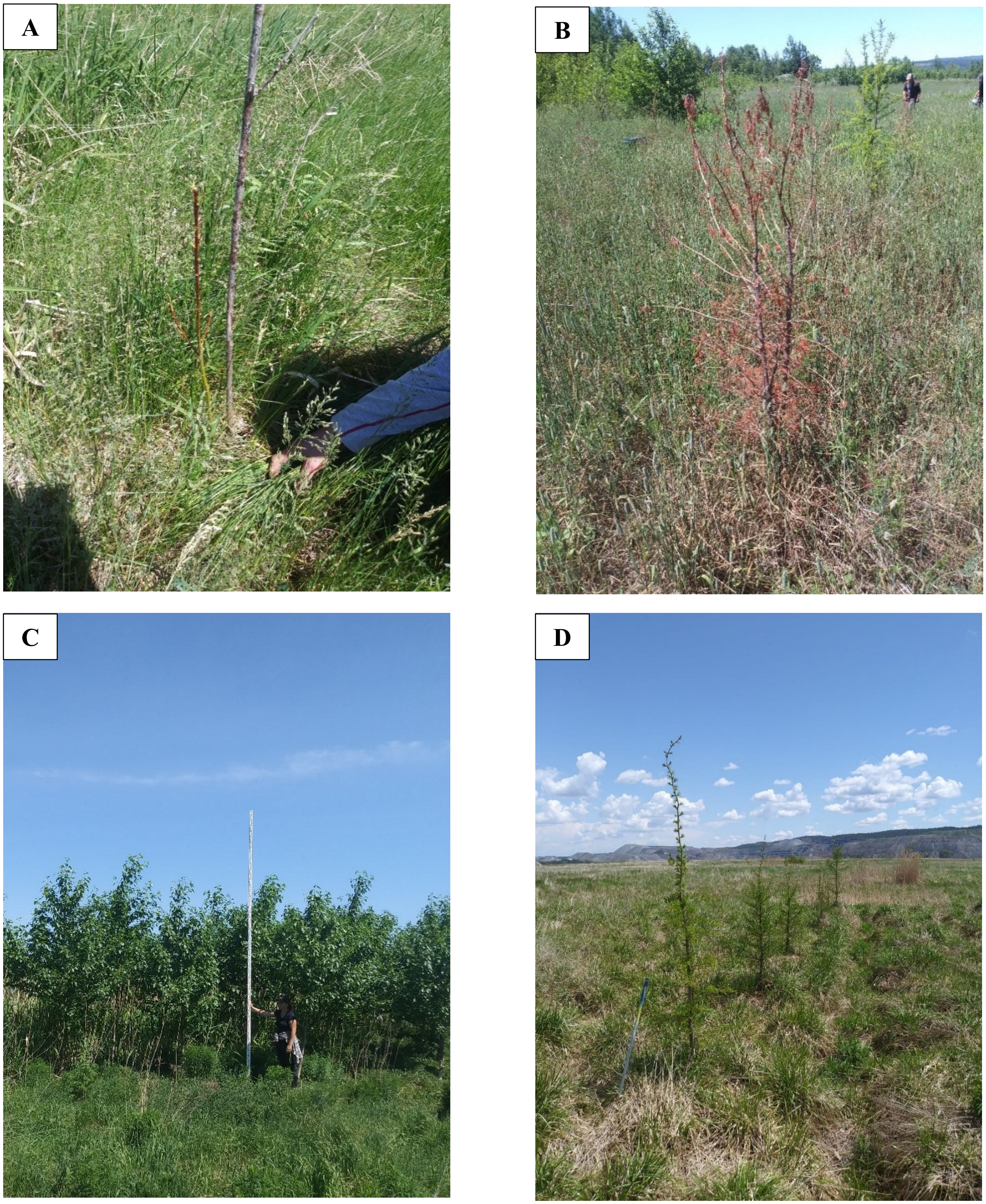

Figure 17. – Example of regrowth of a miyabeana willow stool (A), a tamarack seedling growing on a mound that succumbed to drought (B), a hybrid poplar plot with substantial survival and growth (C), and a tamarack plot with substantial growth (D).

Conclusion

Results from this experiment, with those on trees planted on windrows (main paper), have provided some valuable insights about the best practices to promote the establishment and growth of tree plantations on a former asbestos mine. Technosols colonized by grasses present abiotic (low soil moisture, limited rooting space) and biotic (competition, herbivory) stressors that do not encourage tree survival (below 50% after four growing seasons). Green alder and *miyabeana* willow were the least suitable species for afforestation, whereas tamarack, white spruce and hybrid poplars showed more potential. Mounding was the most successful planting microsite, likely because it mitigated competition for light and provided tree roots with extra soil volume to explore nutrients and water. Yet, foliar nutrition and water use were not studied in this experiment and likely remain a constraint considering the relatively high mortality, even in the case of mounding. For trees that survived, however, they yielded growth rates similar or above those planted on windrows. Survival and growth of these shrubs and trees could perhaps be improved if a more thorough control of competing vegetation is done in the first few years of establishment.

