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RÉSUMÉ

La perte et la fragmentation de l'habitat représentent un danger majeur pour les populations animales. En effet, les perturbations anthropiques toujours plus importantes affectent les habitats et peuvent même menacer la survie de certaines espèces sensibles. L'aménagement forestier, bien que créant une perte d'habitat temporaire, s'étend sur d'immenses superficies et ouvre le couvert forestier à des niveaux jamais expérimentés en forêt boréale de l'est de l'Amérique du Nord. La martre d'Amérique (*Martes americana*), une espèce reconnue comme sensible à la perte et à la fragmentation de l'habitat, peut servir d'espèce indicatrice et guider l'aménagement forestier pour d'autres espèces ayant des besoins d'habitat similaires. L'objectif de cette thèse était donc d'évaluer les effets de la perte (proportion des coupes) et de la fragmentation de l'habitat (dispersion des coupes et quantité des lisières créées) sur la martre, et ce, à différentes échelles. Le projet a par ailleurs été réalisé sur le territoire de la Forêt Modèle Crie de Waswanipi où deux patrons de dispersion des coupes (agglomérées vs. dispersées) ont été utilisés, le second étant considéré comme une mesure d'atténuation des impacts sur l'utilisation traditionnelle du territoire par les Cris de Waswanipi.

Le 1^er^ chapitre avait pour objectif d'évaluer les effets de la perte et de la fragmentation de l'habitat par l'aménagement forestier sur la martre, tant à l'échelle du paysage (100-300 km²) qu'à l'échelle des domaines vitaux (5-10 km²). Nous avons montré que l'abondance des martres diminuait de moitié dans les paysages aménagés par rapport à un paysage naturel, et ce, malgré des quantités de forêts résiduelles similaires. Par contre, à plus fine échelle, l'occupation des sites, l'abondance et la condition physique des martres n'était que faiblement affectées par la perte d'habitat. Le patron de dispersion des coupes ne semblait pas non plus affecter l'abondance, cependant la condition physique des martres était plus faible dans les paysages de coupes agglomérées. Nous concluons que les populations de martres sont certes sensibles à la récolte forestière, mais peu ou pas à la configuration et à la quantité de coupes dans des paysages où ~50% de forêts résiduelles sont maintenues. Ces résultats suggèrent une plus grande tolérance de la martre à la perte et à la fragmentation de l'habitat en forêt boréale que dans d'autres régions plus méridionales.

Dans le 2^e^ chapitre, nous voulions vérifier si cette plus grande tolérance s'observait aussi au niveau de l'utilisation de l'habitat des martres à l'intérieur de leurs domaines vitaux. L'objectif était de documenter la réponse individuelle des martres à la perte et à la fragmentation de l'habitat ainsi que leur sélection d'habitat hivernal. Là encore, la proportion de coupes et la dispersion de celles-ci n'influençaient pas la superficie des domaines vitaux. Par contre, les domaines vitaux diminuaient avec la quantité de forêts mixtes, un type forestier pourtant rare dans le paysage. L'importance des forêts mixtes pour la martre s'est confirmée à la fois dans sa sélection des sites utilisés à l'intérieur des domaines vitaux et dans le positionnement des domaines vitaux et des zones d'activité dans le paysage. Parallèlement, les coupes forestières récentes étaient les habitats les moins utilisés à l'échelle des domaines vitaux, et les milieux ouverts (dont les coupes) et perturbés (routes, lisières) étaient évités dans les zones d'activité. Ainsi, bien que la martre n'utilise pas les parterres de coupes, elle semble bien les tolérer à l'intérieur de leurs territoires.

Le 3^{e} chapitre avait comme objectif d'évaluer l'effet de la création de lisières forêt/coupe sur les mouvements de la martre. Selon la dispersion des coupes, les lisières créées sont soit simples (coupes dispersées) soit doubles (coupes agglomérées). Les lisières simples représentaient des barrières aux déplacements des martres, qui évitaient de s'aventurer dans les coupes. De plus, les déplacements étaient plus parallèles à la lisière et plus linéaires dans la zone 0-100 m que plus profondément dans la forêt. Par ailleurs, la distribution des pistes de martres par rapport à la lisière avait tendance à suivre la distribution des pistes de proies qui étaient plus nombreuses près de la lisière. Les corridors forestiers (lisières doubles) créaient quant à eux un effet de canalisation des mouvements et concentraient les pistes de martres.

Finalement, le 4^{e} chapitre avait pour objectif de développer une vision commune (Cris-scientifiques) de la martre en forêt boréale aménagée, en intégrant les connaissances traditionnelles des trappeurs Cris concernant la martre, de même que leurs perceptions des effets des coupes forestières sur cette espèce, avec les résultats des 3 précédents chapitres. La martre représente à la fois une source de revenus et un symbole du mode de vie traditionnel pour les Cris. Les connaissances des trappeurs Cris se sont révélées très concordantes avec nos résultats aussi bien sur l'écologie de la martre que sur les effets des coupes sur ses populations. Nous pensons que le fait que les deux sources d'information (traditionnelles et scientifiques) proviennent du même territoire d'étude a largement contribué à ce résultat. Les trappeurs Cris se sont par ailleurs montrés nettement plus favorables à l'utilisation des coupes dispersées comparativement aux coupes agglomérées dans leurs territoires de chasse familiaux. Cette vision commune de la martre pourrait contribuer à améliorer la compréhension mutuelle entre les Cris et les aménagistes forestiers et faciliter la gestion collaborative de la forêt dans la région.

