Université du Québec à Rimouski

CHEMINS D'HIVER VS GRAVELÉS : DYNAMIQUE DE RÉGÉNÉRATION ET UTILISATION PAR LA GRANDE FAUNE DANS UN CONTEXTE DE CONSERVATION DU CARIBOU BORÉAL

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RÉSUMÉ

La forêt boréale, plus grand biome terrestre du Canada, constitue un important réservoir de ressources naturelles. Dans l'Est canadien, le déclin des populations boréales du caribou des bois (*Rangifer tarandus caribou*) est principalement attribué à la perte, l'altération et la fragmentation de son habitat, notamment en raison de l'aménagement industriel de la forêt et de la mise en place d'un dense réseau de structures linéaires. Si l'impact des chemins gravelés et des lignes sismiques sur la faune est bien documenté, l'effet des chemins forestiers d'hiver demeure quant à lui peu étudié. Construits sur un sol gelé et utilisés exclusivement en hiver, ces chemins pourraient influencer différemment la composition et la structure de la régénération végétale de même que les déplacements de plusieurs espèces fauniques. Mon projet visait à caractériser les effets de l'âge et du type de chemins forestiers – classe 4 (gravelés) et d'hiver (non gravelés) – et des caractéristiques du paysage sur la composition et la structure de la régénération végétale et, par la suite, de l'utilisation par la faune de ces chemins. L'hypothèse principale stipulait que les chemins d'hiver, impliquant une compaction moindre du sol, faciliteraient le rétablissement plus rapide de la végétation et limiteraient en conséquence les déplacements des prédateurs du caribou et de leurs proies alternatives plus rapidement que les chemins gravelés. Mon étude s'appuyait sur un réseau de 120 sites d'échantillonnage répartis sur 40 km de chemins forestiers le long desquels j'ai installé 160 caméras automatisées. Mes résultats ont indiqué qu'à âge égal, les chemins d'hiver supportaient une plus grande abondance de tiges de conifères et présentaient une compaction moindre. Pour la faune, l'effet du type de chemin s'est avéré moins net, étant déclassé par d'autres variables explicatives. L'ours noir (*Ursus americanus*) a été détecté plus fréquemment sur les chemins de classe 4, tandis que l'orignal (*Alces alces americana*) a montré une utilisation similaire des deux types de chemins. L'intensité d'utilisation par l'ours était positivement liée à la présence d'orignaux et négativement influencée par la proximité de forêts âgées de 90 à 120 ans. Le faible nombre de détections de loups (*Canis lupus*) a limité notre capacité à tester l'effet de certaines covariables liées à l'utilisation des chemins, tandis que les rares détections de caribous sont vraisemblablement attribuables à la très faible densité de l'espèce sur le territoire étudié. Ceci dit, les observations brutes suggèrent une utilisation plus élevée des chemins gravelés par ces deux espèces. Mes résultats ont montré que la compaction des sols constitue un facteur limitant important pour la régénération de la végétation sur les chemins forestiers, et que les chemins d'hiver – généralement moins compactés – présentaient un meilleur potentiel de restauration passive d'habitat forestier. Ces observations pourraient contribuer à orienter les stratégies de gestion dans un contexte de conservation du caribou forestier.

Mots-clés : aménagement forestier, caméras automatisées, caribou des bois, *Rangifer tarandus caribou*, chemins forestiers, régénération passive de la végétation, relations caribou-orignal-loup-ours, structures linéaires.

ABSTRACT

The boreal forest, the largest terrestrial biome in Canada, represents a major reservoir of natural resources. In Eastern Canada, the decline of boreal woodland caribou (*Rangifer tarandus caribou*) populations is primarily attributed to habitat fragmentation, notably caused by industrial forest management and the expansion of dense road networks. While the ecological impacts of gravel roads and seismic lines on wildlife are well documented, the effects of winter logging roads remain understudied. Built over frozen ground and used exclusively during winter, these roads may have different influences on vegetation regrowth and wildlife movement. This study aimed to characterize vegetation regeneration on two types of logging roads—gravel class 4 roads and ungraveled winter roads—and to quantify the effect of road type, age, vegetation recovery, and surrounding landscape features on their use by wildlife. The main hypothesis was that winter roads, being less compacted, would support faster vegetation recovery and consequently limit the movement of predators (*Canis lupus*, *Ursus americanus*) and their alternative prey, moose (*Alces alces americana*). Fieldwork included 120 sampling sites spread along 40 km of roads and 160 motion-triggered cameras. Results indicate that, at equal age, winter roads supported a higher abundance of conifer stems and showed lower soil compaction. Black bears were detected more frequently on class 4 roads, while moose used both road types similarly. Bear road use was positively associated with moose presence and negatively influenced by the proximity of 90–120-year-old forest stands. Due to the low number of wolf (*Canis lupus*) detections, our ability to test the effect of certain covariates related to road use was limited, while the rare caribou detections are most likely attributable to the very low density of the species in the study area. Nevertheless, raw observations indicate a higher use of gravel roads by both species. These findings highlight the role of soil compaction in limiting vegetation recovery and emphasize the interaction between black bears and moose in relation to forest road networks. Our findings show that soil compaction may be an important limiting factor for vegetation recovery on abandoned logging roads, and that winter roads—typically less compacted—may offer greater potential for natural regeneration. These insights could help inform management strategies in support of woodland caribou conservation in managed boreal landscapes.

Keywords: automated cameras; caribou–moose–wolf–bear interactions; forest management; forest roads; linear features; passive vegetation regeneration; woodland caribou (*Rangifer tarandus caribou*).

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TRANSFORMATION ANTHROPIQUES ET IMPACT GLOBAUX SUR LES ÉCOSYSTÈMES

Les activités humaines se multiplient à un rythme préoccupant, entraînant la perte, la fragmentation et l’altération des habitats naturels à travers le monde (Foley et al., 2005 ; Franklin et al., 2016 ; Trombulak & Frissel, 2000). Cette croissance démographique s'accompagne d'une intensification sans précédent de la transformation des milieux naturels, affectant la résilience des écosystèmes à l'échelle planétaire. Ces changements s’inscrivent dans un contexte plus large de crise de la biodiversité, où les activités humaines accélèrent le taux d’extinction des espèces à un niveau alarmant, ce qui conduit certains chercheurs à qualifier la situation actuelle de « sixième extinction de masse » (Ceballos et al., 2015). Depuis le début du XX^{e} siècle, la population mondiale est passée de 1,6 milliard à plus de 8 milliards d’individus, intensifiant la pression sur les écosystèmes terrestres (United Nations, 2022). L’urbanisation rapide, l’expansion agricole, l’aménagement forestier et l’extraction des ressources naturelles en découlant transforment profondément les paysages et compromettent le maintien de la biodiversité (Arneth et al., 2019 ; Foley et al., 2005).

Dans les écosystèmes forestiers, ces transformations se traduisent par la conversion de forêts matures, hétérogènes et structurellement complexes en peuplements plus jeunes, uniformes et souvent monospécifiques, un phénomène observé à l’échelle mondiale au cours des derniers siècles (Curtis et al., 2018 ; Luyssaert et al., 2014 ; Ramankutty & Foley, 1999). Les impacts de cette transformation varient selon les régions et les types d’écosystèmes**,** mais incluent notamment une perte de résilience écologique, une diminution des stocks de biomasse et une vulnérabilité accrue des paysages face aux événements climatiques extrêmes (Maxwell et al., 2016 ; McKinney, 2008 ; Powers & Jetz, 2019). L’intensification des perturbations anthropiques a également réduit la capacité des écosystèmes à se régénérer après des catastrophes naturelles (Gauthier et al., 2015), exacerbant ainsi les effets des incendies de forêt et des sécheresses, qui sont de plus en plus fréquents sous l'effet des changements climatiques (Drapeau et al., 2024).

La forêt boréale canadienne, qui couvre environ 75 % des terres forestières du pays, illustre particulièrement bien ces dynamiques (Ressources naturelles Canada, 2023). L'aménagement forestier y a profondément remodelé la structure d'âge des peuplements et l'organisation spatiale des paysages, au point que, dans certaines régions, l'empreinte cumulative des perturbations anthropiques dépasse désormais celle des perturbations naturelles telles que les incendies de forêt (Bergeron et al., 2006; Boucher et al., 2009 ; Cyr et al., 2009 ; Drapeau et al., 2000). Contrairement aux feux de forêt, qui génèrent une mosaïque d'habitats favorisant une biodiversité élevée, les coupes forestières industrielles produisent des peuplements en régénération relativement homogènes, qui diffèrent des forêts issues de perturbations naturelles tant par leur composition que par leur structure (Bergeron et al., 2006 ; Cyr et al., 2009 ; Harvey et al., 2002).

Cette simplification structurelle s'accompagne d'une fragmentation accrue des paysages forestiers, rendant les forêts matures et âgées de plus en plus rares et isolées, ce qui limite la connectivité écologique et affecte directement les relations faune–habitat, la dynamique des populations et les interactions écologiques à l'échelle des communautés (Fahrig, 2003). La raréfaction des forêts âgées se traduit notamment par un déclin des espèces spécialistes dépendantes d'attributs clés tels que les arbres sénescents et morts (Drapeau et al., 2000 ; Franklin et al., 2002), tandis que les espèces généralistes ou tolérantes aux milieux perturbés deviennent plus abondantes, modifiant la composition des communautés biologiques et le fonctionnement des écosystèmes forestiers (Drapeau et al., 2000).

En forêt boréale, ces changements se reflètent notamment dans l'augmentation de l'abondance de certaines espèces généralistes, comme l'orignal (*Alces alces americana*) et le cerf de Virginie (*Odocoileus virginianus*), qui profitent des habitats jeunes créés par la régénération des coupes forestières (Dussault et al., 2005 ; Kautz et al., 2021). L'orignal, en particulier, bénéficie de la diversité et de l’abondance des ressources alimentaires présentes dans ces habitats en régénération (Peek, 2007), favorisant ainsi des densités élevées dans les paysages aménagés (Frenette et al., 2020). En revanche, les ours noirs (*Ursus americanus*), omnivores opportunistes, exploitent à la fois ces milieux en régénération pour l’alimentation et les habitats plus matures pour la reproduction et l’hivernation (Brodeur et al., 2008), une plasticité comportementale qui leur permet de prospérer dans des paysages fragmentés.

LES STRUCTURES LINÉAIRES COMME ÉLÉMENTS DE FRAGMENTATION DES HABITATS FAUNIQUES DANS LES PAYSAGES AMÉNAGÉS

Les infrastructures linéaires associées à l’exploitation des ressources naturelles, telles que les routes et les lignes sismiques, fragmentent les milieux naturels en créant des barrières physiques et des corridors artificiels, ce qui perturbe les déplacements de la faune et réduit la connectivité entre les populations animales (Dickie et al., 2017 ; Jaeger, 2012 ; Trombulak & Frissell, 2000). Ces structures jouent également un rôle clé dans la modulation des interactions interspécifiques, notamment en facilitant les déplacements des prédateurs. En effet, les loups (*Canis lupus*) utilisent ces corridors pour se déplacer plus efficacement et de manière plus directionnelle, ce qui augmente leur efficacité de recherche, la fréquence des rencontres avec leurs proies et, par conséquent, la pression de prédation exercée sur certaines espèces, notamment l’orignal, dont les densités élevées sont favorisées par l’aménagement forestier (Dickie et al., 2017, 2020 ; James & Stuart-Smith, 2000 ; Whittington et al., 2005).

Cette dynamique entraîne des effets indirects sur d’autres espèces partageant ces milieux, notamment par l’accentuation du phénomène de compétition apparente (*sensu* Holt 1977) entre des espèces de grands cervidés, alors que l’abondance d’une proie secondaire plus productive et moins vulnérable soutient artificiellement des densités accrues de prédateurs qui induisent un déclin chez une proie moins productive et plus vulnérable (DeCesare et al., 2010 ; Serrouya et al., 2015). En forêt boréale aménagée, la compétition apparente est un processus écologique clé (Latham et al., 2011) qui sous-tend l’augmentation des populations d’orignaux, soutenue par la prolifération des jeunes forêts après coupe et entraîne une hausse des densités de loups (Seip 1992). Ces derniers, bénéficiant d'une abondance de proies alternatives, maintiennent des populations élevées et exercent une pression de prédation accrue sur d'autres ongulés moins productifs, comme le caribou des bois (*Rangifer tarandus caribou*, Wittmer et al., 2007). Contrairement aux orignaux, qui affichent des taux de reproduction plus élevés (Moore et al., 2023) et une plus grande tolérance aux paysages aménagés (Peek 2007), le caribou dépend d'habitats plus vastes et plus matures, où il limite volontairement ses interactions avec les prédateurs en évitant les zones récemment perturbées (Festa-Bianchet et al., 2011 ; Lesmerises et al., 2018). Cette stratégie devient inefficace dans un contexte où la perte des forêts matures et âgées réduit considérablement l'accès au caribou à ces habitats refuges, le rendant plus vulnérable aux attaques des loups (Wittmer et al., 2005).

ÉTAT DE LA SITUATION DU CARIBOU DES BOIS : UNE ESPÈCE EN DÉCLIN

Les populations boréales de la sous-espèce nord-américaine du caribou des bois (ci-après caribou boréal) sont présentes dans les forêts boréales du Canada, de l'Alaska et de certaines régions nordiques des États-Unis. Historiquement réparti sur un vaste territoire, le caribou boréal a subi un déclin important et une contraction de l'aire de répartition de ses populations au cours du dernier siècle, en grande partie à cause des perturbations anthropiques (Festa-Bianchet et al., 2011). Ce déclin s'explique par la perte et la fragmentation de son habitat (Natural Resources Canada, 2024), l'augmentation de la pression de prédation (DeMars et al., 2022) et les effets des changements climatiques qui ont modifié les conditions hivernales et la disponibilité des ressources alimentaires (Roesti et al., 2023). À l'échelle mondiale, les populations de caribous subissent des pressions variées selon l'emplacement de leur aire de répartition respective (Vors & Boyce 2009). En Amérique du Nord, le caribou migrateur connaît également un fort déclin, avec des troupeaux ayant perdu plus de 50 % de leurs effectifs en quelques décennies (Mallory & Boyce, 2018). Ces pertes sont principalement attribuées à la perturbation des aires de mise bas et à la compétition accrue avec d'autres ongulés favorisés par les modifications du paysage (Festa-Bianchet et al., 2011).

Au Québec, la situation du caribou boréal est particulièrement préoccupante. Autrefois retrouvé sur l'ensemble du territoire forestier boréal, il occupe désormais des zones de plus en plus restreintes (D'Orangeville et al., 2023), principalement dans des secteurs peu perturbés par l'activité humaine (Rudolph et al., 2017). Les populations de caribous au sud du 49^{e} parallèle sont les plus précaires, certaines ayant déjà disparu ou étant considérées comme non viables à long terme (COSEWIC, 2014). La perte croissante de représentation de la forêt mature et âgée – son habitat essentiel – découlant de nos pratiques d'aménagement extensif et du développement des chemins forestiers entraînent une augmentation des rencontres entre les caribous et leurs prédateurs, exacerbant ainsi le taux de mortalité des individus, notamment des faons (DeMars & Boutin, 2018 ; Vors et al., 2007 ; Wittmer et al., 2007). En réponse à cette situation préoccupante, plusieurs initiatives de conservation ont été mises en place, incluant des projets de restauration d'habitat, des mesures de protection des forêts matures et des programmes de surveillance des populations (Environnement Canada, 2012). Toutefois, les défis restent nombreux, et la nécessité d'une gestion adaptée et intégrée s'avère cruciale pour assurer la pérennité des populations de caribou boréal au Québec et ailleurs en Amérique du Nord. Face à ce déclin préoccupant, la mise en place de stratégies intégrées de conservation et de restauration de son habitat devient une priorité. La restauration des habitats dégradés, notamment des chemins forestiers abandonnés, représente une des solutions prometteuses pour rétablir des conditions plus favorables au retour du caribou boréal (aussi appelé « caribou des bois » au Québec ; Ministère des Forêts, de la Faune et des Parcs, 2023). Comprendre les mécanismes de régénération la végétation et leur influence sur les déplacements de la grande faune est toutefois essentiel afin d'orienter les pratiques de gestion des réseaux routiers pour ultimement structurer les travaux de restauration des habitats fauniques dans les écosystèmes boréaux sous aménagement.

CHEMINS FORESTIERS : TYPOLOGIES, IMPACTS ET ENJEUX ÉCOLOGIQUES

Les infrastructures linéaires, telles que les lignes sismiques, les routes forestières et les chemins d'exploitation, constituent des perturbations majeures dans les écosystèmes boréaux

(Trombulak & Frissell, 2000). Les impacts de ces structures varient selon leur nature, leur largeur et leur durée d'utilisation, certaines infrastructures temporaires ayant un effet moindre (Dabros et al., 2018). Dans l'Ouest canadien, les lignes sismiques ont été particulièrement étudiées pour mieux comprendre leurs impacts sur la végétation et la faune (Dabros et al., 2018 ; Lee & Boutin, 2006 ; van Rensen et al., 2015). Utilisées principalement pour l'exploration pétrolière et gazière, elles se caractérisent par des couloirs étroits et linéaires qui fragmentent les paysages forestiers. De nombreuses études ont montré que ces infrastructures modifient la structure et la composition de la végétation (Dabros et al., 2018 ; Lee & Boutin, 2006), ralentissent la régénération naturelle (Filicetti & Nielsen, 2018 ; van Rensen et al., 2015) et facilitent les déplacements de prédateurs tels que les loups, ce qui augmente la pression de prédation sur certaines espèces sensibles (Dickie et al., 2017 ; van Rensen et al., 2015). Contrairement aux perturbations naturelles (*i.e.* feux de forêt) qui très souvent stimulent la régénération du couvert forestier (Natural Resources Canada, 2024), les lignes sismiques restent ouvertes et utilisables pendant plusieurs décennies, limitant le retour à des conditions forestières fonctionnelles (Lee & Boutin, 2006). Ces impacts ont suscité un intérêt croissant pour mettre en place des initiatives de restauration des lignes sismiques, incluant des approches de régénération passive et active visant à rétablir la couverture végétale et la connectivité des habitats (Dabros et al., 2018 ; Filicetti & Nielsen, 2018 ; van Rensen et al., 2015).

Les impacts des infrastructures linéaires et d'autres perturbations anthropiques sur la biodiversité ont suscité un intérêt croissant de la recherche à l'échelle mondiale (Laurance et al., 2014 ; Torres et al., 2016). En effet, les modifications de composition et de structure des paysages induites par l'activité humaine perturbent divers processus écologiques et compromettent la résilience des écosystèmes (Laurance et al., 2014 ; Wu, 2006), accentuant la vulnérabilité des espèces sensibles face aux pressions environnementales (Pacifici et al., 2015). Bien que les chemins forestiers aient été largement étudiés pour leurs impacts écologiques (Coffin, 2007 ; Laurance et al., 2014), la majorité des travaux portent sur des routes permanentes ou de grande envergure (*i.e.* Forman & Alexander, 1998 ; Ibisch et al., 2016 ; Laurance et al., 2009 ; Trombulak & Frissell, 2000). En revanche, les chemins forestiers de petit calibre, tels que les chemins d'hiver, demeurent encore peu étudiés (mais voir Benoit-Pépin et al., 2024), malgré des impacts potentiellement comparables ou supérieurs sur le niveau de fragmentation des habitats et l'utilisation du territoire par la faune (Marchais et al., 2024).