Supplementary material B: Natural encroachment of nearby forest trees on technosols

Context

Substantial encroachment (see Figure 1a) of the technosols was observed in the plantations two years after establishment. This phenomenon was not observed where no soil reconstruction occurred (Figure 1b). Tree species actively colonizing the sites were trembling aspen (*Populus tremuloides*) and balsam poplar (*Populus balsamifera*). In 2021, we thus partly assessed the magnitude of this phenomenon.

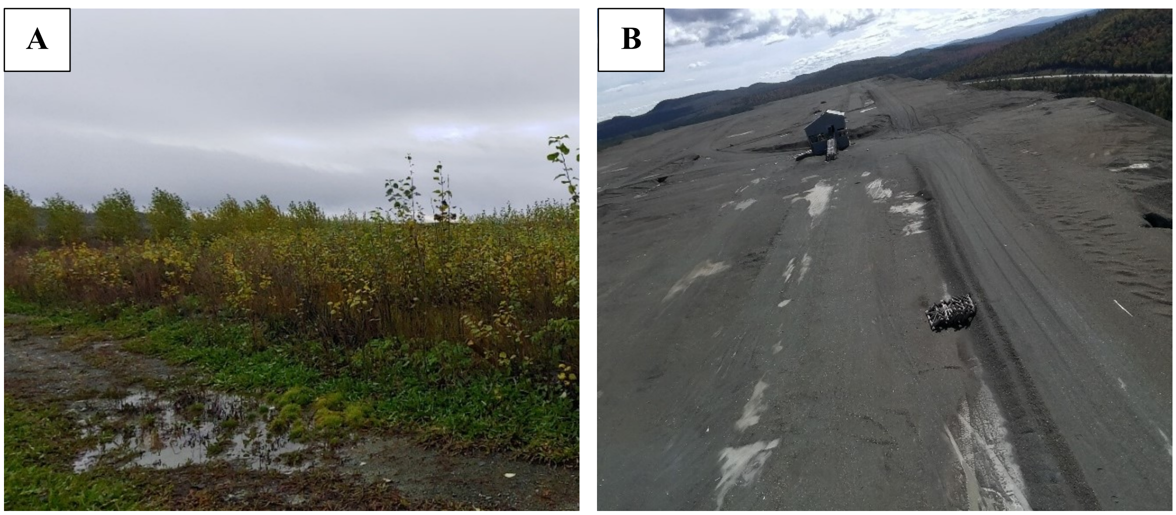

Figure 18. – Sections of the tailings pile with (a) and without (b) soil reconstruction. These sections respectively exhibited active and no colonization by native *Populus*.

Method

In June 2021, automated flights were conducted over each plantation using a M210 drone equipped with a X4S RGB camera (DJI). Flights were performed with Pix4D capture (Pix4D Inc.) on a sunny day at an elevation of 35 m for optimal image resolution. Image overlap was set at 80%. Orthomosaics were built using the orthomosaic module in Pix4Dmapper. These orthomosaics later served as the background maps for the representation of native *Populus* colonization.

Within both plantation sites, the location of all native *Populus* seedlings was recorded using a GPS system (precision < 50 cm) composed of a Trimble R1 GNSS receiver mounted on a 2 m carbon fiber pole and supported by the Terrasync Pro software operated on a Juniper Systems Mesa 2 GEO tablet. A single coordinate (point) was registered for trees growing in low density zones (< one tree m^{-2}), whereas the polygon function of the Terrasync Pro software was used to delimit the higher density zones (> one tree m^{-2}).

The recorded data were then used to layout and calculate the total surface area of the plantations colonized by trees using ArcGIS 10.6.1 (ESRI software). Calculations are thus estimates of the magnitude of colonization in terms of surface area covered by *Populus* canopies rather than number of stems.

Results and discussion

Results show that the development of technosols was highly favorable to natural regeneration (Figure 2). They illustrate that encroachment by *Populus* species native to the local forest was an active process at both plantation sites, whereas plateaus of tailings and waste rock piles where no soil reconstruction was done or where soils were reconstructed but then seeded with grasses are almost entirely bare of native plants. In fact, the two experimental plantations are the only areas at the asbestos mine site where native trees are capable of colonizing. Tailings and waste rocks are unsuitable substrates for plant growth considering they are often compacted, lack organic matter and have coarse textures, low water-holding capacity and limited microbial activity (Cooke and Johnson, 2002). We also observed that trees growing on technosols seeded with grasses had much lower survival rates (Supplementary material A). Grasses usually are more efficient for resources acquisition because they develop a greater root system than young trees (Casper and Jackson, 1997; Carnevale and Montagnini, 2002; Balandier et al., 2006; Kabba et al., 2007).

The area colonized with native *Populus* corresponds to 22% of the total surface of the technosols, both plantations combined. The plantation corresponding to the waste rock plateau was colonized at a greater extent (28 %) compared to that of the tailings plateau (14%). At both sites, the largest zones colonized by native *Populus* were concentrated in the center of the plantations with no preference of technosol type. Although this was not measured *per se*, these areas were in small depressions, which could have been conducive to water collection and thus native *Populus* establishment. The *Populus* genus is particularly susceptible to drought and requires significant amounts of water for maximum survival and growth (Marron et al., 2003; Pinno et al., 2009; Théroux Rancourt et al., 2015).