Bien que la martre soit plus tolérante à la perte et à la fragmentation de l'habitat en forêt boréale, il est impossible de juger si les niveaux de rétention d'habitat imposés par la législation actuelle seront suffisants à long terme pour maintenir cette espèce. Malgré le fait que les coupes dispersées n'apportent qu'un faible gain écologique pour la martre, il est clair que ce type de coupe est socialement plus acceptable pour les trappeurs Cris. Alors que seule la première passe des coupes dispersées a eu lieu pour le moment, il est temps de planifier comment se fera la 2^{e} récolte. Nous recommandons la rétention de corridors pour conserver une connectivité pour les espèces fauniques et la protection des peuplements mixtes qui sont fortement sélectionnés par les martres dans cette région.

Mots-clés

Martre d'Amérique, perte d'habitat, fragmentation, paysage, sélection d'habitat, lisières forêt/coupe, aménagement forestier, autochtone, cogestion

ABSTRACT

Habitat loss and fragmentation represent a major threat for wildlife populations. Ongoing human disturbances affect habitat and could consequently threaten sensitive species survival. Even though forest management creates temporal habitat loss, it covers vast areas and opens up the forest matrix to a degree not previously experienced in eastern boreal forest of North America. American marten (*Martes americana*) is reported as sensitive to habitat loss and fragmentation. Marten could thus be used as focal species that could guide forest management for species having similar habitat requirements. The objective of this thesis was to evaluate effects of habitat loss (proportion of clearcuts) and fragmentation (clearcut dispersion and amount of created edges) on marten at different spatial scales. The study area was located in the Waswanipi Cree Model Forest, where two clearcut dispersion patterns (clustered vs. dispersed) were used, the second being considered to mitigate negative impacts of forest harvesting on traditional activities of indigenous peoples.

The first chapter examined effects of habitat loss and fragmentation by forest harvesting on marten at the landscape (100-300 km²) and the home range (5-10 km²) scales. We showed that marten abundance was two-fold higher in unharvested than in harvested landscapes, despite a similar proportion of mature forests. However, at a finer scale, marten site occupancy, abundance and body condition was only weakly affected by habitat loss. Clearcut dispersion pattern did not influence marten abundance, whereas body condition was reduced in clustered-cut landscapes. We conclude that marten populations are sensitive to habitat loss but not to the configuration and the amount of clearcuts when ~50% residual forests are maintained in the surrounding landscape. These results suggest that marten is more tolerant to habitat loss and fragmentation in coniferous boreal forest that in southern regions.

In the second chapter, we documented individual response to habitat loss and fragmentation as well as habitat selection within winter home ranges, in order to verify if this greater tolerance was also found at smaller scale. Again, proportion and dispersion of clearcuts did not influence marten home range size. However, home ranges decreased with the proportion of mixedwood forests, which are rare in the landscape. This importance of mixedwood forests for marten was confirmed by habitat selection at the home range scale as well as by home range and activity zone locations within the landscape. In parallel, recent clearcuts was the least used habitat at the home range scale and open (including clearcuts) and disturbed (roads and edges) areas were avoided in activity zones. Even though martens did not use clearcuts, they tolerated such areas within their home ranges.

Chapter 3 evaluated effects of creating forest/clearcut edges on marten movements. Single edges were created in dispersed-cut landscapes whereas multiple edges were created in clustered-cut landscapes. Single edges represented barrier for marten movements, as they avoided clearcuts. Moreover, movements were more parallel to the edge and more linear in the 0-100 m zone from the edge than deeper in the forest. However, marten distribution with regards to edge tended to follow prey distribution, being more abundant near edges. Forest corridors (multiple edges) were highly used by marten (concentration effect) and movements were more linear in such habitats (canalization effect).

Chapter 4 aimed to develop a common vision (Cree-scientists) on marten in managed boreal forest, in integrating Cree trappers' traditional knowledge and perceptions on the impacts of forest management on marten with results from the three previous chapters. Marten represented both an income source and a symbol of the traditional Cree way of life. Cree trappers' knowledge was highly concordant with our results on marten ecology as well as on the impacts of timber harvesting on this species. We suggest that this concordance was highly due to the fact that both sources of knowledge (traditional and scientific) came from the same study area. Cree trappers were more favorable to the use of dispersed clearcuts in their familial hunting territories. Such a common vision on marten could contribute to improve mutual understanding between Crees and forest managers and facilitate forest co-management in the region.

Although marten is more tolerant to habitat loss and fragmentation in the boreal forest, we cannot judge if retention levels imposed by the current law would be sufficient to maintain this species. Despite the fact that dispersed clearcuts provided only a weak ecological advantage for martens, this type of clearcut is more socially acceptable for Cree trappers. While the first pass is currently ongoing, it is time to plan how the second cut will be done. We recommend retention of corridors to conserve connectivity for wildlife species and protection of mixed wood stands which are highly selected by martens in the region.