Au Québec, le réseau routier forestier s'étend sur plus de 500 000 kilomètres (Données Québec, 2024), représentant une empreinte écologique significative dans les paysages boréaux aménagés (Boucher et al., 2014). Toutefois, tous les chemins ne s'équivalent pas, et il existe d'importantes différences entre les différents types de voies d'accès, que l'on parle de routes pavées, de chemins de classes (calibre) 1 à 4 et de chemins d'hiver. Les chemins de classes 1 et 2 correspondent à des routes principales utilisées toute l'année et ayant une emprise large (~35 m), avec une chaussée praticable de 8,5 à plus de 9,1 m de largeur. À l'inverse, les chemins de classes 3 et 4 présentent une emprise et une chaussée plus étroites (25 à 30 m, et 7,5 à 8,5 m, respectivement) et sont principalement destinés à des opérations forestières ponctuelles (MFFP, 2023). L'assise des chemins de classe 4, en particulier, est constituée de gravier compacté, ces chemins étant conçus pour supporter le passage de machinerie lourde. La pression exercée sur le sol entraîne une forte compaction, une variable reconnue pour ralentir considérablement la régénération naturelle (Ampoorter et al., 2012), ce qui nécessite généralement des interventions de restauration active impliquant la décompaction du sol pour faciliter le rétablissement d'une couverture végétale fonctionnelle (Nazari et al. 2021). À l'inverse, les chemins d'hiver sont à vocation temporaire, étant construits à l'aide de la neige et de branchage et bénéficiant du gel du substrat pour offrir un soutien au passage de la machinerie.

Dans une des rares études s'étant intéressées aux chemins d'hiver, Benoit-Pépin et al. (2024) ont montré que ces chemins peuvent soutenir une régénération végétale plus dense que les chemins gravelés, en raison notamment d'une compaction plus faible du sol, en particulier dans les zones présentant un bon enracinement latéral à partir de la végétation riveraine. Bien que leur étude se concentrait principalement sur les effets de la végétation sur la présence de la faune, ils ont observé que l'abondance de végétation influençait positivement l'utilisation de certains grands herbivores, indiquant un lien entre structure du couvert et intérêt pour la faune. Leurs résultats mettent ainsi en lumière l'intérêt de mieux explorer les facteurs écologiques et structurels qui influencent la régénération végétale selon la méthode de construction du chemin.

Quelques études ont également quantifié la dynamique de revégétalisation sur les chemins forestiers en fonction de leur âge, des conditions locales des sols et du contexte paysager. Par exemple, St-Pierre et al. (2021) ont montré que les types de chemins présentaient des profils de végétation distincts, influencés à la fois par la compaction du sol et par la structure du paysage environnant. Marchais et al. (2024) ont noté que la couverture végétale augmentait avec l'âge du chemin, mais que ce patron variait selon l'humidité du site, la densité du couvert latéral induit par la végétation, et la pente. Une étude récente de Meecham (2023) a montré que la régénération sur des routes forestières inactives varie fortement selon les conditions du site environnant (sol, contexte paysager et traitements appliqués), ce qui influence l'abondance de la composition des communautés végétales.

Dans un contexte de restauration active, Lacerte et al. (2021) ont mis en évidence que le retour de la végétation pouvait être accéléré par le traitement mécanique du sol, particulièrement sur les chemins plus anciens ou fortement compactés. De plus, Dabros et al. (2018), Filicetti & Nielsen (2018) et van Rensen et al. (2015) ont tous montré que la régénération naturelle sur les anciennes infrastructures linéaires peut être lente et incomplète, notamment dans les sites plus secs ou fortement perturbés, mais qu'elle peut être améliorée par certaines pratiques de réhabilitation ciblées, comme le scarifiage ou l'ajout de matières ligneuses.

Mon mémoire de maîtrise visait donc à approfondir nos connaissances sur les processus de régénération végétale spécifiques aux chemins gravelés et aux chemins d'hiver non gravelés (Figure 0.1), en particulier ceux abandonnés depuis plusieurs décennies, et à évaluer leur utilisation par les principales espèces de grands mammifères impliqués dans le système impliquant le caribou boréal, ses prédateurs et compétiteurs.

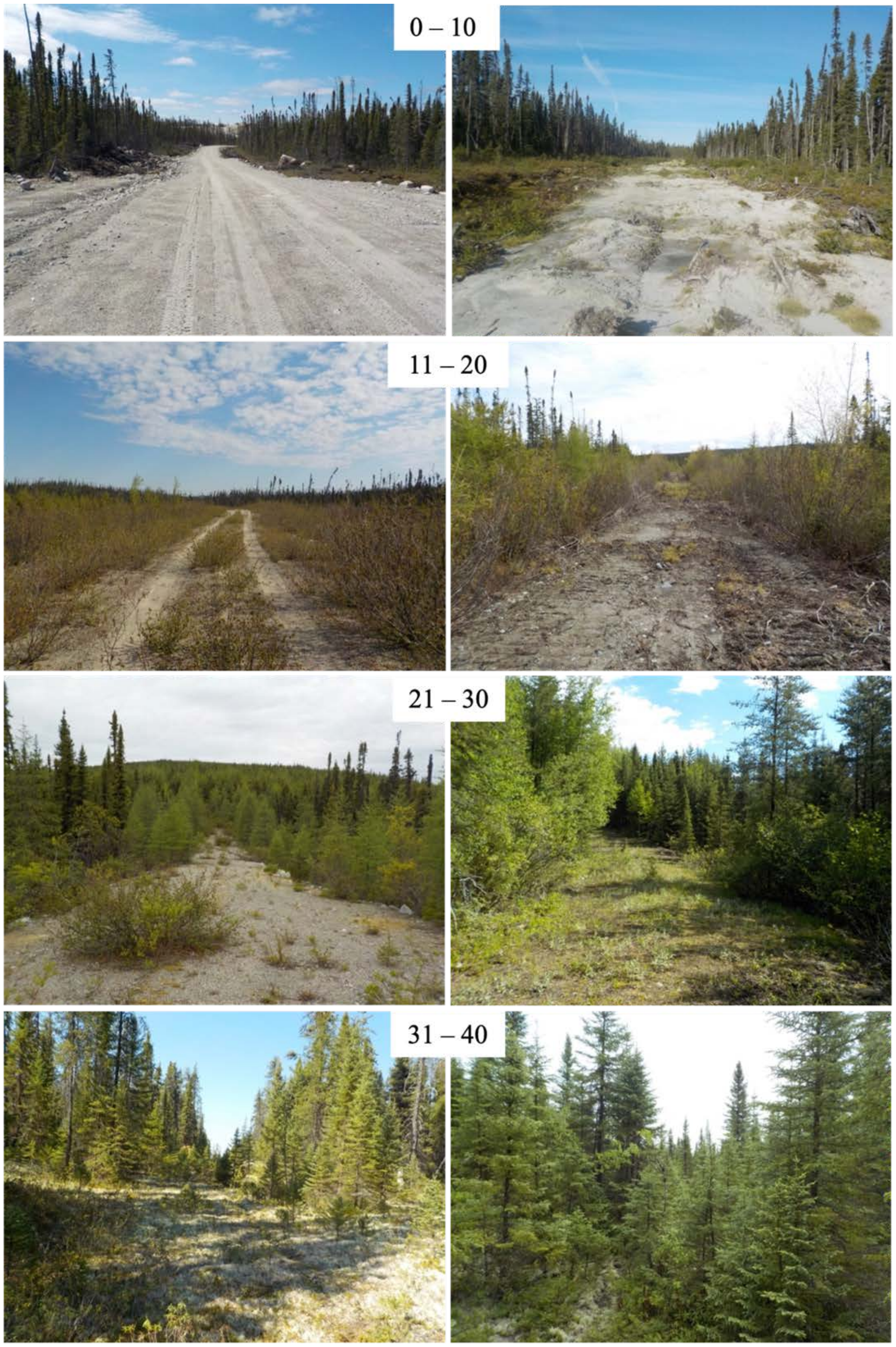

Figure 0.1: État de la régénération végétale passive de chemins forestiers de classe 4 (à gauche) comparativement aux chemins forestiers d’hiver (à droite) dans quatre classes d’âge situé dans le Nord-du-Québec, CA (crédit photo Lisandre Robichaud)

OBJECTIFS, HYPOTHÈSES ET SURVOL DES PRINCIPAUX RÉSULTATS

Ainsi, mon étude s’est intéressée à analyser la dynamique de régénération de la végétation sur les chemins forestiers de petit calibre, c.-à-d. les chemins de classe 4 et les chemins d’hiver, et à quantifier leur utilisation par la grande faune, notamment le caribou boréal et ses prédateurs, tout en tentant d’expliquer la variabilité de ces deux phénomènes. Ultimement, comprendre ces dynamiques s’avérait essentiel afin d’évaluer l’impact de telles infrastructures linéaires sur les relations entre le caribou et ses prédateurs et compétiteurs, de manière à orienter les stratégies de gestion du réseau routier de même que les approches de restauration adaptées aux enjeux de conservation du caribou.

Ces deux types de chemins se distinguant principalement par leur méthode de construction respective, il était attendu, pour mon premier objectif, que les chemins en gravier, en raison de leur compaction plus importante du sol, limitent l’établissement des espèces végétales pour un même âge donné depuis leur construction. À l’inverse, les chemins d’hiver, construits sans excavation, étaient supposés permettre une recolonisation plus rapide de la végétation. Mon deuxième objectif, portant sur les réponses des grands mammifères aux différences de structure et de composition de la végétation, reposait sur l’hypothèse que la densité et la nature du couvert végétal modulent différemment l’attractivité et l’utilisation des chemins selon les espèces. Plus spécifiquement, nous nous attendions à ce qu’une augmentation du couvert végétal le long des chemins entraîne une diminution de leur utilisation par certaines espèces généralistes, telles que le loup, l’orignal et l’ours noir, et qu’elle favorise davantage la présence du caribou, notamment sur les chemins d’hiver plus anciens par rapport aux chemins de classe 4, particulièrement sur les chemins d’hiver plus anciens.

Mes résultats montrent que la régénération de la végétation diffère selon le type de chemin. Bien que la compaction du sol n’ait pas été systématiquement retenue parmi les covariables des modèles les plus parcimonieux selon les critères de sélection, les analyses où cette variable était explicitement considérée indiquent qu’elle joue un rôle structurant dans les patrons de régénération végétale. Globalement, certains modèles indiquaient un effet marginal du type de chemin, alors que d’autres intégraient aussi des variables structurelles et environnementales (milieux adjacents, perturbations). Concernant la faune, les loups étaient observés plus fréquemment sur les chemins gravelés, tandis que les orignaux et les ours noirs utilisaient davantage les chemins plus âgés, possiblement en raison d’une plus grande disponibilité en ressources végétales. Mes résultats fournissent ainsi des bases empiriques utiles pour orienter les stratégies de restauration adaptées aux milieux aménagés, en soulignant l’importance de considérer conjointement le type, l’âge et les caractéristiques des milieux adjacents dans l’évaluation des impacts écologiques des chemins forestiers. Mon étude contribue enfin à une meilleure compréhension des processus de régénération végétale sur les chemins forestiers abandonnés et de leur influence sur leurutilisation des chemins par les grands mammifères, en apportant des connaissances applicables à la gestion du réseau routier forestier et à la conservation du caribou en milieu aménagé.

CHAPITRE 1

À LA CROISÉE DES CHEMINS : CHOISIR LE MOINDRE DE DEUX MAUX POUR LA CONSERVATION DU CARIBOU FORESTIER

1.1 RÉSUMÉ EN FRANÇAIS DE L'ARTICLE

Depuis 2002, les populations boréales du caribou des bois (*Rangifer tarandus caribou*) sont désignées comme espèce menacée en vertu de la Loi sur les espèces en péril du Canada, en raison d'une pression de prédation accrue liée à la perte, l'altération et la fragmentation de son habitat. Les infrastructures linéaires, telles que les chemins forestiers issus de l'exploitation, facilitent l'efficacité des prédateurs, mais leur impact sur les interactions proie–prédateur peut varier selon le type de chemin. À l'aide de relevés de végétation et de 160 caméras automatisées déclenchées par le mouvement, nous avons évalué la régénération végétale et l'utilisation par la grande faune de deux types de chemins forestiers dans l'Est du Canada : les chemins d'hiver temporaires, construits sur sol gelé et désaffectés après une seule saison, et les chemins gravelés de classe 4, aménagés sur sol minéral et entretenus pendant plusieurs années. Comme les chemins d'hiver diffèrent des chemins gravelés par leur mode de construction et leur durée d'utilisation, nous avons examiné comment leurs variables environnementales influençaient la composition végétale et l'utilisation des chemins par la faune pour ces deux types de chemins. Nos résultats montrent que la régénération de la végétation était principalement déterminée par l'âge du chemin et la compaction des sols, l'effet du type de chemin apparaissant plus subtil et dépendant du contexte. Les tiges de conifères étaient plus abondantes sur les chemins d'hiver, tandis que les tiges d'aulnes étaient plus fréquentes sur les chemins gravelés, reflétant des réponses spécifiques aux conditions des chemins. Les données issues des caméras ont révélé que les chemins gravelés de classe 4 étaient plus fréquemment utilisés par l'ours noir (*Ursus americanus*) et l'orignal (*Alces alces americana*) que les chemins d'hiver, alors qu’aucune espèce n’utilisait davantage les chemins d’hiver que les chemins gravelés. Le faible nombre de détections de caribous et de loups (*Canis lupus*) a contraint la réalisation d’analyses quantitatives, reflétant probablement leurs faibles densités locales et l’évitement connu des chemins par le caribou. Ces résultats mettent en lumière l’impact à long terme de la compaction des sols sur la régénération des chemins forestiers et indique que la structure et l’état de régénération des chemins pourraient influencer les réponses écologiques de manière indirecte, notamment via la végétation.

1.2 TAKING THE HIGH ROAD: THE LESSER OF TWO EVILS FOR BOREAL CARIBOU CONSERVATION

ABSTRACT

Since 2002, the boreal population of woodland caribou (*Rangifer tarandus caribou*) has been listed as Threatened under Canada's Species at Risk Act due to increased predation pressure driving the loss, alteration, and fragmentation of habitat. Linear features such as timber harvesting roads are key in facilitating predator efficiency, but their impact on predator–prey interactions may vary depending on road type. Using vegetation surveys and 160 motion-activated camera traps, we assessed vegetation regeneration and large mammal use of two types of logging roads in eastern Canada: temporary winter roads, built over frozen ground and decommissioned after a single season, and class 4 gravel roads, constructed on mineral soil and maintained for multiple years. As winter roads may offer a less ecologically disruptive alternative, we examined how environmental variables influenced vegetation composition and mammal road use on both road types. We found that vegetation regeneration was primarily driven by road age and soil compaction, with road-type effects being subtle and context dependent. Coniferous regeneration was more pronounced on winter roads, whereas alder regeneration was more frequent along gravel roads, reflecting species-specific regeneration patterns in response to road characteristics. Camera trap data revealed that black bears (*Ursus americanus*) and moose (*Alces alces americana*) extensively used class 4 roads when compared to winter roads, with their photographic rates positively correlated, whereas no species used winter roads more intensively than gravel roads. In contrast, low detection rates of caribou and wolves (*Canis lupus*) precluded quantitative analysis, likely reflecting low local densities and caribou's known road avoidance. These findings highlight the long-term impact of soil compaction on forest road regeneration and indicate that road structure and regeneration state may indirectly shape ecological responses through their effects on vegetation.

INTRODUCTION

Human activities are rapidly transforming natural habitats, leading to widespread loss, fragmentation, and degradation (Foley et al., 2005; Trombulak & Frissel, 2000). Among the key drivers of land-use change, timber extraction has profoundly altered the age-class distribution of boreal forests, favoring early successional stands at the expense of older, structurally complex forests (Drapeau et al., 2000; Mackey et al., 2024). This shift has severe repercussions for wildlife species that depend on mature and old, intact forests, notably the boreal population of woodland caribou (*Rangifer tarandus caribou*, hereafter boreal caribou), whose decline has been directly linked to habitat disturbance (Festa-Bianchet et al., 2011; St-Laurent et al., 2022).

Boreal caribou are strongly associated with undisturbed coniferous forests (Rettie & Messier, 2000), particularly stands dominated by pines and spruces (Courtois et al., 2007), where they reduce predation risk by spacing away from more productive habitats that support higher predator densities. However, the conversion of mature forests into regenerating stands has altered this strategy by increasing the abundance of moose (*Alces alces americana*; Brodeur et al., 2008) and white-tailed deer (*Odocoileus virginianus*; Dawe & Boutin, 2016), which in turn sustain greater wolf (*Canis lupus*) populations (Latham et al., 2011). This has intensified predation pressure on caribou through apparent competition (Holt, 1977; Wittmer et al., 2005), as wolves adjust their hunting behavior to exploit more abundant prey (Latham et al., 2013), with detrimental consequences for caribou (James et al., 2004; Wittmer et al., 2007). Black bears (*Ursus americanus*) also benefit from these regenerating habitats due to the abundance of fruiting shrubs and young shoots, which support higher population densities and contribute to an elevated caribou calf mortality (Gustine et al., 2006; Pinard et al., 2012). As a result, the boreal caribou population has been declining across its range, leading to its designation as Threatened under the Canadian *Species at Risk Act* in 2002 (COSEWIC, 2014).

Extensive road networks created by timber harvesting modify predator–prey dynamics by fragmenting caribou habitat and facilitating predator movement by connecting their access to prey throughout managed landscapes (Leblond et al., 2013; Mumma et al., 2018). In the Québec province (eastern Canada) alone, public datasets track over 500,000 kilometers of logging roads (Données Québec, 2024). Unlike narrow seismic lines (~2 m wide), logging roads are considerably wider (~25-35 m) and require substantial excavation and compaction to support heavy machinery over extended periods (Ampoorter et al. 2012; Dabros et al., 2018). These structural modifications significantly alter environmental conditions by increasing light exposure, reducing humidity, modifying wind patterns, and enhancing shrub growth, all of which hinder the regeneration of pre-disturbance vegetation communities once roads are abandoned (Finnegan et al., 2018; Matlack, 1993; St-Pierre et al., 2021).