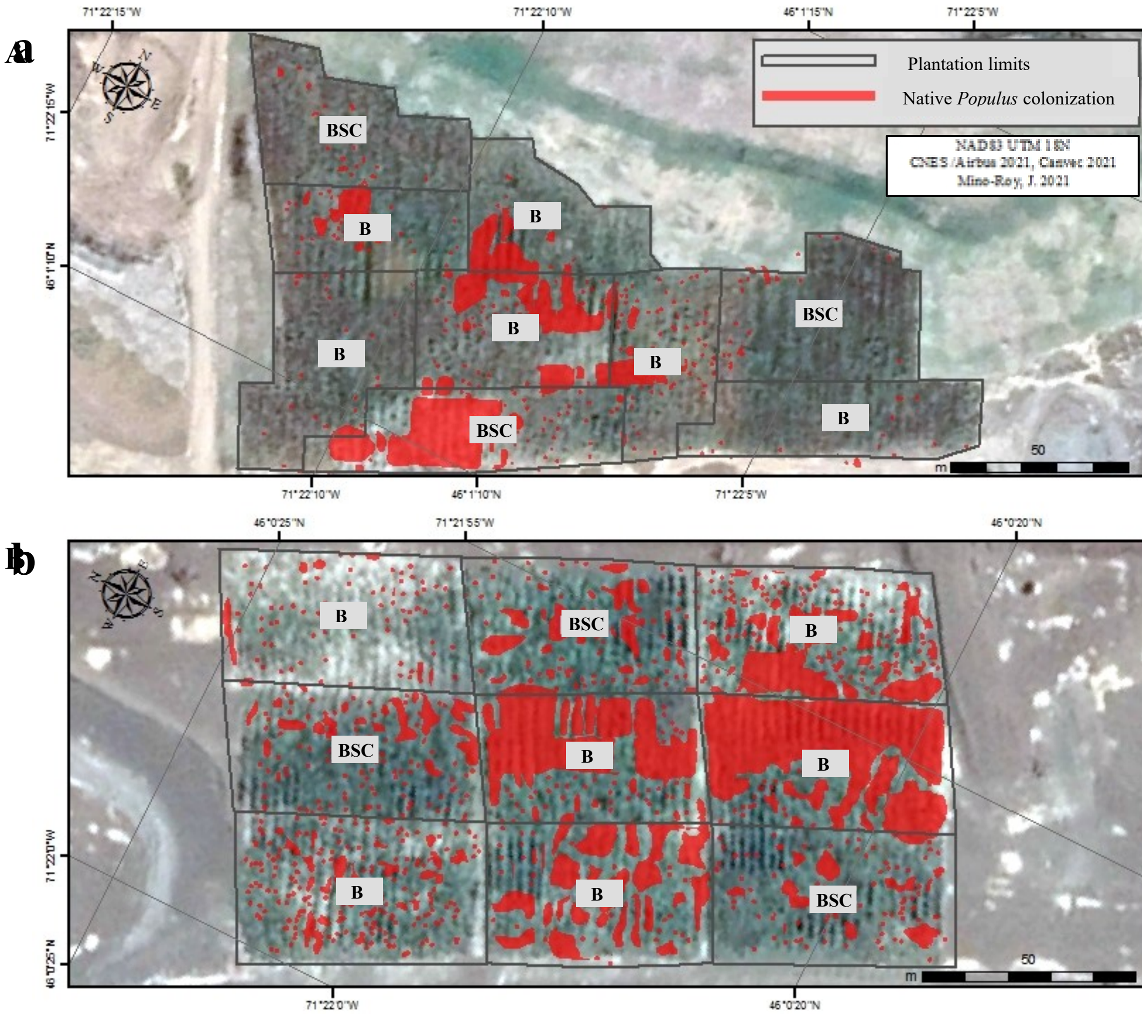

Figure 19. – Native *Populus* colonization (red areas) in waste rock pile plantation (a) and tailings plantation (b). B is for the basic mixture of biosolids and deinking sludges; BSC is for the basic mixture with the addition of Class B contaminated soils.

These findings offer some insight in respect to the characteristics of the soils required to maximise the success of natural regeneration of asbestos mine sites. In the case presented here, soil reconstruction followed by planting (but no seeding of grasses) appeared to be both an interesting economical and ecological approach. On the one hand, greater diversity of plant species, including native species, were reported in other reclamation settings using less intensive restoration techniques (Macdonald et al., 2015; Bruno Rocha Martins et al., 2020). On the other hand, tree planting can increase native tree regeneration by creating favorable microsites for tree seedling establishment and tend to accelerate natural succession (Parrotta et al., 1997; Carnevale and Montagnini, 2002; Bouchard et al., 2018). More research is needed to compare the benefits of restoration techniques of various intensities on native plant regeneration of asbestos mine sites.