Key words

American marten, habitat loss, fragmentation, landscape, habitat selection, forest/clearcut edges, forest management, First Nation, co-management

INTRODUCTION GÉNÉRALE

0.1. Fragmentation de l'habitat

0.1.1. Fragmentation et perte d'habitat

La fragmentation de l'habitat est l'un des sujets en écologie les plus étudiés depuis les 20 dernières années (Fig. 0.1), du fait de la progression constante des perturbations humaines imposées aux paysages. Le terme fragmentation a souvent été confondant dans la littérature scientifique, référant à la fois à la perte d'habitat (réduction de la quantité d'habitat) et à la configuration spatiale des habitats résiduels (superficie et isolement des parcelles d'habitat). Pourtant, de nombreux auteurs considèrent la perte d'habitat et la fragmentation comme deux phénomènes distincts (Fahrig 1997, Haila 1999, McGarigal et Cushman 2002, Lindenmayer et Fischer 2007). Il est maintenant largement accepté que la fragmentation réfère strictement aux changements dans la configuration spatiale d'un habitat donné (rupture de la continuité), dont les effets ne deviennent importants qu'au delà d'un certain niveau de perte d'habitat (Villard 2002).

Le concept de fragmentation de l'habitat origine de la théorie de la « biogéographie insulaire » (MacArthur et Wilson 1967), mais a été ensuite repris et adapté pour des paysages terrestres moins contrastés (voir Haila 2002). Malgré cette littérature grandissante, il est difficile de tirer des principes universels sur les effets de la perte et de la fragmentation de l'habitat car les milieux naturels sont fragmentés à différents niveaux, ce qui complique les comparaisons (Schmiegelow et Mönkkönen 2002). De plus, ils sont soumis à des changements continus dus aux processus naturels, ce qui rend les conclusions des études temporellement obsolètes (Haila 2002). Par ailleurs, les différents organismes réagissent de façon différente aux divers degrés de fragmentation, au contexte environnemental particulier et à des échelles spatiales et temporelles différentes (Haila 2002).

0.1.2. Fragmentation et échelles spatiales

Les effets de la perte et de la fragmentation de l'habitat ont souvent été évalués à l'échelle du paysage, car ils sont souvent étudiés dans un contexte de conservation et d'évaluation des risques d'extinction de certaines espèces ou populations. L'échelle du paysage ne présente pas de définition unique, elle est plutôt relative à l'organisme ou au processus étudiés (Turner et al. 2001). L'étendue spatiale et temporelle de l'échelle du paysage n'est donc pas fixe. De plus, contrairement à d'autres branches de l'écologie, l'écologie du paysage considère une hétérogénéité spatiale et temporelle intrinsèque à son objet d'étude (Forman et Godron 1986). La « théorie de la percolation » prévoit que la continuité d'un habitat ne sera rompue qu'après 40% de perte d'habitat (Gardner et al. 1987). Les parcelles d'habitat ne commenceront à être isolées qu'après 60% de perte d'habitat, mais c'est au-delà de 80% de perte d'habitat que les effets d'isolement seront réellement importants. Andrén (1994) a, quant à lui, prévu la réponse des populations animales le long de ce gradient de perte d'habitat à partir d'une méta-analyse. Selon lui, la réduction de la taille des populations sera d'abord proportionnelle à la perte d'habitat puis, sous un certain seuil (au-delà de 70% de perte d'habitat), le déclin deviendra plus drastique dû à l'effet additif de la fragmentation (Fig 0.2). Cependant, la sensibilité à la fragmentation de l'habitat est spécifique à chaque espèce, selon sa capacité de dispersion, ses spécificités en termes d'habitat et l'échelle spatiale qui la caractérise (With et Crist 1995, Andrén et al. 1997, Mönkkönen et Reunanen 1999). Ainsi, le seuil proposé par Andrén (1994) ne peut être utilisé comme ligne directrice en matière d'aménagement (Mönkkönen et Reunanen 1999). Les principaux effets de la fragmentation peuvent se résumer à une réduction de la superficie des îlots de forêt résiduelle, un plus grand isolement de ceux-ci et une augmentation de la quantité de lisières (Hunter 1990, Fahrig et Merriam 1994, Forman 1995). Les plus petits îlots abritent moins d'individus, le risque d'extinction y est donc plus élevé et le taux de colonisation plus faible (Fahrig et Merriam 1994).

Les effets de la perte et de la fragmentation de l'habitat peuvent se faire sentir à différentes échelles spatiales, à l'échelle du paysage sur les populations, mais aussi à des échelles plus fines qui affectent les individus et leurs comportements. Ces diverses échelles reflètent les échelles de perception que les animaux ont de leur environnement. En effet, Johnson (1980) décrit les différents niveaux de sélection d'habitat : le 1er niveau caractérise l'aire de répartition géographique de l'espèce, le 2^{e} niveau détermine le domaine vital d'un individu ou d'un groupe social, le 3^{e} niveau définit les types d'habitats utilisés à l'intérieur des domaines vitaux et finalement le 4^{e} niveau réfère au micro-site ou à des attributs spécifiques de l'habitat. Ainsi, si la qualité d'un habitat diminue ou si la superficie d'une parcelle d'habitat rétrécit, un individu peut répondre à cette perturbation en incluant des habitats adjacents, parfois sous-optimaux, ou d'autres parcelles du même habitat localisées à proximité (« théorie de la supplémentation », Dunning et al. 1992). Ceci aura donc des conséquences à la fois sur la superficie du domaine vital et sur les types d'habitats utilisés à l'intérieur de celui-ci (niveau 2 et 3 de Johnson 1980). À une échelle encore plus fine, les individus peuvent ajuster leurs déplacements en réponse à la perte d'habitat en évitant des milieux inadéquats (par exemple où le risque de prédation s'avère trop important). Dans ce cas, c'est la connectivité entre les différents habitats qui déterminera l'utilisation des parcelles d'habitat. Or, la perception de la connectivité est spécifique à chaque espèce, selon leurs capacités de déplacement notamment (With et Crist 1995). Afin d'avoir un portrait complet des effets de la perte et de la fragmentation de l'habitat sur une espèce donnée, il apparaît donc important d'étudier un gradient de perte d'habitat le plus large possible afin d'inclure les effets additifs de la fragmentation et d'analyser la réponse de la dite espèce à plusieurs échelles (populations, individus, mouvements).