Logging roads have thus been closely associated with boreal caribou decline across Canada, as they contribute to habitat loss and increased predation risk (Environment Canada, 2011; Rudolph et al., 2017). The expansion of these networks facilitates predator mobility, enabling them to travel faster and farther while increasing their encounter rates with prey (Dickie et al., 2017b; James & Stuart-Smith, 2000; Whittington et al., 2005). This enhanced movement leads to both a numerical response, where predator populations increase in response to greater prey availability, and a functional response, where predators become more efficient at locating and capturing prey as availability rises (Ray, 2014). However, as vegetation regrowth progresses along abandoned roads, the increasing height and density of lateral vegetation can gradually limit the movements of moose, wolves, and bears (Dickie et al., 2017a; Tigner et al., 2014; Van de Vuurst et al., 2017). To mitigate these effects, passive and active restoration approaches have been studied extensively. Passive restoration consists of allowing natural vegetation recovery without human intervention, whereas active restoration involves targeted efforts such as road closure, soil decompaction, or tree planting to accelerate forest regeneration (Bowman et al., 2010; Lacerte et al., 2021, 2022; Scalbert et al., 2023; St-Pierre et al., 2021, 2022). In eastern Canada, St-Pierre et al. (2021) linked slow, passive vegetation regrowth and variation in species composition to the soil compaction of gravel roads, while Lacerte et al. (2021) reported that decompacting roadbeds mechanically was needed to facilitate vegetation regrowth. Both studies recommended using road construction techniques that minimize soil compaction and vegetation damage in caribou ranges, such as winter roads, which require minimal excavation since frozen ground has sufficient load-bearing capacity.

Given the importance of road regeneration in mitigating caribou habitat disturbance, our study focused on comparing vegetation regrowth and wildlife use on two road types that account for 30% of logging roads in eastern Canadian landscapes (Données Québec, 2024): winter logging roads and class 4 logging roads, the latter being compacted gravel roads of similar width. We examined how road type, age, and both local (*e.g.*, soil composition, vegetation cover) and landscape-scale (*e.g.*, habitat composition, connectivity) characteristics influence vegetation dynamics and large mammal use. We hypothesized that the construction methods used for gravel roads result in greater soil compaction than winter roads, thereby limiting vegetation regrowth. We expected that vegetation dynamics would be influenced not only by road type and age, but also by local and landscape-scale conditions. Specifically, we predicted (P1) that winter logging roads would show greater vegetation regrowth than class 4 roads, and that this difference would become more pronounced over time. We also predicted (P2) that older roads would support higher coniferous stem densities, and (P3) that compaction—regardless of road type—would negatively affect stem density. Regarding large mammals, we hypothesized that increasing vegetation cover along roads would limit movements for species such as wolves, moose, and bears, while favoring caribou presence. Therefore, we predicted (P4) that road use by wolves, moose, and bears would decline with increasing vegetation cover, particularly on older winter roads, and (P5) that caribou would use older winter roads more frequently than class 4 roads due to their denser vegetation cover.

MATERIALS AND METHODS

Study area

We tested our hypotheses and predictions in the Nord-du-Québec region, in the province of Québec (eastern Canada). The study area covers ~ 17,700 km^2 across the ranges of two local populations of boreal caribou (Figure 1.1), namely Temiscamie and Assinica (49°–52°'N, 70°–8O°'W; Rudolph et al. 2017) and presents a mean elevation of 374 m. Located in the boreal biome, this area is subject to a continental subpolar climate and receives approximately 700–900 mm of precipitation per year. Variations in average monthly temperatures range from -19°C (January) to +16°C (July; Robitaille and Saucier 1998). The vegetation is representative of the black spruce (*Picea mariana*)–feather moss (*Pleurozium schreberi*) bioclimatic subdomain, in the continuous boreal forest subzone (Robitaille and Saucier 1998). The forest cover is dominated by black spruce in association with jack pine (*Pinus banksiana*) and balsam fir (*Abies balsamea*), with an undergrowth mainly composed of hypnaceous mosses and ericaceous shrubs (Rudolph et al. 2017). Timber harvesting activities and short fire cycles have shaped the forest dynamics of our study area (Bergeron et al., 2006; Gauthier et al. 2015). The most common species of large mammals are the moose, woodland caribou, black bear, and gray wolf.

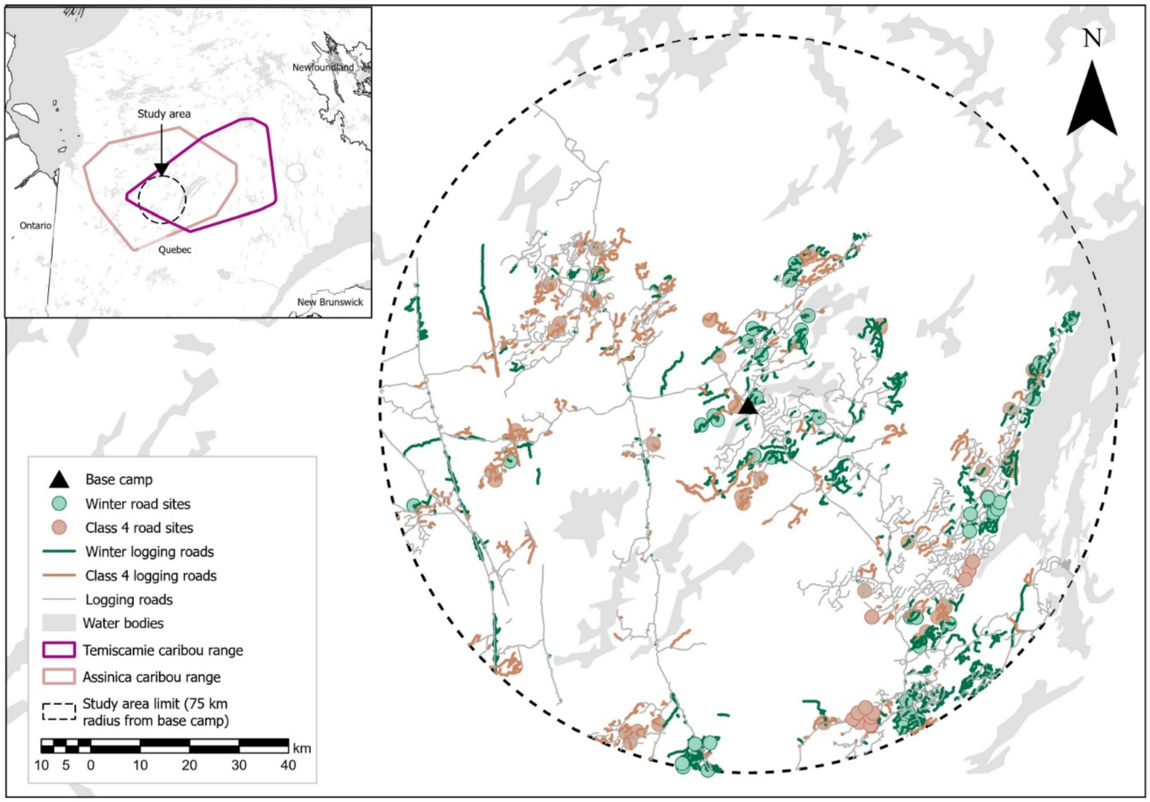

Figure 1.1: Location of the study area, showing the sampled logging road segments in a 75 km radius from a base camp used to study vegetation regrowth and animal use of roads in boreal caribou range located ~120 kilometers north of the town of Chibougamau in Northern Québec (Canada).

Selection of road segments and sampling design

We selected road segments using the 1: 20,000 Routard numerical map published by the Ministère des Ressources naturelles et des Forêts du Québec (hereafter MRNF), which is updated annually. We sorted logging roads based on their classification category and their age (*e.g.*, time since construction or major maintenance). To be selected, road segments needed to be within a 75 km radius from our basecamp (74.5°W, -50.4°N; see Figure 1.1). Among the 1 km road segments retained by these filters, we randomly selected 20 segments of gravel logging roads (class 4 in the Routard) and 20 segments of winter roads, both covering an age gradient (0–10 years old, 11–20 years old, 21–30 years old, 31–40 years old, with 5 replicates per combination of age category and road types) for a total of 40 road segments (*n* = 40). To avoid spatial autocorrelation, reduce pseudoreplication and ensure the representativeness of our data, the distance separating two nearby road segments was of at least 2 km.

Characterization of vegetation and landscape structure

Data collection was conducted from May 31 to August 14, 2021. In each 1 km road segment (1 segment/road), we characterized the regenerating vegetation on the roadbed in a 16 m^2 (2.26 m radius) circular plot, using three plots per road segment (located 400 m apart), for a total of 120 sampling plots distributed on 40 road segments (see Fig. 1.2 for a schematic of a sampling plot). In each of the plots, we assessed the lateral cover (*e.g.*, the degree of vegetation closure) and the vertical cover (*e.g.*, canopy closure) in the center of the right-of-way, using respectively a vegetation profile board (Nudds et al., 1977) and a 50 cm x 50 cm square frame (St-Laurent et al., 2007) held at 1 and 2 m above the ground. Vegetation height was measured in two vertical strata: saplings (1–5 m) and trees (>5 m). For each stem, diameter at breast height (DBH; measured at 1.3 m above ground) was recorded, and stems were counted by species within the road right-of-way (see St-Pierre et al., 2021 for more details). Following St-Pierre et al. (2021), we used a homemade penetrometer to measure soil compaction (1/depth reached by the penetrometer) in each plot. To do so, we dropped a 4.53 kg mass (welded to a metal tube) onto a one-end profiled metal rod (1.30 m long, 1 cm diameter) graduated at every 2.5 cm. After dropping the mass from a 20 cm height, we read the depth reached by the metal rod (a greater depth value indicating a less compacted soil). We repeated this procedure 3 times per tire rut (when not visible, their location was estimated) as well as three times between them, for a total of nine measurements per sampling plot, and then calculated the average soil compaction for the plot. We also noted the orientation (in degrees) of the road segment that we then converted to radians.

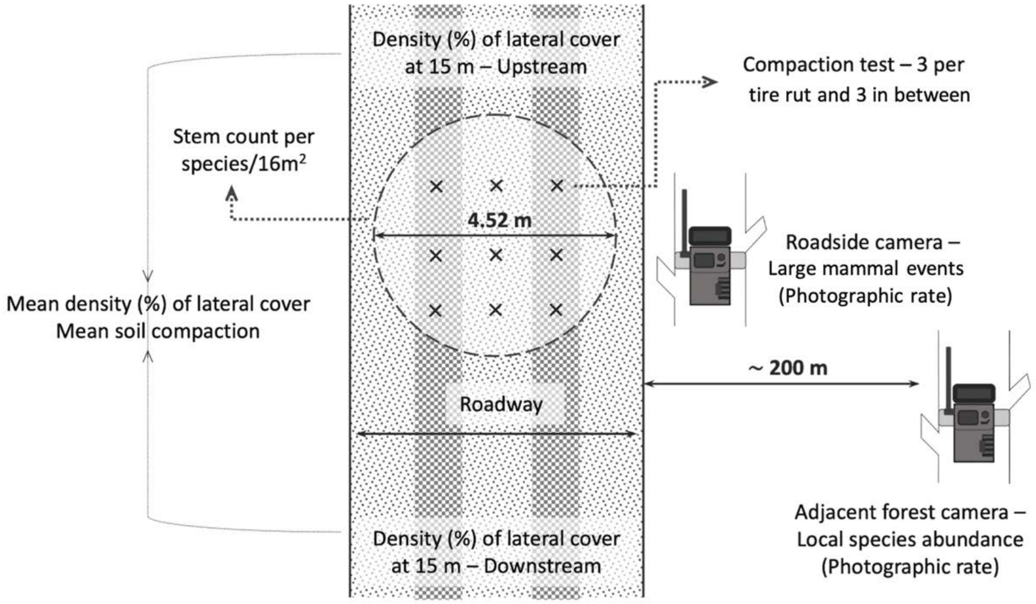

Figure 1.2: Schematic representation of a sampling plot and measurements taken at a camera station (3 per road segment) on logging roads in boreal caribou range, Northern Québec (Canada).

We built a digital elevation model (DEM) using the 1: 50,000 topography maps published by the Canadian government (Government of Canada 2022) to extract the elevation and slope for each sampling plot. To consider the potential influence of surrounding roadside habitats on the vegetation regrowth of logging roads, we classified and categorized the 1: 20,000 digital forest cover maps published by the MRNF. We used the annual updates of the 4th decadal inventory to obtain the most recent version of the forest cover over our study area. The resulting landscape was classified into 12 landcover types, namely 0–5year-old cutovers, 6–20-year-old cutovers, 50–120-year-old deciduous/mixed stands, wetlands, open lichen woodlands, bodies of water (lakes, rivers, and flooded areas), open/non-regenerated stands, regenerated stands, 50–70-year-old coniferous stands, 90–120-year-old coniferous stands, anthropogenic activities (*e.g.*, electric transmission lines, roads, gravel pits) and natural disturbances (*e.g.*, fires, floods, insect outbreaks; Table 1.1). The data collected on each road segment and landcover type were then divided into three groups of explanatory variables: road structure (road type, age category, soil compaction), topography (orientation, elevation, slope), and environment (landcover types). While road type offers a practical classification for management, it may not capture the ecological drivers of vegetation recovery. For plants, soil compaction—often higher on gravel roads—can limit root growth, water infiltration, and seedling establishment (Ampoorter et al., 2012; Picchio et al., 2020; Solgi et al., 2015). Because construction practices differ between road types, compaction and road type are often linked, making it difficult to separate their effects. Past studies that did not account for compaction have found road-type effects that may reflect this confounding relationship (*e.g.*, Benoit-Pépin et al., 2024; Leblond et al., 2011). To better isolate these influences, we compared models with and without the compaction variable. For large mammals, road type may influence use indirectly, through vegetation structure. As roads revegetate, increased cover can reduce visibility, movement efficiency, or perceived safety for some species (Dickie et al., 2020). Thus, wildlife may respond more strongly to vegetation conditions than to road structure itself. We thus included road type in all models to reflect its central importance to our study design and to assess both its direct and vegetation-mediated effects. The percentage of landcover types was calculated within 250 m, 500 m, 750 m, and 1 km radii buffers centered on each sampling site. All spatial analyses and covariate extractions were carried out using ArcGIS 10.2.2 (ESRI, 2012) and R 4.2.1 (RStudio Team, 2020).

Table 1.1: Description of the local environmental covariates determined at each sampling site and land cover types, Northern Québec (Canada).

Variables	***Description***
Local Environmental Variables	
Age (years)	Time since road construction
Slope (°)	Estimated from the slope raster obtained from a Digital Elevation Model (DEM)
Road compaction (cm)	Mean compaction measured using a penetrometer (1/depth)
Road orientation (rad)	Determined using a compass
Lateral cover (%)	Mean percentage of lateral cover at 15 m in both directions parallel to the road
Vertical cover (%)	Mean percentage of vertical cover
Road density (%)	Road density in a 500 m radius buffer (km/km^2)
Elevation (m)	Elevation interpolation
Latitude and longitude	Determined using a handheld GPS
Land cover types	
0–5-year-old cutovers	Proportion of 0–5 years cutovers in a 500 m radius buffer
6–20-year-old cutovers	Proportion of 6–20 years cutovers in a 500 m radius buffer
50–120-year-old deciduous/mixed stands	Proportion of 50–120-year-old deciduous/mixed stands in a 500 m radius buffer
Wetlands	Proportion wetlands in a 500 m radius buffer
Open lichen woodlands	Proportion of open lichen woodlands in a 500 m radius buffer
Open/non-regenerated stands	Proportion of open/non-regenerated stands in a 500 m radius buffer
Regenerated stands	Proportion of 20–50-year-old regenerated stands in a 500 m radius buffer
50–70-year-old coniferous stands	Proportion of 50–70-year-old coniferous stands in a 500 m radius buffer
90–120-year-old coniferous stands	Proportion of 90–120-year-old coniferous stands in a 500 m radius buffer
Natural disturbances	Proportion of 0–20-year-old natural disturbances in a 500 m radius buffer
Anthropogenic activities	Proportion of anthropogenic land uses (*e.g.*, electric transmission lines, roads, gravel pits) within a 500 m radius buffer
Water	Proportion of lakes, rivers, and flooded areas in a 500 m radius buffer

Assessing the intensity of use by large mammals

We employed motion-activated cameras to quantify the occurrence and the intensity of use by caribou, moose, wolves, black bears, and humans (*e.g.*, recreationists, vehicles) along road segments. To do so, 120 motion-activated cameras (Spypoint Force Dark V2.05.20) were installed along the edge of our selected 1 km road segments. Cameras were installed 100 m, 500 m and 900 m from the beginning of the road segment, for a total of three cameras per km (hereafter "roadside cameras"), facing the roadway diagonally to maximize the distance travelled by an animal in front of the camera. To quantify and assess the effect of changes in the local density of individuals of the different species, we installed an additional motion-activated camera ~200 m from the road (following Selvan et al., 2014; St-Pierre et al., 2022), hereafter "adjacent forest cameras." These cameras were randomly distributed within our road segments for a total of 40 forest cameras. Based on Lacerte et al. (2022), we used the photographic rate as a proxy of local abundance using the network of cameras deployed in the forest adjacent to road segments. Each camera (roadside and adjacent forest) was installed ~1.2 m from the ground, a height at which we would detect medium to large terrestrial mammals (Pettigrew et al., 2021; St-Pierre et al., 2021). Then, every camera (roadside and adjacent forest) was programmed to record three pictures at 5 seconds intervals between detections and note the time and date of camera deployment. Camera traps were active from the end of May to mid-August 2021.

We used the software Timelapse2 (Greenberg, 2016) to classify the photos of four large mammal species: wolves, bears, moose, and caribou. Following Keim et al. (2019), we classified every camera detection of the studied species as an "event". In cases where more than one individual of the same species appeared on a given photo, the event reflected the number of individuals on that frame (*e.g.*, an adult caribou and a calf = two events). In the case where an individual repetitively triggered the camera trap, we avoided duplicate counts of the same animal by considering the photographs as one event if they were taken within a 30-minute interval (Rovero and Zimmermann, 2016; Tattersall et al., 2020a, b). The number of operational days for each camera was noted for both road and forest cameras, allowing data comparability across the 160 camera traps. We then used the frequency of events recorded per species at each camera station during the operational days (data collection period) to measure the intensity of use for every camera (O'Brien et al., 2010; Tobler et al., 2015).

Statistical analyses

Drivers of vegetation regrowth on logging roads

We used a partial canonical correspondence analysis (pCCA: ; Kenkel, 2006; ter Braak & Verdonschot, 1995) to represent the variation in composition and structure of roadside vegetation regrowth across sampling sites, based on the number of stems recorded, while simultaneously considering their response to a set of explanatory environmental variables. Unlike univariate analyses, multivariate approaches such as canonical methods are well suited for examining relationships between multiple variables (here, vegetation regrowth and environmental factors) measured at the same sampling sites, which serve as statistical units (Legendre & Legendre, 1998).

To account for differences in vegetation structure and composition, we grouped all stems into three distinct categories: 1) coniferous (black spruce, balsam fir, tamarack *Larix laricina*, and jack pine), 2) deciduous (willow *Salix* sp., paper birch *Betula papyrifera*, and trembling aspen *Populus tremuloides*), and 3) alder (*Alnus* sp.). These categories reflect ecologically relevant vegetation types for large mammals, as coniferous stems contribute to caribou habitat suitability (Courbin et al., 2009; Ray, 2014), deciduous stems provide forage for moose and bears (Bastille-Rousseau et al., 2011; Laurian et al., 2012; Lesmerises et al., 2015), and fast-growing shrubs such as alders can suppress caribou-preferred ground vegetation, including mosses, sphagnum, and lichens (Gill et al., 2014; Myers-Smith et al., 2006).

According to Zuur et al. (2007), pCCAs cannot be performed when some sampling units contain null values for a given variable. Consequently, we removed 68 sampling plots (56.7% of our 120 plots) in which no stems were recorded for this specific analysis. We applied a Hellinger transformation to the remaining 52 plots to reduce the sensitivity of pCCAs to the over-representation of certain species in a limited number of plots (Kenkel, 2006; Legendre & Gallagher, 2001).

Prior to analysis, we calculated the proportion of different land cover types within buffers surrounding each sampling plot (as defined above in section *2.3 – Characterization of vegetation and landscape structure*). To select the most appropriate buffer size for our multivariate analysis, we computed four pCCAs using all variables within four different buffers (*e.g.*, 250 m, 500 m, 750 m and 1 km). The most parsimonious buffer radii were selected by ranking them using Akaike's Information Criterion corrected for small sample size (AIC_c), following Lesmerises et al. (2018). To minimize multicollinearity, we used the Variance Inflation Factor (VIF) as a guideline, setting an upper threshold of 5 (Oksanen et al., 2019). Industrial forest harvesting in this region follows a latitudinal gradient, with logging expanding from south to north over time (Bouchard & Pothier, 2011; Lafontaine et al., 2019). This spatial pattern created collinearity between latitude and road age, as northern roads were logged more recently than southern ones. Since road age was of greater ecological relevance to vegetation regrowth, we excluded latitude from our models. However, elevation and longitude were retained as covariates given their known influence on vegetation dynamics (Pausas & Austin, 2001). We assessed the significance of the final model, its axes, and the predictor variables using permutation tests in the vegan package in R (Oksanen et al., 2019; R Core Team, 2024; Table S1).

We used a multiple logistic regression to determine the factors influencing the occurrence of regenerating stems (*e.g.*, whether at least one stem >1.30 m was present). Each sampling site was assigned a binary value: 1 if at least one coniferous, deciduous/mixed, or alder stem was recorded, and 0 otherwise. Using the base R glm function (R Core Team, 2024), we compared estimated marginal means (EMMs) among vegetation categories with post hoc contrasts via the emmeans package (Lenth et al., 2021) across the 14 candidate models (see Table S2 for model compositions). Model validation included testing deviations from expected distributions (Kolmogorov-Smirnov test, outlier test), examining standardized residuals relative to predicted values (quantile regression, DHARMa package; Hartig, 2020), and comparing models to the null model (likelihood ratio test).

We used a negative binomial regression to assess the factors influencing the abundance of regenerating stems (*e.g.*, number of stems >1.30 m per plot). This approach is appropriate for count data, particularly when overdispersion is present (*e.g.*, when variance exceeds the mean; Bliss & Fisher, 1953). We examined coniferous, deciduous/mixed, and alder stems as response variables, considering road type (winter logging road or class 4 logging road) and age class (0–10, 11–20, 21–30, and 31–40 years old) as predictors. To ensure data integrity, we removed all lines containing missing values. Model structures mirrored those used in the logistic regression analysis (see model compositions detailed in Table S3, S4 and S5). Model validation followed the same steps as before, including standardized residual diagnostics (DHARMa package; Hartig, 2020), likelihood ratio tests against null models, and the Hosmer-Lemeshow test for goodness of fit (Hosmer & Lemeshow, 2000). For each model, we calculated marginal R^2 values representing the variance explained by fixed effects using the performance package in R (Lüdecke et al., 2021), which is suitable for generalized linear models including negative binomial and zero-inflated beta regressions.

Finally, we analyzed the lateral vegetation cover along roads using a zero-inflated beta mixed regression. This method is recommended for continuous data constrained within the [0,1] interval and characterized by an excess of zero values (Ferrari & Cribari Neto, 2004). We converted lateral cover percentages (0–100) into proportions (0–1) and addressed the limitations of beta regressions, which cannot handle exact 0 or 1 values, by following Damgaard & Irvine (2019) and transforming values of 1 into 0.999 and values of 0 into 0.001. Model validation involved assessing deviations from expected distributions (DHARMa package; Hartig, 2020), examining the beta distribution of residuals, evaluating model fit using R^2, and comparing models to the null model.

All continuous predictor variables were standardized prior to model fitting. This standardization allowed direct comparison of effect sizes among covariates within and across models, facilitating interpretation of their relative influence while retaining a hypothesis-driven, information-theoretic modeling framework. For all three analytical approaches (logistic regression, negative binomial regression, and zero-inflated beta regression), we built

14 candidate models by combining key covariates into three groups: road structure, topography, and landcover types. Some models also included interactions between covariates (see Table S6). Each model accounted for spatial autocorrelation using principal coordinates of neighbour matrices (PCNM; see Borcard & Legendre, 2002) and incorporated the appropriate buffer size, which varied by analysis. The most parsimonious buffer radius was retained based on AIC_c ranking. To assess whether compaction may have masked the effect of road type, we also ran an alternate set of candidate models for each vegetation analysis with the compaction variable excluded. These models were evaluated using the same procedures, and results are presented in the supplementary materials when $\Delta AIC_c < 2$, indicating they were competitive. The best-ranked models were selected using AIC_c.

Assessing the intensity of use by large mammals

We began the exploration of our camera-trap data with a hierarchical multiple ANOVA to compare the average photographic rate (*e.g.*, number of detections divided by the sampling effort in days) for each species between road types. We used mixed-effects Poisson regressions to describe the relationships between different ecological factors and their effects on road use by our four focal species. We modeled the detection counts as a function of several independent variables pooled in seven groups: road structure, vegetation regrowth, species-specific habitat, co-use (predator-prey-all, *e.g.*, variables accounting for the simultaneous use of roads by multiple species, including predators and prey), anthropogenic footprint, road-scale, and landcovers variables (see Tables S7 and S8).

We built 31 candidate models using the variables included in these seven groups, combining them and testing interactions between some variables (see Tables S7 and S8) to translate our hypotheses into statistical equations. The mixed effects structure of the model allowed us to account for random effects such as the road segment ID. Moreover, we included principal coordinates of neighbour matrices (PCNM: see Borcard and Legendre 2002) to account for spatial autocorrelation in our study design (PCNM were generated for each species), and the local density of the focal species as a covariate to control for the variation in species density within the study area. The effort (number of days a camera was active) was also included as a fixed effect. The candidate models were ranked using AIC_c for each studied species. Road type was included in all models regardless of selection by AIC_c, as it represents a central variable in our study design and hypotheses.

For each analysis and variable tested, we assessed the effect of spatial scale by incorporating four buffer sizes (250 m, 500 m, 750 m, and 1 km) around the relevant features to account for species-specific responses to proximity and selected the most parsimonious one using AIC_c. Some of the models tested had convergence and singularity issues and therefore had to be disregarded. We used the VIF to diagnose multicollinearity problems and verified model assumptions by testing deviations from the expected distribution and comparing standardized residuals against the predicted values (Hartig 2020). Overdispersion was checked in the count data to ensure the appropriateness of the Poisson distribution. Model fit was assessed using the AUC, and the final models were compared to null models. Statistical test assumptions were respected throughout. The mixed Poisson regression was implemented using the glmmTMB (Brooks et al., 2020) and DHARMa (Hartig, 2020) packages. All statistical analyses were performed using R 4.4.0 (R Core Team, 2024).

RESULTS

Drivers of vegetation regrowth on logging roads

The best-ranked model explaining plant occurrence in our 120 plots included variables related to road structure (Table 1.2) and had an excellent fit (AUC = 0.985). The type of road did not significantly influence the presence of vegetation in the middle of the roadway, but road age had a clear effect (Table 1.2). The probability of finding plants was significantly lower on roads aged 11–20 years compared to those aged 31–40 years, while roads aged 0–10 years were characterized by a complete absence of vegetation. Lastly, soil compaction had a negative effect on the occurrence of plants on logging roads, regardless of road type, as no significant interaction between compaction and road type was detected.

Table 1.2: Coefficients (β) and 95% confidence intervals (95% CI) of the different variables included in the best-ranked multiple logistic regression model used to explain plant occurrence (Occ. %) on logging roads in Northern Québec (Canada) in 2021. Covariates for which the 95% CI did not encompass zero were considered significant and are shown in bold font. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Covariates	Road structure	
	β	95% CI [Lower : Upper]
Intercept	**3.886**	**[1.270 : 7.559]**
Road – Class 4	–1.208	[–3.736 : 0.856]
Age 0-10	–	–
Age 11-20	**–6.967**	**[–12.015 : –3.834]**
Age 21-30	–2.149	[–5.156 : –0.299]
Soil compaction	**–1.423**	**[–3.195 : –0.243]**
PCNM2	0.951	[–0.625 : 2.819]
PCNM12	**-1.380**	[–3.176 : –0.223]
PCNM22	0.598	[–0.153 : 1.628]
Tjur's $R^{2:}$ 0.797		

The partial canonical correspondence analysis (pCCA) linking the number of stems (coniferous, deciduous, and alder) to environmental variables on 58 sampling plots where stems were present was significant (*P* = 0.005, Table 3), explaining 53.5% of the variation in stem abundance. The first two axes of the pCCA model explained 36.2% and 17.3% of the variance, respectively (Fig. 3). Four covariates had a significant effect on stem abundance: road age (*P* = 0.030), lateral cover (*P* = 0.010), soil compaction (*P* = 0.010), and vertical cover (*P* = 0.030). The proportion of 50–120-year-old deciduous/mixed stands and open lichen woodland were marginally significant (*P* = 0.100 and 0.065, respectively). The first axis (pCCA1, from left to right quadrants) was significant (*P* = 0.005) and had an eigenvalue of 0.343 (Table 1.3), indicating a strong increasing gradient of soil compaction and lateral cover and a decreasing gradient of road age. The second axis (pCCA2, from bottom to top quadrants) was not significant (*P* = 0.705, eigenvalue = 0.164) and was primarily related to changes in vertical cover. The ellipses delineating the 95% confidence intervals for the two types of logging roads were partially overlapping but remained distinct from each other (Figure 1.3). Coniferous stems (black spruce, balsam fir, tamarack, and jack pine) were more abundant on winter logging roads than on gravel roads and were negatively associated with soil compaction and positively associated with road age. Deciduous stems (willow, paper birch, and trembling aspen) were associated with an increase in vertical cover but showed no strong correlation with compaction. Alders were mostly found on class 4 logging roads, where both compaction and lateral cover were higher. Among the three groups, only coniferous stems showed a clear negative response to compaction.

Table 1.3: Eigenvalue, correlation, proportion explained, *F* and *p* values of the two axes of the partial canonical correspondence analysis (pCCA) relating the number of coniferous, deciduous, and alder stems to the logging road type and environmental variables surrounding the sampling sites, Northern Québec (Canada).

Axis	*Eigenvalue*	*Correlation*	*Proportion explained*	*F*	*p value*
CCA 1	0.343	0.826	0.362	36.56	0.005
CCA 2	0.164	0.607	0.173	17.45	0.705
Model				2.370	0.005

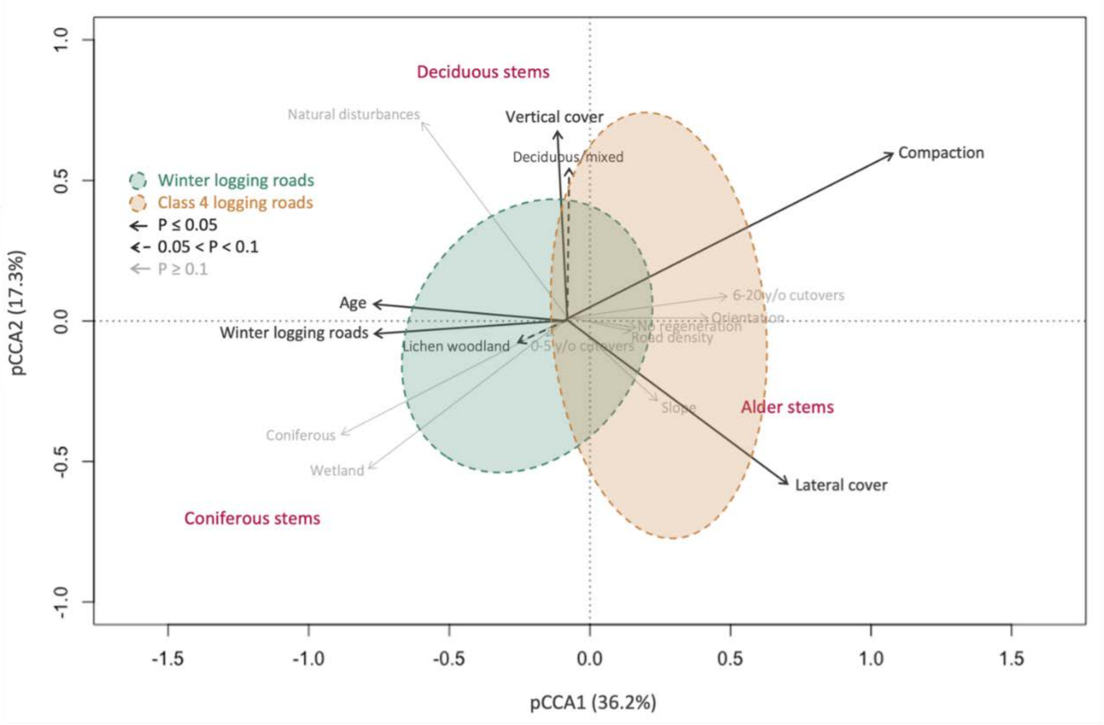

Figure 1.3: Biplot representing the first two axes of the partial canonical correspondence analysis (pCCA) exposing the relationship between the number of stems per vegetation group (coniferous, deciduous, and alder) in relation to the road type and to the different environmental variables used to evaluate the vegetation regeneration of logging roads in Northern Québec (Canada). Ellipses represent 95% confidence intervals of each logging road type (class 4 or winter).

The best-ranked negative binomial regression model explaining the number of coniferous stems >1.30 m in the middle of the roadway included only road structure variables (Table 1.4) and yielded a marginal R^2 of 0.992. In this model, soil compaction had a negative effect, while road type and road age were not significant predictors. For deciduous/mixed stems >1.30 m, the best-ranked model included road structure variables without compaction and had a marginal R^2 of 0.996. This model showed a significantly lower number of deciduous stems on class 4 roads compared to winter roads, and a marginal effect of road type was also detected (Table 1.4). Finally, the best model explaining the number of alder stems >1.30 m included both road structure and environmental variables without compaction. In this model, alder abundance was significantly higher on winter roads, and positively associated with wetland cover, natural disturbances, and the proportion of nearby old coniferous stands. Due to singularity in the random effect, the R² yielded a marginal value of 0.999, indicating strong explanatory power of the fixed effects in predicting alder stem abundance (Table 1.4).

Table 1.4: Coefficients (β) and 95% confidence intervals (95% CI) of the different variables included in the best-ranked negative binomial regression model used to explain the number of coniferous, deciduous and alder stems on logging roads in Northern Québec (Canada) in 2021. Covariates for which the 95% CI did not encompass zero were considered significant and are shown in bold font. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Response variable	Covariates	Model	
		Road structure	
Coniferous stems		β	95% CI [Lower : Upper]
	Intercept	0.501	[-0.436 : 1.438]
	Road – Class 4	–0.630	[–1.535 : 0.274]
	Age 0–10	–	–
	Age 11–20	–	–
	Age 21–30	–0.254	[–1.062 : 0.555]
	Soil compaction	**–0.770**	**[–1.376 : –0.164]**
	PCNM20	–1.932	[–6.094 : 2.229]
	PCNM26	**3.206**	**[1.743 : 4.669]**
Marginal $R^{2:}$ 0.992			
Deciduous stems		Road structure (without soil compaction)	
		β	95% CI [Lower : Upper]
	Intercept	**1.684**	**[0.873 : 2.494]**
	Road – Class 4	**–0.868**	**[–1.784 : 0.048]**
	Age 0–10	–	–
	Age 11–20	–	–
	Age 21–30	0.189	[–0.733 : 1.111]
	PCNM5	–0.055	[–0.6111 : 0.501]
	PCNM10	0.058	[–0.532 : 0.50]
Marginal R^2: 0.994			

Alder stems		Road Structure + Environment (without soil compaction)	
		β	95% CI [Lower : Upper]
	Intercept	**1.613**	**[0.455 : 2.771]**
	Road – Class 4	**–0.978**	**[–1.893 : –0.064]**
	Age 0–10	–	–
	Age 11–20	–0.966	[–3.197 : 1.265]
	Age 21–30	**1.076**	**[0.175 : 1.976]**
	0–5 years cutovers	0.356	[–0.837 : 1.549]
	Wetlands	**–0.820**	**[–1.368 : –0.273]**
	Open lichen woodlands	–0.546	[–1.211 : 0.119]
	50–120-year-old deciduous/mixed stands	–0.747	[–1.747 : 0.254]
	Natural disturbances	**–2.93**	**[–4.541 : –1.313]**
	90–120 years coniferous stands	**–1.309**	**[–2.159 : –0.459]**
	Road density	–0.226	[–0.607 : 0.155]
	PCNM5	–0.166	[–0.625 : 0.294]
Marginal R^2: 0.999			

The best-ranked model explaining lateral cover 1 m aboveground on logging roads included variables related to road structure (Table 1.5) and had a marginal R^2 of 0.745. This zero-inflated beta regression (*e.g.*, the logistic regression component of the analysis) indicated no significant effect of road type on the occurrence of lateral cover. However, road age categories significantly influenced the percentage of lateral cover, with roads aged 0–10, 11–20, and 21–30 years all differing significantly from roads aged 31–40 years (Table 1.5). Soil compaction within logging roads also had a negative effect on lateral cover.

Table 1.5: Coefficients (β) and 95% confidence intervals (95% CI) of the different variables included in the best-ranked model zero-inflated beta mixed regression used to explain the percentage of lateral cover on logging roads in Northern Québec (Canada) in 2021. Covariates for which the 95% CI did not encompass zero were considered significant and are shown in bold font. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Covariates	Road structure	
	β	95% CI [Lower : Upper]
Intercept	**0.874**	**[0.363 : 1.385]**
Road – Class 4	– 0.107	[–0.573 : 0.359]
Age 0–10	**–2.635**	**[–3.339 : –1.932]**
Age 11–20	**–2.200**	**[–2.855 : –1.536]**
Age 21–30	**–0.760**	**[–1.386 : –0.003]**
Soil compaction	**–0.267**	**[–0.530 : –0.003]**
PCNM5	**2.388**	**[0.069 : 4.707]**
PCNM12	**–2.802**	**[–5.457 : –0.147]**
PCNM20	0.442	[–2.264 : 3.149]
Marginal $R^{2:}$ 0.745		

Assessing the intensity of use by large mammals

Our camera-trap sampling effort on roads totaled 6,173 camera-days, during which we detected 247 black bear, 112 moose, 53 caribou, and 53 gray wolf events. We could not conduct statistical analyses for both caribou and wolves due to the very low sample size. On forest cameras, our effort reached 2,119 camera-days**,** detecting 33 black bear, 10 moose, and 2 caribou events, while wolves were not detected. When converted into photographic rates and compared between road types, we observed that moose had a similar average photographic rate on class 4 and winter roads, whereas black bears were detected more frequently on class 4 gravel roads (Figure 1.4). Caribou and wolves showed similar average photographic rates between road types, but the spread of detections appeared greater on class 4 roads, indicating higher variability in use (Figure 1.4). However, given the limited number of detections and absence of statistical testing, these patterns should be interpreted with caution.