0.1.3. Le cas de la forêt boréale

La forêt boréale a de tout temps été dynamique. En effet, les feux de forêts, plus ou moins fréquents, plus ou moins intenses, et couvrant de plus ou moins grandes superficies ont toujours induit des changements récurrents du couvert forestier dans le paysage (Gauthier et al. 1996, Bergeron et al. 1998, Bergeron et al. 2004). Ainsi, la forêt boréale subit périodiquement des pertes d'habitat à l'échelle locale et régionale de même qu'elle se fragmente naturellement. La distribution des classes d'âge des peuplements forestiers créée par les perturbations naturelles suit une courbe exponentielle négative (Fig. 0.3), les territoires sous régimes naturels de feux comportant une proportion importante de vieux peuplements. Ainsi, on sait par exemple que plus de 50% des forêts dépassent 100 ans

(Harper et al. 2002) dans un paysage où le cycle de feux est d'environ 140 ans (Bergeron et al. 2001). Cependant, dans les dernières décennies, les coupes forestières ont ouvert de plus en plus la matrice forestière et les niveaux de perte et de fragmentation de l'habitat atteints se situent maintenant en dehors de la plage de variabilité naturelle que la forêt boréale de l'est de l'Amérique du Nord ait jamais connue (Cyr et al. 2009). L'exploitation forestière a même dépassé les feux comme principal agent perturbateur à grande échelle, et ce, dans différents secteurs de la forêt boréale (Schroeder et Perera 2002, Perron 2003, Drapeau et al. 2009). La distribution des classes d'âge des peuplements forestiers issue d'un régime équien de coupes forestières est très différente de la forêt naturelle (Fig. 0.3) et tronque la portion des vieux peuplements qui sont ciblés par l'industrie. Les espèces inféodées aux vieilles forêts ou à des attributs d'habitat présents dans les vieilles forêts (par exemple, le bois mort) sont donc à risque, comme c'est le cas actuellement en Fenno-Scandinavie (Berg et al. 1994).

Malgré ce constat, le contexte particulier créé par les coupes forestières est rarement considéré dans les théories sur la perte et la fragmentation de l'habitat (Boutin et Hebert 2002), car on ne se situe pas dans un cas extrême de conversion des terres, comme c'est le cas de l'agriculture ou de l'urbanisation (Forman et Godron 1986). Les paysages résultants sont plutôt caractérisés par 1) une perte temporaire de l'habitat forestier, car les parterres de coupes régénèrent en forêt (Mönkkönen et Reunanen 1999), 2) l'importante matrice forestière environnante composée de l'habitat original (Schmiegelow et al. 1997, Drapeau et al. 2000) peut encore jouer un rôle de « source » pour les populations locales (Pulliam 1988), du moins pour le moment, et 3) l'habitat original résiduel est encore fortement connecté grâce à des corridors forestiers entre les parterres de coupes (Boutin et Hebert 2002).

0.2. Contexte d'initiation de la thèse

0.2.1. La modification de la loi sur les forêts de 2001

La présente thèse a été planifiée dans le contexte de la refonte de la loi sur les forêts (L.R.Q., c. F-4.1) adoptée en 2001 au Québec. Ces modifications découlent notamment de la sortie du documentaire « L'erreur boréale » réalisé par Robert Monderie et Richard Desjardins en 1999, qui a révélé au public québécois le fonctionnement de l'aménagement forestier en forêt boréale. Ce film a entraîné beaucoup de débats publics et de controverses.

Parallèlement, une consultation publique (1998), un rapport du vérificateur général sur la gestion des forêts (2000) ainsi qu'une commission parlementaire (2000) ont aussi contribué à inciter le gouvernement du Québec à réviser les règles encadrant l'industrie forestière et l'aménagement des forêts publiques.

Cette modification de la loi sur les forêts a conduit à un nouveau règlement sur les méthodes de dispersion des aires de récolte qui visait à implanter l'utilisation de « coupes en mosaïque » pour 60% des coupes réalisées sur les terres de l'État, à compter de 2005 (article 79.8 du Règlement sur les normes d'intervention (RNI) dans les forêts du domaine de l'État, R.R.Q., c. F-4.1, r.7). Les « coupes en mosaïque» sont des coupes avec protection de la régénération et des sols (CPRS) effectuées de manière à conserver, à l'intérieur de la limite du chantier de récolte, une « forêt résiduelle » ayant des caractéristiques particulières en terme de largeur, hauteur et densité du couvert (art. 79.2, RNI). La « forêt résiduelle » est une superficie forestière laissée intacte à proximité de la zone coupée. En fait, on peut se représenter grossièrement le principe des coupes en mosaïque comme un échiquier où chaque coupe sera adjacente à une forêt résiduelle de composition similaire et de superficie au moins égale à la zone coupée. Cette forêt résiduelle sera conservée tant et aussi longtemps que la zone coupée n'aura pas atteint 3 m de régénération (art. 79.7). Elle pourra alors être récoltée à son tour. Les deux superficies (coupée et résiduelle) sont planifiées simultanément et inscrites au plan d'aménagement forestier produit par les compagnies forestières.