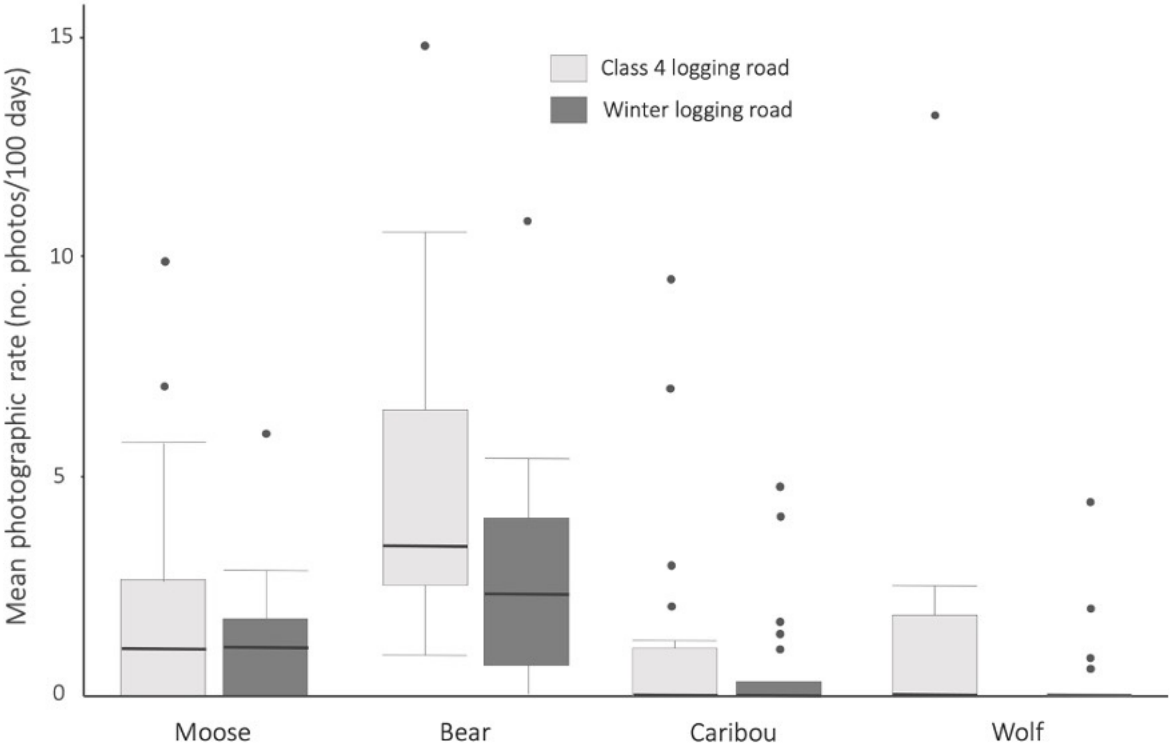

Figure 1.4: Mean photographic rate (no. photos/100 days, ± 95% CI) of four species of large mammals (moose, bear, caribou, and wolf) by road type (class 4 and winter), Northern Québec (Canada). Outliers are shown as individual data points, representing photographic rates significantly higher or lower than the typical range of observations.

This rather crude comparison can be refined using covariates referring to the surrounding landscape and interactions between large mammal species. For black bear intensity of use of roads, the best-ranked model included species-specific habitat and co-use (prey) covariates and showed moderate explanatory power (marginal $R^2 = 0.418$; conditional $R^2 = 0.564$). Both the photographic rate of moose on roads and the sampling effort were positively associated with black bear detections. In contrast, the presence of 90-120-year-old stands and wetlands near the road was negatively related to bear activity (Table 1.6). For moose, the best-ranked model included habitat and co-use (predator) covariates and had a higher explanatory power (marginal $R^2 = 0.567$; conditional $R^2 = 0.678$). Moose road use was positively associated with black bear photographic rates and wetland cover, while it was negatively linked to the proportion of regenerating stands (Table 1.6).

Table 1.6: Best-ranked mixed Poisson regression model explaining black bear and moose intensity of use of logging roads in Northern Québec (Canada) in 2021. Covariates are presented with their coefficient (β) and 95% confidence interval. Covariates for which the 95% CI did not encompass zero were considered significant and are shown in bold font. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Covariates	Black bear		Moose	
	β	95% CI [Lower : Upper]	β	95% CI [Lower : Upper]
Intercept	**0.50**	**[0.21 : 0.80]**	**-0.98**	**[-1.64 : -0.32]**
Road type	-0.22	[-0.67 : 0.22]	-0.07	[-0.81 : 0.67]
Moose use	**0.18**	**[0.05 : 0.32]**		
Caribou use	-0.08	[-0.29 : 0.13]		
Wolf use			0.19	[-0.12 : 0.49]
Bear use			**0.40**	**[0.16 : 0.64]**
Anthropogenic use			0.07	[-0.23 : 0.36]
Coniferous stand	**-0.23**	**[-0.44 : -0.01]**	-0.03	[-0.28 : 0.22]
Deciduous/mixed stand	0.06	[-0.11 : 0.22]	-0.28	[-0.71 : 0.16]
Regeneration	0.08	[-0.16 : 0.32]	**-0.95**	**[-1.58 : -0.33]**
No Regeneration	-0.03	[-0.23 : 0.17]	-0.40	[-0.83 : 0.04]
Wetland	**-0.20**	**[-0.42 : -0.02]**	**0.26**	**[0.03 : 0.50]**
Effort	**0.34**	**[0.06 : 0.62]**	0.43	[-0.16 : 1.02]
Local density	0.05	[-0.16 : 0.25]	-0.01	[-0.41 : 0.38]
PCNM5			-0.03	[-0.42 : 0.35]
PCNM13	**0.15**	**[0.02 : 0.28]**		
PCNM22			-0.10	[-0.41 : 0.20]
PCNM24	**-0.23**	**[-0.41 : -0.05]**		
Conditional R^2:		0.621		0.777
Marginal R^2:		0.467		0.641

DISCUSSION

We explored the differences in vegetation regrowth and large mammal use between two contrasting road types and found only partial support for our initial hypotheses and predictions. Contrary to expectations, winter roads did not consistently support greater vegetation regrowth (P1) nor higher caribou use (P5) than class 4 roads. Although road type was central to our study design, its influence appeared more limited than anticipated. Soil compaction emerged as a key factor influencing vegetation recovery, and large mammal responses varied according to biotic and abiotic road-level conditions.

Effect of road type on vegetation and wildlife responses

Our analyses revealed that road type alone was not a significant predictor of the presence or abundance of regenerating vegetation or lateral cover on logging roads (P1, P2, P3). While our partial canonical correspondence analysis (pCCA), which included only plots where vegetation was present, showed that coniferous stem abundance was higher on winter roads, this pattern was not corroborated by other models that used the full dataset (P1). Furthermore, our updated models indicate that the apparent effect of road type on vegetation may be mediated through structural factors such as soil compaction, which was significantly lower on winter roads (see Fig. S1). Since compaction limits the process of seedling establishment and, hence, influences species composition, its inclusion in previous models likely reduced the independent explanatory power of road type, which serves more as a proxy for these physical attributes (P4).

In contrast, our updated model for deciduous/mixed stems (excluding compaction and including road structure variables only) showed a significant negative effect of class 4 roads compared to winter roads and a marginal effect of road age (21–30 years), indicating that road type may influence the recovery of certain groups of plant species under specific conditions (P1, P2). For alders, the best-ranked model included both road structure and landcover variables (excluding compaction) and showed that alder stem abundance was significantly higher on winter logging roads and in areas with more wetlands, recent natural disturbances, and older coniferous stands (P1, P5). These results support the notion that alder establishment is shaped by both local road characteristics and broader habitat context, consistent with its role as a disturbance-tolerant, nitrogen-fixing pioneer species (Chapin et al., 1994; Finnegan et al., 2018; St-Pierre et al., 2021). The difficulty in computing R^2 due to singularity in the random effect suggests that part of the variability in alder responses may be driven by site-specific factors that were not explicitly measured in this study, such as fine-scale soil properties, microtopography, or local hydrological conditions. This result highlights the importance of unmeasured site-level heterogeneity in shaping vegetation responses along forest roads and suggests that alder establishment may be particularly sensitive to local environmental constraints beyond the variables included in our models. Our results also showed that lateral vegetation cover increased significantly with road age, particularly on roads over 30 years old (P2). This supports the view that structural complexity and vegetative shading require decades to re-establish after severe disturbance (Harper et al., 2005; Matlack, 1993).

Regarding large mammal activity, road type did not end up being a significant predictor of road use by black bears or moose in the best-ranked models. This indicates that road type may not directly shape large mammal responses beyond the effects mediated through vegetation structure or surrounding habitat context. Contrary to our initial prediction (P5), both species were frequently detected on both road types, indicating that early successional vegetation along winter and gravel roads likely offered comparable foraging opportunities.

Influence of time since disturbance on vegetation regrowth

Road age emerged as a key driver of vegetation regrowth, partially supporting our prediction (P2) that older roads would support more advanced vegetation recovery. Specifically, the likelihood of plant occurrence was significantly lower on 11–20-year-old roads compared to those aged 31–40 years, and no vegetation was recorded on roads under 10 years of age. These results are consistent with studies showing that vegetation recovery on disturbed sites is a lengthy process, influenced by time-dependent factors such as organic matter accumulation, soil stabilization, and seed bank development (Goss, 2022; SÉPAQ, 2014).

However, despite this overall age-related increase in vegetation occurrence, our results did not show a significant increase in coniferous stem abundance across road age categories in the regression models, indicating that conifer recruitment is constrained by additional factors. Coniferous species often require specific microsite conditions and adequate seed dispersal from nearby stands (St-Pierre et al., 2021), which may not be met uniformly across road segments. Our pCCA did indicate that conifer stem abundance was associated with older road age, suggesting that the influence of time since disturbance emerges through multivariate ecological gradients rather than as a strong univariate effect. This highlights the value of complementary multivariate approaches in capturing subtle, context-dependent patterns that may not be detected by regression models alone, particularly for late-successional species whose establishment depends on the joint effects of soil development, canopy structure, and propagule availability. This contrast may also reflect differences in the underlying data structure, as the pCCA was performed on a subset of plots where stems were present, thereby focusing on ecological gradients operating once recruitment has occurred, whereas the regression models included the full set of sampled plots. For deciduous and mixed stems, the best-ranked model showed a marginally positive effect of the 21–30-year-old age class, indicating that this age window may offer more favorable conditions for recruitment. This result aligns with a possible delayed recruitment effect, likely related to improving environmental conditions such as light availability and soil structure over time. These gains may still be moderated by dispersal limitations and interspecific competition. Alder stem abundance also peaked in the 21–30-year-old category and declined in the oldest class, supporting the idea that alder thrives during intermediate stages of succession. This pattern is consistent with alder’s ecological profile as a nitrogen-fixing pioneer species tolerant of disturbance (Chapin et al., 1994), particularly under conditions of elevated light and improving soil moisture—both of which are more likely as compaction eases over time (Haeussler & Coates, 1986).

Role of soil compaction in shaping vegetation recovery

As predicted (P3), soil compaction emerged as a key constraint on vegetation recovery, significantly reducing both plant occurrence and total stem abundance in our regression models. Compacted soils limit water infiltration, impede root penetration, and reduce microbial activity, ultimately hindering seedling establishment and growth (Ampoorter et al., 2011; Cambi et al., 2015; Naghdi et al., 2016; Mills et al., 2021). These effects were especially marked for coniferous species, which showed lower densities in compacted substrates.

Although alders are typically described as tolerant of disturbed and compacted environments (Finnegan et al., 2018; St-Pierre et al., 2021), the best-ranked model for alder stem abundance did not retain compaction as a significant predictor. This indicates that while compaction may influence alder recruitment under certain conditions, other factors—such as road age, road type, and surrounding environmental features—may play a more prominent role. In fact, our results showed that alder stem density was significantly higher on winter roads and on 21–30-year-old roadbeds, indicating that intermediate successional stages and less heavily built roads may offer more favorable conditions for this pioneer species.

Our pCCA nonetheless revealed a general negative relationship between compaction and total stem abundance, providing multivariate confirmation of its limiting effects across species. This was echoed in subsequent models, where compaction—not road type per se—was identified as the main driver of stem abundance. Coniferous stems were more abundant on winter roads, likely reflecting their lower levels of compaction. Lateral vegetation cover was likewise lower in more compacted roadbeds, reinforcing the idea that compaction influences both plant structure and ground-level microhabitats. Moreover, the negative relationship between soil compaction and lateral cover, revealed through our zero-inflated beta regression, highlights the capacity of mechanical disturbance to constrain plant growth not only in the roadbed center but also at the edges—a pattern consistent with research on compaction gradients in linear clearings (Gill et al., 2014; St-Pierre et al., 2021; Wirth et al., 2009).

Influence of landscape context on vegetation and wildlife responses

Among the surrounding landcover types examined, only a few showed significant or marginal effects on vegetation regeneration, indicating a modest but context-dependent influence of adjacent habitats on the roadbed. Our best-ranked model for alder stems >1.3 m retained several landcover variables, including wetlands, natural disturbances, and old coniferous stands, indicating that surrounding habitat composition can shape regeneration patterns—at least for disturbance-adapted species such as alder. These associations may reflect environmental gradients (*e.g.*, moisture, light availability) or propagule pressure linked to adjacent ecosystems.

In contrast, other landcover types like 50–120-year-old deciduous/mixed stands and open lichen woodlands showed only weak associations with vegetation abundance. This aligns with previous studies indicating that proximity to seed sources is not always sufficient to ensure regeneration, particularly in heavily disturbed substrates like compacted roads (Flinn & Vellend, 2005; Holl, 1999). In such cases, microsite limitations, such as poor soil structure, low nutrient availability, or surface sealing, can strongly inhibit germination and early survival, even when propagules are available from the surrounding forest matrix (MacDougall & Wilson, 2007; Turley et al., 2017).

Deciduous and mixed stands within our study area are composed primarily of species such as paper birch, trembling aspen, and willow, which can act as potential seed sources. However, their impact on roadbed regeneration may be constrained by dispersal limitations, competition, and edaphic conditions, such as nutrient availability or soil texture (de Souza et al., 2022). Seedling establishment is often strongly influenced by dispersal timing and microsite suitability, which may not always be met on compacted road surfaces (Harper et al., 2005; Moles & Westoby, 2004; Schupp, 1995).

In the case of open lichen woodlands—common in nutrient-poor, dry-mesic podzolic soils (Payette et al., 2000)—suboptimal conditions such as low soil moisture, sparse organic horizons, and high diurnal temperature fluctuations may further limit successful colonization of the roadbed (Gao et al., 2023). These ecosystems, dominated by *Cladonia* spp., are typically associated with low productivity and limited vegetative regrowth capacity.

Large mammal occurrence was also shaped by surrounding landcover. Moose detections were positively associated with wetlands and negatively associated with the proportion of regenerating stands, likely reflecting habitat selection patterns rather than simple avoidance of specific cover types. Wetlands and adjacent disturbed forest patches provide abundant, high-quality forage for moose (Dussault et al., 2005; Fisher & Burton, 2018; Peek, 2007), leading to a concentration of moose activity in landscapes where early successional browse and wetland resources intersect, and consequently increasing the use of nearby roads and other linear features for movement and foraging. Similarly, black bears were more frequently detected in areas with regenerating vegetation and wetlands—habitats that support fruiting shrubs, herbaceous forage, and dense cover essential for foraging and denning (Kautz et al., 2021; Lesmerises et al., 2013; Lewis et al., 2021). In such contexts, the combination of abundant adjacent food resources and the relatively low energetic cost of movement along linear features may increase the likelihood that bears use roads both as travel routes and opportunistic foraging areas. These findings support the hypothesis that both species use linear features as movement corridors and foraging routes (Dickie et al., 2017b; Mumma et al., 2018), as predicted in P4.

Notably, our models retained a positive effect of moose photographic rates on black bear detections, indicating a spatial association between the two species. While this may reflect shared habitat preferences, the possibility of behavioral interactions, such as opportunistic predation on moose calves by bears, cannot be ruled out (Ballard, 1994; Bastille-Rousseau et al., 2011). However, our data do not permit inference on the nature of this relationship, which could stem from indirect co-occurrence rather than direct interaction.

Our findings show that while adjacent landcover can influence vegetation and wildlife use in post-logging landscapes, its role in shaping roadside vegetation dynamics is often secondary to more immediate factors such as soil compaction, road age, and physical structure. Topographic variables such as slope, elevation, and aspect were not retained in the most parsimonious models explaining plant presence or stem abundance. This is consistent with findings from other studies in relatively homogeneous boreal landscapes, where topographic variation plays a minor role compared to more immediate drivers like soil compaction or hydrologic disturbance (Johnson & Miyanishi, 2008; Pausas & Austin, 2001). Additionally, the limited elevation range in our study area may have reduced potential topographic effects.

Study limitations

Our study faced a few limitations that merit consideration but do not undermine the ecological relevance of our findings. Most notably, we were unable to statistically assess road use by caribou and wolves due to their very low detection rates in our camera trap data. This outcome is consistent with their low population densities in the region: approximately 1–2 caribou per 100 km² and 4–6 wolves per 1,000 km² (MFFP, 2019, 2020). Although this limitation prevented us from formally testing prediction 5, the absence of caribou detections is itself consistent with their known avoidance of anthropogenic disturbances such as roads (Leblond et al., 2013; Lesmerises et al., 2013; Mumma et al., 2019; Rudolph et al., 2017). Moreover, the inclusion of black bears and moose—species that are widespread and ecologically influential in boreal systems—provides valuable insights into the potential facilitation of predator–prey interactions and the use of linear features by large mammals. Moose are a known vector of apparent competition, while black bears are significant calf predators in some caribou ranges (Bastille-Rousseau et al., 2011; Pinard et al., 2012). Their inclusion thus contributes meaningfully to understanding broader wildlife dynamics in disturbed landscapes.