Malgré l'incorporation des coupes en mosaïque dans la loi sur les forêts, ce type de coupe n'était au début des années 2000 que peu répandu et donc peu testé au Québec. Le principe dérivait à la fois d'un système de récolte « à 2 passes » développé, entre autres, en Alberta et en Ontario et du principe de « forêt mosaïque », un système « à 3 passes » développé en sapinière à la forêt Montmorency par l'Université Laval (Bélanger 2001). Les coupes en mosaïque, telles qu'inscrites dans la loi, se voulaient à l'origine une mesure d'harmonisation forêt-faune. Cependant, malgré la décision de les employer à grande échelle au Québec, la réponse de la faune à ces coupes n'avait pas vraiment été testée, ni dans les autres domaines que la sapinière (par exemple en pessière), ni dans de grands paysages.

0.2.2. La « Paix des Braves » de 2002

Parallèlement aux changements apportés à la loi sur les forêts, la signature de l'« Entente concernant une nouvelle relation entre le Gouvernement du Québec et les Cris du Québec » (aussi appelée « Entente Cris-Québec » et « Paix des braves ») en 2002 encourageait elle aussi l'utilisation des coupes en mosaïque sur le territoire sous entente, à hauteur de 75% de l'ensemble des coupes réalisées (art. 3.11). L'Entente Cris-Québec instaurait aussi un « régime forestier adapté » afin de minimiser les conflits d'usage entre l'exploitation forestière et les activités traditionnelles des Cris reliées à la récolte faunique notamment. Par exemple, elle impose la délimitation de zones d'intérêt particulier pour les Cris à l'intérieur des territoires de chasse traditionnels familiaux. Les maîtres de trappe décidaient de la localisation de ces zones : 1% de protection intégrale autour des sites culturels (par exemple camps, sites archéologiques et funéraires, source d'eau potable, art. 3.9) et 25% pour le maintien ou l'amélioration des habitats fauniques (art. 3.10). Dans cette 2^{e} zone (25%), seules les coupes en mosaïque ou les coupes partielles sont tolérées. De plus, certains critères spécifiques s'appliquent, visant notamment la limitation de la quantité de coupes permise et l'augmentation du délai avant la récolte des forêts résiduelles adjacentes aux coupes (la régénération doit atteindre 7 m au lieu de 3 m). L'entente fixe aussi une limite maximale (40%) à la quantité de forêts jeunes (issues de coupes ou de feux de moins de 20 ans) tolérée dans chaque territoire de chasse familial et impose la conservation en tout temps de 30% de forêts de plus de 7 m (art. 3.11).

Le territoire sous entente couvre une grande partie de la pessière noire à mousses du Nord-du-Québec, tel que défini par la Convention de la Baie James et du Nord Québécois (CBJNQ) en 1975. La CBJNQ définit par ailleurs sur ce territoire trois catégories de terres. Les terres de catégorie I sont réservées à l'usage et au bénéfice exclusif des Cris (~1-1,5%). Les terres de catégorie II sont des terres provinciales où les droits de chasse, pêche et piégeage sont exclusifs aux Cris, par contre l'exploitation des autres ressources (par exemple forestières) est ouverte à d'autres intérêts (mais doit se faire en collaboration avec la communauté locale Crie, ~ 15%). Enfin, les terres de catégorie III sont des terres publiques provinciales, mais avec des droits exclusifs concernant quelques espèces fauniques. En d'autres termes, l'exploitation forestière commerciale est permise sur les terres de catégorie II et III, telle que pratiquée sur l'ensemble du territoire de la province, alors que seules des entreprises Cries peuvent pratiquer la récolte de bois sur les terres de catégorie I.

0.2.3. La Forêt Modèle Crie de Waswanipi

La Forêt Modèle Crie de Waswanipi (FMCW) faisait partie du Réseau Canadien des Forêts Modèles, programme financé par le Service Canadien des Forêts du Ministère des Ressources Naturelles du Canada de 1992 à 2007. La FMCW, fondée en 1997, était la seule forêt modèle gérée par une communauté autochtone au Canada. L'objectif principal des forêts modèles était de développer, tester et partager des solutions d'aménagement forestier durable. La FMCW avait plus spécifiquement comme but de développer des stratégies pour minimiser les conflits entre les utilisateurs du territoire Cris et les aménagistes forestiers à travers l'implication de la communauté, la participation et le partage des connaissances. Ce maître projet a abouti à la publication d'un document présentant un processus collaboratif de planification de l'exploitation forestière (Waswanipi Cree Model Forest 2007), qui a d'ailleurs été exporté par la suite dans d'autres communautés Cries. Elle a aussi parrainé des projets de recherche sur des espèces fauniques d'importance pour les Cris, tels que le lièvre d'Amérique (*Lepus americanus*, Jacqmain 2003) et l'orignal (*Alces alces*, Jacqmain 2008). La communauté de Waswanipi a aussi ciblé la martre d'Amérique (*Martes americana*) comme espèce affectée par les coupes forestières et pour laquelle elle souhaitait voir se réaliser un projet en lien avec la foresterie sur leur territoire.

Le territoire couvert par la FMCW regroupe les 62 territoires de chasse traditionnels des Cris de Waswanipi (36 505 km^2). Les premières coupes forestières y ont eu lieu dans les années 1970 de manière sporadique. L'exploitation forestière intensive a réellement commencé au milieu des années 1980, d'abord au sud et le long des grands axes routiers (routes 113 et 109), puis de plus en plus vers le nord. On retrouve donc sur le territoire de grandes superficies coupées selon l'approche de dispersion conventionnelle. Celle-ci se caractérise par des coupes agglomérées, séparées par des « séparateurs de coupe » de 60 ou 100 m ou par des « bandes riveraines » de 20-30 m de part et d'autre d'un cours d'eau (selon le RNI, Fig 0.4). Plus récemment, vers le milieu des années 1990, deux secteurs ont aussi été exploités avec des coupes en mosaïque (dispersées, Fig. 0.4). Le premier suivait le modèle assez peu répandu à cette époque des « coupes à 2 passes », dites « par blocs équivalents », utilisées en Alberta et en Ontario durant la même période, mais qui deviendront la norme dans la loi sur les forêts de 2001. Dans ce cas, les coupes représentent environ 50% de la superficie lors de la 1^{re} passe. Le second secteur, situé sur les terres de catégorie I des Cris (selon la CBJNQ), suivait plutôt un modèle de « coupes à 3 passes », tel que développé à la Forêt Montmorency par l'Université Laval (principe de « forêt mosaïque », Bélanger 2001). Dans ce 2^e cas, les Cris avaient décidé d'adopter cette approche qui conservait en tout temps plus de forêts que de coupes sur le territoire, puisque chaque passe représente environ 30% du territoire. Ainsi, dans ce secteur, les coupes en mosaïque ne sont pas celles recommandées dans la loi sur les forêts ni dans l'Entente Cris-Québec. Cependant d'un point de vue de dispersion des coupes, les deux types de coupes en mosaïque présentent les mêmes caractéristiques, soient des coupes dispersées, séparées par des blocs de forêts résiduelles de superficie au moins égale à la coupe adjacente. Dans tous les cas, les coupes étaient de type coupe avec protection de la régénération et des sols (CPRS). Le territoire de la FMCW représente donc un terrain propice à étudier les effets des différents modes de dispersion des coupes forestières sur la faune. Notons que dans cette thèse nous n'avons pas traité les 2 types de coupes en mosaïque séparément, ne considérant que le patron de dispersion des coupes (agglomérées vs. dispersées). D'ailleurs, dans les 2 cas, l'industrie n'a réalisé que la première passe, ainsi seules la superficie des coupes et la quantité de forêt résiduelle diffèrent légèrement.

0.3. La martre d'Amérique comme modèle d'étude en réponse aux stratégies de dispersion des aires de coupe

0.3.1. La martre d'Amérique

La martre d'Amérique est un petit mustélidé spécialiste des forêts de l'Amérique du Nord. Son aire de distribution s'étend presque partout au Canada, ainsi que dans l'extrême nord et à l'ouest des États-Unis (Powell et al. 2003). Son espérance de vie en nature est d'environ 8 à 10 ans. Elle se reproduit habituellement à partir de l'âge de 2 ans et produit entre 1 et 5 jeunes par an (3-4 en moyenne). À travers son aire de répartition, la martre occupe une grande diversité de types forestiers (Bissonette et al. 1997): forêts résineuses

(Koehler et al. 1990, Slauson et Zielinski 2001), forêts mixtes (Bateman 1986, Potvin et al. 2000), forêts feuillues (Poole et al. 2004), forêts de seconde venue (Thompson et Curran 1995, Bowman et Robitaille 1997) et forêts ayant subi une épidémie de tordeuse des bourgeons de l'épinette (Chapin et al. 1997a). Les martres adultes, mâles et femelles, protègent un territoire et y sont fidèles (O'Doherty et al. 1997, Payer et al. 2004) entre les saisons et les années. La superficie des domaines vitaux semble très variable selon les régions, reflétant la qualité des habitats fréquentés (de 2,0 à 27,6 km² pour les femelles et de 2,6 à 45,0 km² pour les mâles, respectivement au Maine (Phillips et al. 1998) et au Labrador (Smith et Schaefer 2002)). Cependant, en moyenne, les femelles protègent des territoires de 5 km² et les mâles de 9 km². Ces territoires sont pratiquement exclusifs entre individus d'un même sexe, mais il existe un recoupement entre les sexes (Katnik et al. 1994, Payer et al. 2004).

L'hiver représente la saison critique pour la martre (Lofroth et Steventon 1991) qui ne possède pas de grandes réserves adipeuses (Harlow 1994, Robitaille et Cobb 2003) et doit donc adopter une stratégie comportementale pour maintenir une certaine balance énergétique (Drew et Bissonette 1997). Elle doit coordonner sa température corporelle, l'utilisation de sites de repos offrant une bonne protection thermique et la durée de ses périodes d'activités et de repos en fonction des conditions climatiques (Buskirk et al. 1988), par exemple en étant plus active durant le jour lorsque les températures sont plus élevées, ou en choisissant des tanières localisées sous la neige (thermiquement plus isolées) lors des grands froids. Durant cette période, les forêts complexes à dominance résineuse pourraient offrir une meilleure isolation, de nombreux débris ligneux permettant l'accès aux proies sous-nivales (Sherburne et Bissonette 1994) et aux abris (Chapin et al. 1997b), et un bon couvert latéral (Bissonette et Broekhuizen 1995) et vertical (Koehler et Hornocker 1977, Spencer et al. 1983, Smith et Schaefer 2002) de protection contre les prédateurs (Hargis et McCullough 1984, Drew 1995).