Another methodological limitation relates to our use of motion-activated camera traps. While highly effective for detecting the presence and activity of species along specific road segments, this tool is not designed to assess habitat selection or broader spatial behavior. However, by capturing species use at the scale of individual road segments, camera traps provide a valuable “road-level” perspective that complements more spatially explicit approaches such as telemetry. They enable assessment of how animals respond to the physical and vegetative attributes of roads themselves—insights that are particularly relevant to land managers.

Finally, our study focuses on mid-term recovery (0–40 years post-abandonment), which captures early and intermediate successional stages but does not provide a full view of long-term recovery trajectories. Boreal forest regeneration is a slow process, and critical dynamics, such as conifer recruitment, understory closure, or the return of late-successional species, may only become apparent over longer timescales. Nonetheless, our results help illuminate key bottlenecks to recovery (*e.g.*, soil compaction, lack of lateral cover) and provide an empirical foundation for management decisions in the early decades following disturbance. Continued monitoring over longer timeframes will be necessary to assess whether these trajectories ultimately converge toward reference conditions or require restoration interventions.

In addition to these temporal constraints, some spatial and ecological variables relevant to vegetation recovery were not explicitly measured. While road structure influenced lateral cover, other unmeasured environmental conditions, such as localized soil moisture, nutrient availability, or disturbance history, may also contribute to heterogeneity in vegetation patterns. Previous studies have shown that roadside disturbances can alter adjacent microhabitats, affecting vegetation recruitment and cover in nearby zones (Harper et al., 2005). The presence of deep wheel ruts, commonly observed along forest roads, can locally intensify soil disturbance and compaction, thereby constraining vegetation establishment and recovery. Additionally, road width is known to play a role in vegetation dynamics, with wider roads often experiencing greater compaction and reduced vegetative cover along the margins (Laurance et al., 2009). Though our study did not explicitly measure road width, the observed link between compaction and cover may indirectly reflect this relationship.

Management implications

Our study offers several insights relevant to habitat management in boreal ecosystems, particularly in relation to abandoned logging roads. While we observed higher conifer stem abundance on winter roads in preliminary analyses, our final regression models revealed that this pattern was better explained by road structure—specifically soil compaction and road age—than by road type per se. For deciduous and alder stems, the best-supported models included either road type alone (for deciduous stems) or road type in combination with environmental variables (for alder stems), indicating a more complex and context-dependent influence of road characteristics.

This contrasts with the findings of Benoit-Pépin et al. (2024), who reported more pronounced differences between winter and gravel roads in a nearby region. However, their study did not measure road compaction (a more direct abiotic factor influencing the process of vegetation recovery) nor did it account for the age of linear infrastructures. As a result, their models may have attributed effects to road classification that were in fact driven by differences in substrate properties or time since disturbance.

In contrast, our results highlight that vegetation regeneration along forest roads is strongly shaped by a combination of soil compaction, road age, and surrounding habitat context, rather than by road type labels alone. The combination of these three factors provides a more in depth understanding of underlying ecological processes associated with vegetation regeneration on forest roads. Il also outines additional insight into the regeneration trajectories of winter roads, whose potential for passive recovery appears to vary substantially depending on edaphic conditions, time since abandonment and neighboring habitats. Whereas road age emerged as a useful predictor of vegetation regrowth, particularly for determining plant presence, it must be interpreted in conjunction with other variables such as soil compaction, landcover, and road construction history to accurately characterize recovery potential. Our study thus underscores the importance of integrating both abiotic properties and temporal factors when interpreting vegetation responses to linear disturbances. It also supports the need for broader cross-regional comparisons before formulating generalized management recommendations based solely on broad road categories.

The slow progression of passive restoration observed across our study area, even on winterroads, highlights the importance of considering active restoration strategies, especially on roads with persistent compaction or limited regeneration. Mechanical decompaction, targeted seeding, and reforestation have been shown to improve microsite conditions and promote plant recruitment (Lacerte et al., 2021; Muñoz-Rojas et al., 2016). Management strategies should prioritize decommissioning and active restoration of gravel logging roads, particularly in zones where natural recovery is hindered by soil constraints or a lack of propagule sources (Finnegan et al., 2018). Restoration strategies should also account for lateral vegetation, as promoting recovery along road edges may enhance site resilience and contribute to broader ecosystem recovery by improving microhabitat structure and buffering edge effects. The effectiveness and ecological impacts of such interventions should be assessed through long-term monitoring to avoid unintended consequences and to support adaptive management.

Our results also emphasize the importance of minimizing soil compaction at the initial phase of road construction in future harvested forest landscapes. This could facilitate soil rehabilitation and promote faster vegetation recovery in the active restoration of linear insfrastructures.

Finally, the frequent road use by black bears and moose, both of which were associated with local vegetation characteristics and broader habitat context, emphasizes the ecological role of linear features in shaping large mammal behavior. While our study did not assess habitat preference directly, the consistent presence of these species on both road types indicates that regenerating roads provide foraging opportunities and possibly travel corridors. Nonetheless, their distribution patterns highlight the importance of considering landscape configuration in ecological recovery and habitat management strategies. This has important implications for wildlife management and conservation, particularly in regions where predator–prey dynamics, road access, and human–wildlife interactions are of concern. Planning restoration efforts of harvesting roads should not only account for vegetation recovery but also for the potential role of abandoned roads in facilitating species movement and interactions across fragmented landscapes.

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1.3 SUPPORTING INFORMATION

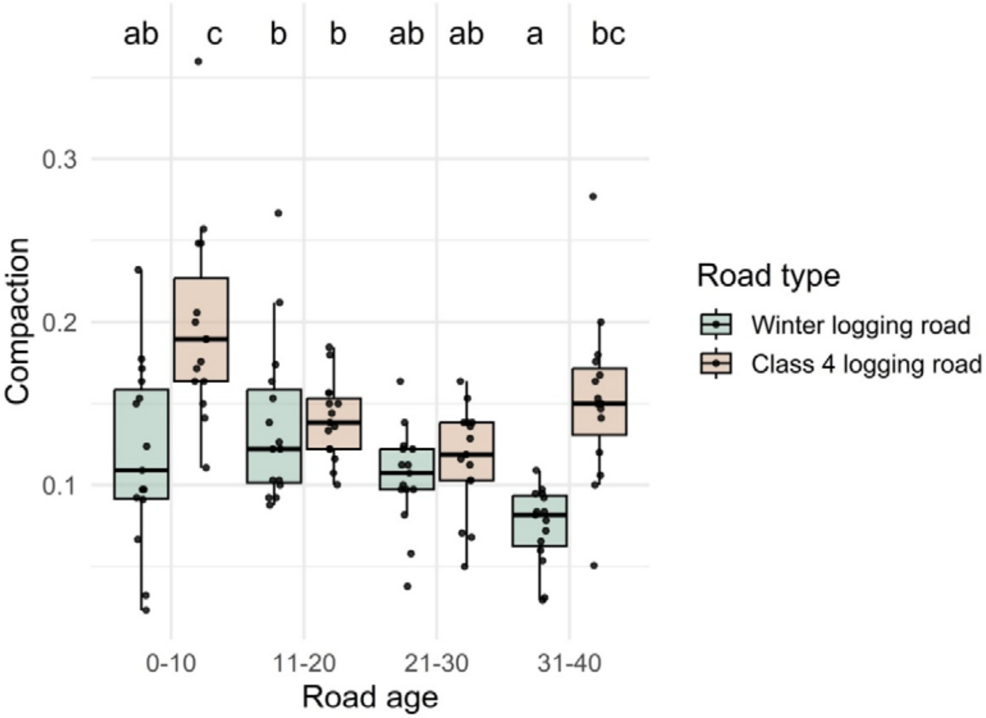

Figure S1: Soil compaction as a function of road age categories and road type. The interaction of road age and type on soil compaction was significant at the 95% level (ANOVA, F7,112 = 10.5, p<0.001). Post-hoc pairwise comparisons were conducted using estimated marginal means with Sidak correction for multiple testing (via the emmeans R package). Groups with different letters are statistically different from each other. These results indicate that winter roads are significantly less compacted than corresponding class 4 logging roads in the first 10 years after construction and after 30 years of use.

Table S1: Mean ($\bar{X}$), standard deviation (SD), range (min–max), chi-square (X2), *F* and *p* values of the covariates used in the partial canonical correspondence analysis (pCCA) used to link variations in the number of coniferous, deciduous and alder stems to the logging road type and environmental variables surrounding the sampling sites, Northern Québec (Canada).

Predictor variables	$\bar{X}$ ± *SD*	*Range*	X^2	*F*	*p value*
Age (years)	29.94 ± 5.26	20.00 – 36.00	0.016	3.28	0.035
Slope (°)	8.04 ± 7.29	1.00 – 29.00	0.006	0.60	0.530
Road compaction (cm)	0.11 ± 0.04	0.03 – 0.18	0.053	6.71	0.005
Road orientation (rad)	2.79 ± 1.76	0.24 – 6.28	0.012	1.89	0.135
Lateral cover (%)	66.57 ± 30.68	6.88 – 100.00	0.016	7.03	0.005
Vertical cover (%)	18.41 ± 27.86	0.00 – 92.50	0.045	2.50	0.085
Road density (%)	2.26 ± 0.72	1.28 – 3.84	0.009	1.38	0.250
0–5-year-old cuttings	2.05 ± 4.50	0.00 – 16.36	0.012	1.58	0.210
6–20-year-old cuttings	19.29 ± 27.40	0.00 – 97.13	0.006	0.59	0.585
50–120-year-old deciduous/mixed stands	0.49 ± 1.74	0.00 – 8.89	0.007	2.22	0.140
Wetlands	10.70 ± 11.13	0.00 – 38.09	0.001	0.15	0.865
Open lichen woodlands	0.21 ± 0.74	0.00 – 3.65	0.005	1.76	0.185
Open/no-regenerated stands	8.98 ± 11.97	0.00 – 54.84	0.003	0.53	0.590
Regenerated stands	28.80 ± 25.19	0.00 – 79.91	0.006	1.84	0.160
90–120-year-old coniferous stands	11.54 ± 10.53	0.00 – 48.56	0.014	1.66	0.375
Natural disturbances	4.65 ± 13.17	0.00 – 64.44	0.025	4.62	0.190

Table S2: Candidate models used to model plant occurrence (Occ. %) on logging roads using multiple logistic regression in caribou range in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Response variable	Model	Explanatory covariates	LL	k	ΔAIC_c	ω
Plant occurrence (Occ. %)	**1 – Road structure**	**Road type + Age category + Soil compaction + PCNM2 + PCNM12 + PCNM20**	**-19.6**	**9**	**0.00**	**0.574**
	1.1 – Model 1, without Compaction	Road type + Age category + PCNM2 + PCNM12 + PCNM20	-22.5	8	3.51	0.099
	2 – Road structure + Road type*Age category	Road type + Age category + Soil compaction + Road type*Age category + PCNM2 + PCNM12 + PCNM20	-19.1	12	6.50	0.022
	2.1 – Model 2, without Compaction	Road type + Age category + Road type*Age category + PCNM2 + PCNM12 + PCNM20	-22.0	11	9.69	0.005
	3 – Road structure +Road type*Compaction	Road type + Age category + Soil compaction + Road type*Soil compaction + PCNM2 + PCNM12 + PCNM20	-19.1	10	1.30	0.300
	4 – Topography	Orientation + Elevation + Slope + PCNM2 + PCNM12 + PCNM20	-63.6	7	83.40	0.000
	5 – Environment	0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM2 + PCNM12 + PCNM20	-26.6	13	23.87	0.000
	6 – Road structure + Topography	Road type + Soil compaction + Orientation + Elevation + Slope + PCNM2 + PCNM12 + PCNM20	-54.1	9	68.94	0.000
	6.1 – Model 6, without Compaction	Road type + Orientation + Elevation + Slope + PCNM2 + PCNM12 + PCNM20	-63.4	8	85.26	0.000
	7 – Road structure + Environment	Road type + Age category + Soil compaction + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM2 + PCNM12 + PCNM20	-22.4	14	17.90	0.000

	7.1 – Model 7, without Compaction	Road type + Age category + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM2 + PCNM12 + PCNM20	-29.6	17	29.78	0.000
	8 – Topography + Environement	Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + PCNM2 + PCNM12 + PCNM20	-29.9	16	35.47	0.000
	9 – Full Model	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0-5 yrso cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + PCNM2 + PCNM12 + PCNM20	-25.3	16	29.00	0.000
	9.1 – Model 9, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + PCNM2 + PCNM12 + PCNM20	-31.7	15	39.26	0.000
	10 – Full Model + Elevation*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + Elevation* Regeneration + PCNM2 + PCNM12 + PCNM20	-19.6	18	23.06	0.000
	10.1 – Model 10, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + Elevation* Regeneration + PCNM2 + PCNM12 + PCNM20	-26.9	17	34.93	0.000

Table S3: Candidate models used to model number of coniferous stems on logging roads using negative binomial regression in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation; PCNM values with absolute magnitude < 0.001 were truncated to zero prior to scaling to improve model stability.

Response variable	Model	Explanatory covariates	LL	k	ΔAIC_c	ω
Coniferous stems	**1 – Road structure**	**Road type + Age category + Soil compaction + PCNM20 + PCNM26**	**-97.2**	**10**	**0.00**	**0.458**
	1.1 – Model 1, without Compaction	Road type + Age category + PCNM20 + PCNM26	-100.2	9	3.77	0.070
	2 – Road structure + Road type*Age category	Road type + Age category + Soil compaction + Road type*Age category + PCNM20 + PCNM26	-97.1	13	7.28	0.012
	2.1 – Model 2, without Compaction	Road type + Age category + Road type*Age category + PCNM20 + PCNM26	-99.5	12	9.48	0.004
	3 – Road structure + Road type*Compaction	Road type + Age category + Soil compaction + Road type*Soil compaction + PCNM20 + PCNM26	-96.1	11	0.28	0.399
	4 – Topography	Orientation + Elevation + Slope + PCNM20 + PCNM26	-119.4	8	39.64	0.000
	5 – Environment	0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + PCNM20 + PCNM26	-112.8	14	41.29	0.000
	6 – Road structure + Topography	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + PCNM20 + PCNM26	-95.7	13	4.37	0.052
	6.1 – Model 6, without Compaction	Road type + Age category + Orientation + Elevation + Slope + PCNM20 + PCNM26	-99.3	8	9.15	0.005
	7 – Road structure + Environment	Road type + Age category + Soil compaction + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM20 + PCNM26	-93.3	19	15.91	0.000

	7.1 – Model 7, without Compaction	Road type + Age category + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM20 + PCNM26	-99.2	18	24.81	0.000
	8 – Topography + Environment	Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM20 + PCNM26	-111.0	17	45.72	0.000
	9 – Full Model	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM20 + PCNM26	-94.7	19	18.64	0.000
	9.1 – Model 9, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM20 + PCNM26	-107.4	18	41.27	0.000
	10 – Full Model + Elevation*Coniferous stand	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM20 + PCNM26	-96.0	19	21.25	0.000
	10.1 – Model 10, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM20 + PCNM26	-108.9	18	44.29	0.000
	11 – Full Model + Slope*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM20 + PCNM26	-92.3	20	16.78	0.000

	11.1 – Model 11, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0-5 yrs cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM20 + PCNM26	-107.0	19	43.3	0.000
	12 – Full Model + Orientation*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM20 + PCNM26	-94.5	20	21.22	0.000
	12.1 – Model 12, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM20 + PCNM26	-106.9	19	43.10	0.000
	13 – Full Model + Compaction*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Compaction *Regeneration + PCNM20 + PCNM26	-93.8	20	19.64	0.000
	14 – Complete + Road type* Coniferous stand	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Road type + Coniferous stand + PCNM20 + PCNM26	-93.3	20	18.66	0.000
	14.1 – Model 14, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Road type + Coniferous stand + PCNM20 + PCNM26	-106.5	19	42.27	0.000

Table S4: Candidate models used to model number of deciduous stems on logging roads using negative binomial regression in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Response variable	Model	Explanatory covariates	LL	k	ΔAIC_c	ω
Deciduous stems	1 – Road structure	Road type + Age category + Soil compaction + PCNM5 + PCNM12 + PCNM20	-142.8	10	2.34	0.172
	1.1 – Model 1, without Compaction	**Road type + Age category + PCNM5 + PCNM12 + PCNM20**	**-142.8**	**9**	**0.00**	**0.556**
	2 – Road structure + Road type*Age category	Road type + Age category + Soil compaction + Road type*Age category + PCNM5 + PCNM12 + PCNM20	-142.8	13	9.75	0.004
	2.1 – Model 2, without Compaction	Road type + Age category + Road type*Age category + PCNM5 + PCNM12 + PCNM20	-142.8	12	7.26	0.015
	3 – Road structure + Road type*Compaction	Road type + Age category + Soil compaction + Road type*Soil compaction + PCNM5 + PCNM12 + PCNM20	-141.8	11	2.70	0.144
	4 – Topography	Orientation + Elevation + Slope + PCNM5 + PCNM12 + PCNM20	-161.6	8	35.23	0.000
	5 – Environment	0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	-152.8	14	32.36	0.000
	6 – Road structure + Topography	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + PCNM5 + PCNM12 + PCNM20	-141.0	13	6.04	0.027
	6.1 – Model 6, without Compaction	Road type + Age category + Orientation + Elevation + Slope + PCNM5 + PCNM12 + PCNM20	-141.1	12	3.83	0.082

	7 – Road structure + Environment	Road type + Soil compaction + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	-146.6	16	25.09	0.00
	7.1 – Model 7, without Compaction	Road type + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	-150.1	15	29.53	0.000
	8 – Topography + Environment	Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	-149.4	17	33.45	0.000
	9 – Full Model	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	-142.9	19	25.98	0.000
	9.1 – Model 9, without Compaction	Road type + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	-145.5	18	28.42	0.000
	10 – Full Model + Elevation* Coniferous stand	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM5 + PCNM12 + PCNM20	-141.9	20	26.89	0.000

10.1 – Model 10, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM5 + PCNM12 + PCNM20	-144.2	19	28.83	0.000
11 – Full Model + Slope*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM5 + PCNM12 + PCNM20	-141.0	20	25.08	0.000
11.1 – Model 11, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM5 + PCNM12 + PCNM20	-143.4	19	26.98	0.000
12 – Full Model + Orientation*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM5 + PCNM12 + PCNM20	-142.8	20	28.79	0.000
12.1 – Model 12, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM5 + PCNM12 + PCNM20	-145.4	19	31.06	0.000

	13 – Full Model + Compaction*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Compaction *Regeneration + PCNM5 + PCNM12 + PCNM20	-142.4	20	27.93	0.000
	14 – Full Model + Road type*Coniferous stand	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Road type + Coniferous stand + PCNM5 + PCNM12 + PCNM20	-142.6	20	28.46	0.000
	14.1 – Model 14, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Coniferous stand + Road density + Road type + Coniferous stand + PCNM5 + PCNM12 + PCNM20	-145.2	19	30.70	0.000

Table S5: Candidate models used to model number of alder stems on logging roads using negative binomial regression in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Response variable	Model	Explanatory covariates	LL	k	ΔAIC_c	ω
Alder stems	1 – Road structure	Road type + Age category + Soil compaction + PCNM5	-211.9	9	13.41	0.001
	1.1 – Model 1, without Compaction	Road type + Age category + PCNM5	-211.9	8	11.08	0.003
	2 – Road structure + Road type*Age category	Road type + Age category + Soil compaction + Road type*Age category + PCNM5	-210.2	12	17.30	0.000
	2.1 – Model 2, without Compaction	Road type + Soil compaction + Road type*Age category + PCNM5	-210.3	11	15.11	0.000
	3 – Road structure + Road type*Compaction	Road type + Age category + Soil compaction + Road type*Soil compaction + PCNM5	-211.3	10	14.58	0.001
	4 – Topography	Orientation + Elevation + Slope + PCNM5	-229.0	7	42.98	0.000
	5 – Environment	0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5	-210.0	13	19.58	0.000
	6 – Road structure + Topography	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + PCNM5	-211.2	12	19.43	0.000
	6.1 – Model 6, without Compaction	Road type + Age category + Orientation + Elevation + Slope + PCNM5	-211.2	11	16.98	0.000
	7 – Road structure + Environment	Road type + Soil compaction + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Coniferous stand + Road density + PCNM5	-197.7	16	2.66	0.208

	7.1 – Model 7, without Compaction	**Road type + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Coniferous stand + Road density + PCNM5**	**-197.7**	**15**	**0.00**	**0.786**
	8 – Topography + Environment	Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Coniferous stand + Road density + PCNM5	-209.4	16	26.17	0.000
	9 – Full Model	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5	-203.8	17	17.76	0.000
	9.1 – Model 9, without Compaction	Road type + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5	-206.8	16	20.93	0.000
	10 – Full Model + Elevation*Coniferous stand	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM5	-203.8	18	20.37	0.000
	10.1 – Model 10, without Compaction	Road type + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM5	-206.8	17	23.61	0.000

	11 – Full Model + Slope*Regeneration	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM5	-203.7	18	20.21	0.000
	11.1 – Model 11, without Compaction	Road type + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM5	-206.7	17	23.41	0.000
	12 – Full Model + Orientation*Regeneration	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM5	-203.8	18	20.46	0.000
	12.1 – Model 12, without Compaction	Road type + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM5	-206.7	17	23.47	0.000
	13 – Full Model + Compaction*Regeneration	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Compaction *Regeneration + PCNM5	-203.8	18	20.47	0.000

	14 – Full Model + Road type* Coniferous stand	Road type + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Road type + PCNM5	-203.4	18	19.56	0.000
	14.1 – Model 14, without Compaction	Road type + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + Natural disturbances + Regeneration + Coniferous stand + Road density + Road type + PCNM5	-206.8	17	23.64	0.000

Table S6: Candidate models used to explain the variation in lateral cover on logging roads using zero-inflated beta mixed regressions in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Response variable	Model	Explanatory covariates	LL	k	ΔAIC_c	ω
Lateral cover (%)	**1 – Road structure**	**Road type + Age category + Soil compaction + PCNM5 + PCNM12 + PCNM20**	**176.5**	**11**	**0.00**	**0.363**
	1.1 – Model 1, without Compaction	Road type + Age category + PCNM5 + PCNM12 + PCNM20	174.5	10	1.57	0.166
	2 – Road Structure + Road type*Age category	Road type + Age category + Soil compaction + Road type*Age category + PCNM5 + PCNM12 + PCNM20	178.7	14	3.21	0.073
	2.1 – Model 2, without Compaction	Road type + Age category + Road type*Age category + PCNM5 + PCNM12 + PCNM20	177.1	13	3.97	0.050
	3 – Road Structure + Road type*Compaction	Road type + Age category + Soil compaction + Road type*Soil compaction + PCNM5 + PCNM12 + PCNM20	176.9	12	1.74	0.152
	4 – Topography	Orientation + Elevation + Slope + PCNM5 + PCNM12 + PCNM20	148.8	9	50.68	0.000
	5 – Environment	0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	168.5	15	26.21	0.000
	6 – Road Structure + Topography	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + PCNM5 + PCNM12 + PCNM20	178.9	14	2.87	0.086

	6.1 – Model 6, without Compaction	Road type + Age category + Orientation + Elevation + Slope + PCNM5 + PCNM12 + PCNM20	177.8	13	2.39	0.110
	7 – Road Structure + Environment	Road type + Age category + Soil compaction + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	176.0	17	16.63	0.000
	7.1 – Model 7, without Compaction	Road type + Age category + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	169.8	16	26.36	0.000
	8 – Topography + Environment	Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	170.7	18	29.97	0.000
	9 – Full Model	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	177.0	20	23.04	0.000
	9.1 – Model 9, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + PCNM5 + PCNM12 + PCNM20	177.0	19	29.71	0.000

10 – Full Model + Elevation*Coniferous stand	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM5 + PCNM12 + PCNM20_	177.1	21	25.34	0.000
10.1 – Model 10, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Elevation*Coniferous stand + PCNM5 + PCNM12 + PCNM20_	173.1	20	30.91	0.000
11 – Full Model + Slope*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM5 + PCNM12 + PCNM20	177.9	21	24.24	0.000
11.1 – Model 11, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Slope*Regeneration + PCNM5 + PCNM12 + PCNM20	173.6	20	30.00	0.000
12 – Full Model + Orientation*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM5 + PCNM12 + PCNM20	177.2	21	25.67	0.000

	12.1 – Model 12, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Orientation*Regeneration + PCNM5 + PCNM12 + PCNM20	172.3	20	32.52	0.000
	13 – Complete + Compaction*Regeneration	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Compaction *Regeneration + PCNM5 + PCNM12 + PCNM20	179.7	21	20.60	0.000
	14 – Complete + Road type* Coniferous stand	Road type + Age category + Soil compaction + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Road type + Coniferous stand + PCNM5 + PCNM12 + PCNM20	179.4	20	18.29	0.000
	14.1 – Model 14, without Compaction	Road type + Age category + Orientation + Elevation + Slope + 0–5-y-o cutovers + Wetland + Lichen Woodland + Deciduous/mixed stand + No regeneration + Natural disturbances + Regeneration + Coniferous stand + Road density + Road type + Coniferous stand + PCNM5 + PCNM12 + PCNM20	176.0	19	22.32	0.000

Table S7: Candidate models used to assess the intensity of use of logging roads by black bears using mixed Poisson regression in caribou range in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age variable, and 'winter logging road' was used as the reference category for the road type variable. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Model	Explanatory covariates	LL	k	ΔAIC_c	ω
1 – Road structure	Road type + Age category + Slope + Lateral cover + PCNM15 + Effort + Local density + (1|ID_ROAD)	-214.4	12	12.59	0.001
2 – Road structure + Anthropogenic use	Road type + Age category + Slope + Lateral cover + Anthropogenic + PCNM15 + Effort + Local density + (1|ID_ROAD)	-214.2	13	14.65	0.000
3 – Road structure + Age category*Lateral cover	Road type + Slope + Age category*Lateral cover + PCNM15 + Effort + Local density + (1|ID_ROAD)	-207.3	12	6.03	0.020
4 – Road structure + Anthropogenic footprint	Road type + Age category + Slope + Lateral cover + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM15 + Effort + Local density + (1|ID_ROAD)	-209.7	15	3.12	0.087
5 – Road structure + Habitat	Road type + Age category + Slope + Lateral cover + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + (1|ID_ROAD)	-208.3	17	5.35	0.029
6 – Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-219.0	7	14.33	0.000
7 – Vegetation regrowth + Road structure	Coniferous stems + Deciduous stems + Alder stems + Road type + Age category + Slope + Lateral cover + PCNM15 + Effort + Local density + (1|ID_ROAD)	-214.3	15	20.08	0.000
8 – Co-use – Predators	Wolf use + Anthropogenic use + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-218.3	6	10.65	0.002
9 – Co-use – Prey	Moose use + Caribou use + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-214.6	6	3.22	0.083

10 – Co-use – All	Wolf use + Anthropogenic use + Moose use + Caribou use + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-213.2	8	5.14	0.032
11 –Co-use – Predator + Road structure	Road type + Age category + Slope + Lateral cover + Wolf use + Human use + PCNM15 + Effort + Local density + (1|ID_ROAD)	-212.6	14	14.00	0.000
12 – Co-use – Prey+ Road structure	Road type + Age category + Slope + Lateral cover + Moose use + Caribou use + PCNM15 + Effort + Local density + (1|ID_ROAD)	-209.5	14	7.76	0.009
13 – Co-use – Predator + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Wolf use + Human use + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-217.9	9	17.09	0.000
14 –Co-use – Prey + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Moose use + Caribou use + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-214.2	9	9.65	0.003
15 – Road-scale	Road type + Age category + Slope + Lateral cover + Coniferous stems + Deciduous stems + Alder stems + Wolf use + Anthropogenic use + Moose use + Caribou use + PCNM15 + Effort + Local density + (1|ID_ROAD)	-207.3	19	16.99	0.000
16 – Road-scale + Age category*Lateral cover	Road type + Age category*Lateral cover + Slope + Coniferous stems + Deciduous stems + Alder stems + Wolf use + Anthropogenic use + Moose use + Caribou use + PCNM15 + Effort + Local density + (1|ID_ROAD)	-203.8	19	9.97	0.003
17 – Anthropogenic footprint	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-214.0	7	4.39	0.046
18 – Anthropogenic footprint + Road type*6–20-y-o cutovers	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers*Road type + PCNM15 + Effort + Local density + (1|ID_ROAD)	-212.6	9	6.39	0.017

19 – Anthropogenic footprint + Age category*6–20-y-o cutovers	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers*Age category + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-209.0	11	9.32	0.004
20 – Anthropogenic footprint + Road density* Anthropogenic use	Road density*Anthropogenic use + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-213.1	8	7.52	0.010
21 – Anthropogenic footprint + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-213.8	10	11.24	0.002
22 – Anthropogenic footprint + Co-use – Predator	Wolf use + Anthropogenic use + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-213.2	9	7.57	0.009
23 – Anthropogenic footprint + Co-use – Prey	Moose use + Caribou use + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-210.8	9	2.91	0.097
24 – Habitat	Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-211.2	9	3.64	0.067
25 – Habitat + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-213.8	12	11.24	0.002
26 – Habitat + Co-use – Predator	Wolf use + Anthropogenic use + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-210.4	11	7.08	0.012
27 – Habitat + Co-use – Prey	**Moose use + Caribou use + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)**	**-206.9**	**11**	**0.00**	**0.415**

28 – Surrounding environment-scale	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-208.2	12	5.18	0.031
29 – Surrounding environment-scale + Road density*6-20 y-o cutovers	Road density*6–20-y-o cutovers + 0–5-y-o cutovers + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-208.1	12	7.65	0.009
30 – Surrounding environment-scale + Road density* Regeneration	Road density*Regeneration + 0–5-y-o cutovers + 6–20-y-o cutovers + Deciduous/mixed stand + No regeneration + Coniferous stand + Wetland + PCNM15 + Effort + Local density + Road type + (1|ID_ROAD)	-208.0	12	7.32	0.011
31 – Full model	Road type + Age category + Slope + Lateral cover + Coniferous stems + Deciduous stems + Alder stems + Wolf use + Anthropogenic use + Moose use + Caribou use + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + Coniferous stems + Deciduous stems + PCNM15 + Effort + Local density + (1|ID_ROAD)	-200.3	29	18.09	0.000

Table S8: Candidate models used to assess the intensity of use of logging roads by moose using mixed Poisson regression in caribou range in Northern Québec (Canada) in 2021. The model's name, covariates included, log likelihood (LL), number of predictors (k), distance to the best-ranked model (ΔAIC_c) and weight (ω) are shown. 'Age 31–40' was used as the reference category for the age covariate, and 'winter logging road' was used as the reference category for the road type covariatev. PCNM: vector of principal coordinates of neighbour matrices controlling for spatial autocorrelation.

Model	Explanatory covariates	LL	k	ΔAIC_c	ω
1 – Road structure	Road type + Age category + Slope + Lateral cover + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-138.6	13	15.31	0.000
2 – Road structure + Anthropogenic use	Road type + Age category + Slope + Lateral cover + Anthropogenic use + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-138.6	14	17.83	0.000
3 – Road structure + Age category* Lateral cover	Road type + Slope + Age category*Lateral cover + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-136.7	13	19.18	0.000
4 – Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-144.5	8	19.72	0.000
5 – Vegetation regrowth + Road structure	Coniferous stems + Deciduous stems + Alder stems + Road type + Age category + Slope + Lateral cover + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-136.2	16	18.12	0.000
6 – Co-use – Predators	Wolf use + Bear use + Anthropogenic use + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-140.0	8	10.76	0.004
7 – Co-use – Caribou	Caribou use + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-146.0	6	18.15	0.000
8 – Co-use – All	Wolf use + Anthropogenic use + Bear use + Caribou use + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-139.4	9	11.94	0.002
9 –Co-use – Predator + Road structure	Road type + Age category + Slope + Lateral cover + Wolf use + Bear use + Anthropogenic use + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-132.8	16	11.43	0.003
10 –Co-use – Caribou + Road structure	Road type + Age category + Slope + Lateral cover + Caribou use + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-138.6	14	17.81	0.000

11 – Co-use – Predator + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Wolf use + Bear use + Anthropogenic use + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-138.2	11	14.40	0.001
12 –Co-use – Caribou + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Caribou use + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-144.3	9	21.73	0.000
13 – Road-scale	Road type + Age category + Slope + Lateral cover + Coniferous stems + Deciduous stems + Alder stems + Wolf use + Anthropogenic use + Bear use + Caribou use + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-130.5	20	17.68	0.000
14 – Road-scale + Age category* Lateral cover	Road type + Age category*Lateral cover + Slope + Coniferous stems + Deciduous stems + Alder stems + Wolf use + Anthropogenic use + Bear use + Caribou use + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-132.0	20	20.66	0.000
15 – Anthropogenic footprint	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-139.4	8	9.64	0.007
16 – Anthropogenic footprint + Road type*6–20 y-o cutovers	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers*Road type + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-138.2	10	12.07	0.002
17 – Anthropogenic footprint + Age category*6⁻20-y-o cutovers	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers*Age category + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-132.0	12	9.81	0.006
18 – Anthropogenic footprint + Road density*Regeneration	Road density*Regeneration + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-136.0	9	7.60	0.018
19 –Anthropogenic footprint + Road structure	Road type + Age category + Slope + Lateral cover + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-138.9	16	15.83	0.000

20 – Anthropogenic footprint + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-137.2	11	12.50	0.002
21 – Anthropogenic footprint + Co-use – Predator	Wolf use + Bear use + Anthropogenic use + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-134.6	11	7.27	0.022
22 – Anthropogenic footprint + Co-use – Caribou	Caribou use + Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-139.4	9	11.91	0.002
23 – Habitat	Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-135.7	10	7.04	0.024
24 – Habitat + Road structure	Road type + Age category + Slope + Lateral cover + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11+ Effort + Local density + Road type + (1|ID_ROAD)	-135.3	10	13.66	0.001
25 – Habitat + Vegetation regrowth	Coniferous stems + Deciduous stems + Alder stems + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-133.8	13	10.78	0.004
26 – Habitat + Co-use – Predator	**Wolf use + Bear use + Anthropogenic use + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11 + Effort + Road type + Local density + (1|ID_ROAD)**	**-128.4**	**12**	**0.00**	**0.819**
27 – Habitat + Co-use – Caribou	Caribou use + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-135.7	11	9.50	0.007
28 – Surrounding environment-scale	Road density + 0–5-y-o cutovers + 6–20-y-o cutovers + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11 + Effort + Local density + Road type + (1|ID_ROAD)	-131.7	13	6.61	0.030

29 – Surrounding environment-scale + Road density*6–20-y-o cutovers	Road density*6–20-y-o cutovers + 0–5-y-o cutovers + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11+ Effort + Local density + Road type + (1|ID_ROAD)	-131.2	13	8.19	0.014
30 – Surrounding environment-scale + Road density* Regeneration	Road density*Regeneration + 0–5-y-o cutovers + 6–20-y-o cutovers + Deciduous/mixed stand + No regeneration + Coniferous stand + Wetland + PCNM4 + PCNM11+ Effort + Local density + Road type + (1|ID_ROAD)	-130.4	13	6.48	0.032
31 – Full model	Road type + Age category + Slope + Lateral cover + Coniferous stems + Deciduous stems + Alder stems + Wolf use + Anthropogenic use + Bear use + Caribou use + Road density + 0–5-y-o cutovers + 6–20-yrs cutovers + Deciduous/mixed stand + No regeneration + Regeneration + Coniferous stand + Wetland + Coniferous stems + Deciduous stems + PCNM4 + PCNM11 + Effort + Local density + (1|ID_ROAD)	-122.2	30	16.19	0.000

CONCLUSION GÉNÉRALE

RETOUR SUR LE CONTEXTE ET LES OBJECTIFS DE L'ÉTUDE

Les structures linéaires, telles que les chemins forestiers, sont largement reconnues pour altérer les dynamiques fauniques dans les écosystèmes boréaux dont les habitats sont fragmentés par les activités industrielles extractives (Pasher et al., 2013 ; Trombulak & Frissell, 2000). En facilitant les déplacements des prédateurs, en augmentant l'accessibilité aux proies alternatives et en réduisant la connectivité fonctionnelle entre les habitats fragmentés, ces corridors anthropiques contribuent au déclin de plusieurs espèces sensibles à la perturbation, dont les populations locales du caribou boréal (*Rangifer tarandus caribou*) (COSEWIC 2014, 2016 ; Dyer et al., 2001). Dans ce contexte, documenter la capacité de certaines de ces infrastructures à se régénérer naturellement - constitue un enjeu important pour évaluer leur empreinte écologique à long terme sur le territoire. Cette régénération passive des structures linéaires, lorsqu'elle est suffisante pour atténuer leur attractivité pour les espèces problématiques et favoriser un retour graduel vers des conditions forestières naturelles, peut représenter une voie complémentaire aux stratégies de restauration active (par intervention humaine), particulièrement dans les paysages fortement perturbés, dans un contexte où les ressources d'intervention demeurent limitées.