Le régime alimentaire de la martre est principalement carnivore durant l'hiver, alors qu'elle est opportuniste l'été, saison où elle mangera volontiers des insectes et des baies. Dans la majorité des études, les petits mammifères (*Myodes* et/ou *Microtus* selon les régions) représentent la proie principale, cependant le lièvre semble plus important dans l'est de l'Amérique du Nord (Martin 1994), peut-être en raison du meilleur apport énergétique qu'il représente (Cumberland et al. 2001). Les populations de lièvres (10 ans, Elton et Nicholson 1942, Krebs et al. 2001) et d'au moins une espèce de petits mammifères, le campagnol à dos roux (*Myodes gapperi*) (4 ans, Cheveau et al. 2004), sont cycliques en forêt boréale de l'est de l'Amérique du Nord. Cette dernière espèce s'avère être la proie la plus abondante dans cette région (Cheveau 2003), et une part potentiellement importante de l'alimentation de la martre entre deux captures de lièvres (Thompson et Colgan 1990). Ainsi, la présence ou non de ces deux proies peut influencer grandement le comportement de chasse de la martre, qui peut alors adapter son alimentation en se tournant vers des proies alternatives, telles les écureuils ou les gallinacés. La disponibilité et la vulnérabilité des proies varient entre les habitats, ce qui peut en conséquence affecter la sélection d'habitat de la martre (Coffin et al. 1997, Andruskiw et al. 2008).

0.3.2. Une espèce sensible aux perturbations de l'habitat

La martre d'Amérique compte parmi les organismes menacés par la perte et la fragmentation de l'habitat et par le déclin des vieilles forêts. De nombreuses études ont montré que la martre constitue une des espèces les plus sensibles aux perturbations forestières anthropiques (Thompson 1991, Chapin et al. 1998, Payer et Harrison 1999, Potvin et al. 2000). Par exemple, Soutière (1979) a enregistré une réduction de 60% de la densité de martres après la récolte de 67% du territoire en Nouvelle-Écosse, et Thompson (1994) a observé une baisse de 90% dans un territoire où il ne restait plus que quelques îlots résiduels (90% de récolte) en Ontario. Les chercheurs ont longtemps pensé que la martre était inféodée aux forêts matures de conifères (Strickland et Douglas 1987, Buskirk et Powell 1994), cependant il semble maintenant accepté qu'elle recherche plutôt une structure forestière verticale et horizontale complexe (Chapin et al. 1997a) lui garantissant l'accès aux proies, la présence d'abris durant l'hiver et une protection contre les prédateurs (Thompson et Harestad 1994). Cette structure complexe se retrouve principalement dans des forêts assez âgées, mais peut aussi être présente dans des peuplements plus jeunes qui ont subi un chablis ou une épidémie d'insectes. De plus, la martre évite clairement les milieux ouverts (Drew 1995) et ne peut tolérer plus de 30-40% de son domaine vital en coupes forestières en régénération de moins de 6 m (Chapin et al. 1998, Fuller et Harrison 2000, Payer et Harrison 2000, Potvin et al. 2000). Hargis et al. (1999) ont par ailleurs montré qu'à l'échelle du paysage la présence de la martre était compromise à partir du seuil de 25% de milieux ouverts. Ces caractéristiques, ajoutées au fait que la martre est un prédateur qui occupe un grand domaine vital (proportionnellement trois fois plus grand que prédit par sa masse corporelle, Lindstedt et al. 1986, Buskirk et McDonald 1989), en font une bonne candidate comme espèce indicatrice du rôle fonctionnel des forêts résiduelles à l'échelle du paysage (Bissonette et al. 1989, Buskirk et Ruggiero 1994).

0.3.3. Une espèce d'intérêt économique et culturel

La martre est d'autre part une espèce extrêmement prisée aussi bien par les trappeurs blancs qu'autochtones. Ainsi, en 2008-2009, 23 800 martres ont été piégées au Québec, générant des revenus de plus d'un million de dollars, ce qui en fait l'espèce piégée qui génère le plus de retombées économiques (Bulletin Fourrure Québec, MRNF). Pour les Cris de Waswanipi, la martre représente la 2^e espèce piégée en terme de nombre de peaux vendues (40%, moyenne sur les neuf dernières années d'après les statistiques de piégeage du MRNF, comm. pers.) mais la première en terme de revenus générés avec 53% (basé sur la valeur moyenne annuelle des prix des fourrures, Bulletin Fourrure Québec, MRNF, Tableau 0.1). D'un point de vue culturel, la martre fait partie des espèces d'intérêt pour les Cris avec l'orignal, le castor, le lièvre, les poissons, le caribou et les perdrix (Entente Cris-Québec, art. 3.10). Même s'il est impossible pour les Cris de prioriser des espèces (leur vision du monde est holiste), les espèces « utilisées » sont quand même plus critiques à leurs yeux.