Bien que plusieurs travaux aient évalué l'efficacité de différentes méthodes de restauration dans le contexte des lignes sismiques de l'Ouest canadien (*i.e.* Dickie et al., 2017 ; Filicetti & Nielsen, 2018 ; Keim et al., 2019), et que l'impact écologique des chemins forestiers ait fait l'objet de nombreuses recherches dans l'est du pays (*i.e.* Lacerte et al., 2021,2022 ; Lee & Boutin 2006 ; St-Pierre et al., 2021, 2022), les chemins d'hiver demeurent largement sous-étudiés. Ces chemins, construits sur sol gelé et dépourvus de gravier, pourraient pourtant représenter des infrastructures à impact écologique moindre. Dans une étude récente menée au sud de Val-d'Or, Benoit-Pépin et al. (2024) ont montré que les chemins forestiers gravelés étaient significativement plus utilisés par les prédateurs que les chemins d'hiver et les milieux riverains, un patron principalement expliqué par des différences marquées de couvert latéral entre les différents types de structures linéaires et la forêt adjacente. Toutefois, les milieux riverains, bien qu'ils constituent des habitats linéaires, ne correspondent pas à des chemins forestiers au sens strict, mais plutôt à des voies de déplacement naturelles pour la faune. Cette distinction pourrait en partie influencer les relations observées entre les types de structures linéaires et leur utilisation par les espèces, notamment en raison de fonctions écologiques et de contraintes de déplacement différentes. Sur cette base, les auteurs recommandent de prioriser la restauration active des chemins gravelés, qui concentrent l'essentiel de l'activité des espèces associées au déclin du caribou. Les travaux de Lacerte et al. (2021, 2022) ont par ailleurs montré que la densité du couvert forestier adjacent influençait la régénération végétale et son utilisation par la grande faune, tandis que St-Pierre et al. (2021) ont souligné l'effet limitant de la compaction des sols sur la régénération naturelle. Dans une seconde étude, St-Pierre et al. (2022) ont montré que les caractéristiques physiques des chemins conditionnaient leur utilisation par les mammifères : les loups (*Canis lupus*) privilégiaient les segments à faible couvert latéral, alors que les orignaux (*Alces alces americana*) fréquentaient davantage les chemins âgés de 11 à 20 ans.

Dans ce contexte, l'objectif principal de mon étude était de comparer la régénération de la végétation et l'utilisation des grands mammifères de ces deux types de chemins forestiers en forêt boréale – les chemins d'hiver et les chemins de classe 4 – en expliquant la variabilité de ces deux phénomènes. Ultimement, documenter la dynamiques de végétalisation des chemins et leur utilisation par la grande faune s'avérait essentiel pour d'évaluer l'impact des infrastructures linéaires sur les relations entre le caribou boréal et ses prédateurs et compétiteurs, pour orienter les stratégies de gestion du réseau routier et les approches de restauration adaptées aux enjeux de conservation du caribou. Celle-ci peut mener, selon les contextes, soit vers une reprise forestière caractérisée par un retour progressif à une composition et une structure similaire à celles des peuplements adjacents, notamment par l'établissement de conifères tolérants à l'ombre comme l'épinette noire, soit vers un état alternatif dominé par des espèces pionnières ou arbustives (*i.e.* saules, aulnes), reflétant une régénération plus lente ou bloquée dans les milieux perturbés. Quant aux déterminants des niveaux d'utilisation de ces chemins par une communauté de grands mammifères – incluant le caribou boréal, le loup gris, l'ours noir (*Ursus americanus*) et l'orignal – mon étude s'inscrivait dans une perspective appliquée de conservation du caribou boréal, une espèce menacée, dont l'aire de répartition au nord du Québec est fortement fragmentée par les réseaux linéaires issus de l'aménagement forestier.

RÉSUMÉ DES PRINCIPAUX RÉSULTATS

L'analyse comparative des chemins d'hiver et des chemins de classe 4 a mis en évidence des différences dans la composition et la structure de la végétation, révélant que la régénération s'implantant passivement sur les assises différait selon le type de chemin, le temps écoulé depuis leur construction, ainsi que les caractéristiques du sol et des milieux adjacents. Les chemins d'hiver, caractérisés dans cette étude par une structure non excavée et une absence de traitement mécanique du sol, ont montré une reprise végétale plus marquée que les chemins de classe 4, généralement associés à une structure excavée et compactée. Toutefois, cette différence ne semble pas avoir été suffisante pour entraîner une diminution mesurable de la fréquentation par l'ours noir et l'orignal. Ce constat appuie l'idée que les conditions initiales de construction influencent fortement le potentiel de régénération passive, notamment en ce qui concerne la colonisation par les conifères (St-Pierre et al., 2021). L'âge du chemin était également positivement associé à la probabilité de présence ainsi qu'à l'abondance de tiges végétale, deux variables plus élevées sur les segments les plus âgés. Toutefois, la régénération observée ne correspondait pas toujours à une trajectoire de reboisement vers un couvert forestier fermé ou représentatif des pessières matures. Plusieurs tronçons étaient dominés par des espèces herbacées ou arbustives, comme l'aulne, indiquant que la régénération passive peut mener, dans certains contextes, à des états alternatifs peu structurés ou à succession lente (Jasinski & Payette, 2005 ; St-Pierre et al., 2021). Par ailleurs, la composition floristique variait en fonction de plusieurs facteurs environnementaux. L'abondance des tiges était influencée par la compaction du sol, la couverture latérale et verticale, ainsi que par la proportion de peuplements résineux et de milieux ouverts à proximité. Ces résultats soulignent l'importance de l'environnement local dans les trajectoires de régénération passive, et confirment que le retour spontané à un couvert forestier structuré dépend à la fois des propriétés physiques du chemin et de son contexte paysager (Dabros et al., 2018 ; Finnegan et al., 2018).

En ce qui concerne l'utilisation des chemins par la grande faune, mes analyses ont révélé des résultats contrastés entre les espèces. En raison du faible nombre de photographies de caribous et de loups, il n'a pas été possible de tirer de conclusions statistiquement robustes pour ces deux espèces reconnues pour se trouver à très faible densité en forêt boréale (*i.e.* 1-2 caribou/100 km^2; 4-6 loups/1000 km^2). Toutefois, les analyses menées pour l'orignal et l'ours noir ont permis d'identifier certains facteurs influençant leur présence sur les chemins forestiers. Chez l'ours, l'utilisation des chemins était positivement associée à celle de l'orignal, tandis qu'elle était négativement liée à la proportion de peuplements anciens (90–120 ans) à proximité. L'utilisation des chemins par l'orignal était quant à elle positivement corrélée à la proportion de milieux humides et à la présence de l'ours, et négativement associée à la proportion de coupes forestières en régénération. Aucune différence marquée dans l'intensité d'utilisation des chemins selon leur type (chemin d'hiver vs classe 4) n'a été observée pour l'orignal. Bien que l'ours ait été détecté plus fréquemment sur les chemins de classe 4, cette différence repose sur une comparaison des taux photographiques bruts et n'a pas fait l'objet d'analyses statistiques permettant de contrôler les sources de variation confondante. De plus, les modèles explicatifs suggèrent que l'utilisation des chemins par l'ours est davantage liée au contexte paysager et à la co-utilisation par l'orignal qu'au type de chemin en soi.. Mes résultats indiquent que, pour ces deux espèces, les caractéristiques des milieux adjacents et les interactions interspécifiques jouent un rôle plus déterminant que le type de chemin dans l'utilisation de ces corridors forestiers.

PORTÉE ET IMPLICATIONS DE L'ÉTUDE

Contributions théoriques

Mon étude contribue à améliorer notre compréhension des processus de régénération passive dans les forêts boréales perturbées, en mettant en évidence que certains chemins abandonnés développent une couverture végétale dominée par des espèces arbustives ou pionnières, peu représentatives des pessières matures environnantes (sensu Jasinski & Payette, 2005). Ces résultats tendent à indiquer que, sous certaines conditions, la régénération passive peut conduire à des états végétatifs alternatifs stables, appuyés par la littérature sur les effets de seuils écologiques liés au sol, à la lumière ou au manque de sources de propagules (Boulangeat et al., 2012 ; Suding et al., 2004).

Une contribution théorique supplémentaire de cette étude tient à la prise en compte explicite de l'âge de création des infrastructures linéaires, une dimension rarement intégrée dans les travaux portant sur la régénération des chemins forestiers. Nos résultats montrent que le temps écoulé depuis la perturbation constitue un facteur déterminant des trajectoires de reprise de la végétation, indépendamment du type de chemin. La dimension temporelle permet de mieux comprendre la variabilité observée dans les conditions de compaction des sols et de régénération végétale, en particulier sur les chemins d'hiver. Ellemet en évidence le mécanisme du temps requis pour la régénération du couvert végétalqui n'est pas abordé dans es études reposant uniquement sur des comparaisons spatiales, telles que celle de Benoît-Pépin et al. (2024).

Par ailleurs, l'intégration d'une composante faunique permet de discuter du découplage entre structure floristique et fonctionnalité écologique, un enjeu théorique central dans l'évaluation des habitats restaurés. Certains chemins faiblement utilisés par la grande faune présentent une végétation dense, tandis que d'autres plus fréquentés sont dominés par des espèces plus attractives. Ce constat souligne la nécessité d'évaluer les infrastructures abandonnées à la fois sur leur structure végétale et sur leur rôle fonctionnel dans les réseaux écologiques.

Contributions appliquées

Dans le contexte québécois de la conservation du caribou boréal, mes résultats permettent de moduler les interventions de restauration de l’habitat en fonction des caractéristiques des chemins et du paysage. Les chemins d’hiver, qui présentent une compaction plus faible, une meilleure reprise végétale et une utilisation généralement faible par la grande faune, pourraient ne pas nécessiter d’interventions intensives à court terme. À l’inverse, les chemins de classe 4, plus compactés, plus lents à se régénérer et souvent associés à une végétation plus attractive pour les proies alternatives, pourraient représenter des cibles prioritaires pour d’éventuelles mesures de restauration active. Mes résultats peuvent ainsi aider à cibler les segments où les efforts de restauration seraient les plus efficaces, en tenant compte à la fois du type de chemin, du stade de régénération végétale et de l’attractivité pour la faune. Ils appuient une approche de priorisation adaptative, dans laquelle le type de perturbation, les caractéristiques des milieux adjacents (*i.e.* proportion de peuplements résineux, présence de milieux humides ou de coupes récentes) et les patrons d’utilisation faunique doivent être pris en compte conjointement pour orienter les efforts de restauration (Bauduin et al., 2020 ; Ray, 2014). En complément, des mesures de gestion ciblée, comme le contrôle spatial ou démographique des populations de prédateurs, pourraient aussi être envisagées dans les secteurs les plus sensibles, particulièrement lorsque la restauration végétale seule ne suffit pas à réduire les risques de prédation indirecte (COSEWIC 2014 ; Serrouya et al., 2019).

Enfin, mon étude souligne que la restauration des chemins forestiers ne peut être pensée uniquement en fonction de leurs conditions locales, mais doit aussi intégrer les caractéristiques des milieux adjacents, que j’ai identifiées comme des facteurs déterminants de la régénération végétale et de l’utilisation par la faune, afin de mieux contribuer à la restauration de l’habitat du caribou. Les résultats obtenus dans mon aire d’étude contribuent à orienter les futures stratégies d’aménagement, que ce soit pour réduire les risques de prédation, reconnecter des noyaux de population isolés ou restaurer la résilience des paysages boréaux à long terme.

LIMITES DE L’ÉTUDE

Comme toute étude de terrain menée en contexte écologique hétérogène, mon projet comporte certaines limites méthodologiques et temporelles qu’il convient de considérer dans l’interprétation des résultats. D’abord, bien que les chemins étudiés aient été abandonnés depuis plusieurs années, les données récoltées dans cette étude sur la végétation et la faune des chemins ne reflètent qu’un instantané à court terme de leur dynamique. La succession végétale en milieu boréal peut s’étendre sur plusieurs décennies (Harper et al., 2001), et les trajectoires observées aujourd’hui ne garantissent pas l’atteinte d’un état stable ou souhaité à long terme (Johnstone et al., 2010). Il en va de même pour la réponse de la faune, qui peut évoluer dans le temps en fonction des ressources disponibles (Dussault et al., 2005), des capacités de déplacement des espèces (Bastille-Rousseau et al., 2011) ou de l’évolution du couvert végétal à mesure que les milieux se referment ou changent de composition (Fisher & Burton, 2018). Par ailleurs, la faible fréquence de détection du loup et du caribou dans les données issues des caméras automatisées limite la capacité d’évaluer adéquatement l’utilisation des chemins par ces espèces clé à la compréhension de la dynamique de prédation du caribou (James & Stuart-Smith 2000). Par ailleurs, bien que les caméras utilisées dans cette étude (Spypoint Force Dark V2.05.20) soient dotées d’une vitesse de déclenchement rapide, la détectabilité des espèces demeure influencée par leur comportement ainsi que par leur mode et leur vitesse de déplacement. En effet, certains travaux ont montré que la probabilité de détection par caméras automatisées varie selon la vitesse de déplacement, la trajectoire empruntée et l’angle d’approche des individus, ce qui peut entraîner une sous-détection de certaines espèces ou de certains types de déplacements (Gagnon-Labrosse, 2022). Chez le loup, notamment, des déplacements rapides ou perpendiculaires au champ de détection peuvent réduire la probabilité de détection par les caméras automatisées (Rowcliffe et al., 2008). En revanche, pour le caribou forestier, le faible nombre de détections observé est vraisemblablement attribuable à sa très faible densité sur le territoire d’étude plutôt qu’à des limitations du dispositif de détection. Dans ce contexte, bien que des inventaires complémentaires fondés sur des indices de présence auraient pu permettre de mieux qualifier l’intensité d’utilisation du territoire par le caribou, les résultats obtenus à partir des pièges photographiques demeurent cohérents avec l’état de la population dans la région étudiée.

Enfin, une part importante de la variation observée dans la composition végétale (environ 71 %) et dans l’utilisation des chemins par les mammifères demeure inexpliquée par les modèles statistiques utilisés. Cela indique que d’autres variables, telles que les propriétés physico-chimiques du sol, l’historique précis des perturbations, la disponibilité des propagules ou la dispersion naturelle des espèces, pourraient exercer une influence significative (Hart & Chen, 2008 ; Kremer et al., 2012 ; Pinno & Hawkes, 2015). L’intégration de ces variables dans de futurs protocoles d’échantillonnage permettrait d’approfondir la compréhension des mécanismes sous-jacents à la régénération passive dans les corridors forestiers.

PERSPECTIVES

Les résultats de mon étude ouvrent plusieurs avenues de recherche et d’application pour approfondir la compréhension des dynamiques de régénération passive des chemins dans les forêts boréales aménagées, et pour renforcer l’efficacité des interventions futures en matière de conservation. Un premier axe de recherche porte sur le suivi à long terme des trajectoires de régénération des chemins forestiers. En forêt boréale, la maturation des peuplements et la stabilisation des assemblages végétaux peuvent s’échelonner sur plusieurs décennies (Greene et al., 1999 ; Harper et al., 2001), et la réponse de la faune peut suivre des patrons temporels différés, influencés par l’évolution de la structure du couvert, la disponibilité en ressources ou les conditions microclimatiques (Gauthier et al., 2014). Un suivi prolongé permettrait ainsi de déterminer si les trajectoires initiales observées – notamment sur les chemins d’hiver – convergent vers des états représentatifs des pessières matures, ou s’orientent vers des états alternatifs moins favorables pour le caribou.

Un deuxième axe concerne l’intégration de variables édaphiques et microclimatiques dans les modèles explicatifs de la régénération végétale. Des travaux antérieurs ont montré que la compaction du sol, le drainage, la texture ou encore la teneur en matière organique peuvent influencer la capacité des espèces forestières à recoloniser les corridors anthropiques (Dabros et al., 2018 ; St-Pierre et al., 2021). L'inclusion de ces variables permettrait d'améliorer la prévision du potentiel de restauration passive, en identifiant plus précisément les segments de chemin propices à une reprise naturelle rapide ou, au contraire, susceptibles de stagner sans intervention. Ces éléments soulignent que le potentiel de régénération passive des chemins d'hiver dépend non seulement de leur caractère temporaire en tant qu'infrastructures d'exploitation forestière – utilisées sur une courte période et généralement abandonnées après les opérations – mais aussi de la qualité de leur mise en œuvre.

Au-delà des variables édaphiques classiquement considérées, des conditions de gel insuffisantes, des dégels hâtifs ou une circulation prolongée de la machinerie peuvent entraîner un orniérage marqué du sol, en particulier des horizons superficiels, susceptible de compromettre durablement la régénération de la végétation sur les assises des chemins (Ampoorter et al., 2012; Kremers & Boosten, 2018). Dans un contexte de changements climatiques caractérisé par des hivers plus courts et un gel du sol plus tardif, le maintien de bonnes pratiques d'aménagement des chemins d'hiver pourrait donc devenir un enjeu central déterminant leur contribution potentielle à la restauration des paysages boréaux, notamment dans les territoires d'intérêt pour le caribou forestier. Des travaux ont par ailleurs intégré des mesures explicites d'orniérage, notamment à l'aide de classes fondées sur la profondeur et la sévérité des ornières, permettant de relier plus directement les conditions d'exploitation aux perturbations du sol et aux processus de récupération des chemins forestiers (Curran et al., 2007; Page-Dumroese et al., 2009).

Sur le plan faunique, bien que mon étude ait montré une faible utilisation des chemins à court terme, il serait pertinent d'évaluer, dans de futurs projets, l'effet d'aménagements physiques supplémentaires — tels que le dépôt systématique de gros débris ligneux ou l'implantation d'andains transversaux — afin de restreindre les capacités de déplacement des prédateurs et des proies alternatives le long des chemins (Dickie et al., 2017 ; Keim et al., 2019). Ces éléments, combinés à l'analyse de la végétation et du paysage, pourraient contribuer à mieux comprendre les conditions favorables à une réappropriation sélective des corridors par les espèces cibles. Une autre perspective importante consiste à intégrer mes résultats à des analyses de connectivité fonctionnelle à l'échelle du paysage. En évaluant quels types de segments de chemin contribuent ou nuisent à la connectivité entre habitats favorables au caribou (Bauduin et al., 2020 ; McRae et al., 2008), il serait possible de cibler plus efficacement les zones prioritaires pour la restauration, qu'elle soit passive ou assistée.

Enfin, bien que mon étude ait été menée dans un contexte écologique spécifique du Québec boréal, mes résultats peuvent servir de point de départ à des projets réalisés dans d'autres régions caractérisées par des régimes de perturbation ou des compositions forestières différentes (Environnement Canada 2011 ; Finnegan et al., 2018). Une telle approche contribuerait à consolider des recommandations de restauration adaptables à l'ensemble de l'aire de répartition du caribou boréal. Mon étude met d'ailleurs en lumière le potentiel de la régénération passive comme outil de restauration dans certaines conditions, notamment sur les chemins d'hiver faiblement compactés. En montrant que l'utilisation des chemins forestiers par la grande faune n'est pas strictement déterminée par le type de chemin, mes résultats permettent de nuancer la perception d'une empreinte écologique linéaire universellement persistante. Ils soulignent plutôt l'importance d'adopter une approche analytique détaillée qui tienne compte des caractéristiques intrinsèques des chemins, des milieux adjacents et de leur contexte paysager et de la dynamique temporelle de infrastructures linéaires afin de guider nos interventions de restauration qui soient fondées sur des données écologiques et adaptées à la diversité des milieux boréaux. Dans un contexte où l'empreinte humaine sur les écosystèmes continue de s'intensifier (Foley et al., 2005 ; Maxwell et al., 2016), comprendre les conditions dans lesquelles la nature peut se restaurer par elle-même devient un levier essentiel pour concilier conservation de la biodiversité et aménagement du territoire.

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