0.4. Objectifs de la thèse

Cette thèse se divise en quatre chapitres. Les trois premiers examinent des aspects écologiques liés à la perte et de la fragmentation de l'habitat sur la martre d'Amérique. Le quatrième aborde un aspect plus sociologique, impliquant les connaissances des trappeurs de martres Cris et leurs perceptions des effets de la récolte forestière sur cette espèce. Pour la partie écologique, la thèse a été articulée autour d'échelles spatiales de plus en plus fines (Tableau 0.2). Ces trois chapitres analysent tour à tour l'échelle du paysage au niveau de la population locale, l'échelle des domaines vitaux à travers les individus de la population locale et l'échelle fine au niveau des mouvements des individus (Tableau 0.2). Notre site d'étude, la FMCW, étant composée à la fois de paysages récoltés par coupes conventionnelles (agglomérées) et de paysages aménagés par coupes en mosaïque (dispersées), offre l'opportunité de séparer les effets liés à la perte d'habitat (c'est-à-dire la quantité de coupes) des effets liés à la fragmentation (c'est-à-dire la dispersion des coupes et quantité de lisières créées).

Le chapitre 1 évaluera les effets de la perte et de la fragmentation de l'habitat par l'exploitation forestière sur la martre, à l'échelle du paysage, en forêt boréale du nord-ouest du Québec. Nous comparerons dans un premier temps l'abondance des martres dans un paysage (plusieurs centaines de km^2) naturel (non coupé) avec des paysages coupés, soit par coupes agglomérées, soit par coupes dispersées. Ensuite, nous déterminerons si la réponse des populations de martre à la perte et à la fragmentation de l'habitat, à différents niveaux (présence-absence, abondance et condition des individus), est proportionnelle à la perte d'habitat ou s'il existe un seuil critique, et ce, le long d'un gradient de perte d'habitat.

Le chapitre 2 documentera la réponse individuelle des martres à la perte et à la fragmentation de l'habitat ainsi que la sélection d'habitat hivernal en forêt boréale aménagée. Dans ce chapitre, nous tenterons de déterminer la stratégie adoptée par les individus en réponse à la perte d'habitat à l'intérieur de leurs domaines vitaux hivernaux. Dans de telles études, il est indispensable d'identifier conjointement quels sont les habitats sélectionnés et évités par les martres afin d'éclairer notre compréhension de la stratégie adoptée face à la perte d'habitat.

Le chapitre 3 évaluera l'effet de la création de lisières forêt/coupe abruptes sur les mouvements de la martre, ainsi que l'utilisation des corridors forestiers (séparateurs et bandes riveraines) dans la forêt boréale aménagée par l'industrie forestière. Considérant que la martre évite les milieux ouverts, comment se comporte-t-elle en présence d'une lisière abrupte? De plus, nous documenterons en quoi les bandes résiduelles étroites laissées dans les paysages de coupes agglomérées jouent un rôle de connectivité pour la martre.

Le chapitre 4 présentera les connaissances traditionnelles des trappeurs Cris concernant la martre de même que leurs perceptions des effets des coupes forestières sur cette espèce. Le but est ici de développer une vision commune (Cris-scientifiques) de la martre en forêt boréale aménagée en faisant ressortir les éléments concordants et discordants entre les connaissances des trappeurs et les résultats des trois précédents chapitres de la thèse. L'établissement d'une vision commune représente la première étape vers l'implantation d'une approche de gestion collaborative impliquant la communauté dans les processus de décision.

La conclusion générale synthétise les résultats des différents chapitres de cette thèse autour de thèmes communs abordés aux différentes échelles spatiales. Ces résultats sont aussi mis en lien avec la législation actuelle afin de faire ressortir les défis à anticiper. Enfin, deux annexes ont été ajoutées à la fin de la thèse. La première peut servir d'introduction pour le 4^{e} chapitre, présentant l'intérêt des connaissances traditionnelles écologiques et leur contribution à l'aménagement forestier. La deuxième décrit une nouvelle méthode d'anesthésie des martres développée en collaboration avec des médecins vétérinaires pour les besoins de ce projet.

Tableau 0.1. Statistiques de piégeage de la martre pour le territoire de Waswanipi, telles que compilées par le Ministère des Ressources Naturelles et de la Faune du Québec (zone 88 de piégeage et de gestion des animaux à fourrure (PGAF), région Nord-du-Québec) entre 2000 et 2009.

	Nombre de peaux vendues				Revenu généré		
Année	Martre	Total	%	Prix moyen	Martre	Total	%
2008-2009	188	457	41	43,23 $	8 127 $	13 679 $	59
2007-2008	324	811	40	55,77 $	18 069 $	30 458 $	59
2006-2007	384	1300	30	59,30 $	22 771 $	48 250 $	47
2005-2006	802	1741	46	90,70 $	72 741 $	108 793 $	67
2004-2005	998	2171	46	50,57 $	50 469 $	85 206 $	59
2003-2004	613	1815	34	43,62 $	26 739 $	67 206 $	40
2002-2003	370	1197	31	41,88 $	15 496 $	41 660 $	37
2001-2002	685	1596	43	45,03 $	30 846 $	59 700 $	52
2000-2001	1204	2509	48	36,22 $	43 609 $	80 276 $	54

Tableau 0.2. Échelles spatiales analysées dans les différents chapitres du projet de thèse.

Échelle		Niveau de résolution	Éléments suivis	Chapitre
Paysage	(150-500 km^2)	Population	Occupation des paysages	1
Domaine vital	(1-11 km^2)	Sous population	Sélection d'habitats	2
Fine	(500 m)	Individus	Mouvements	3

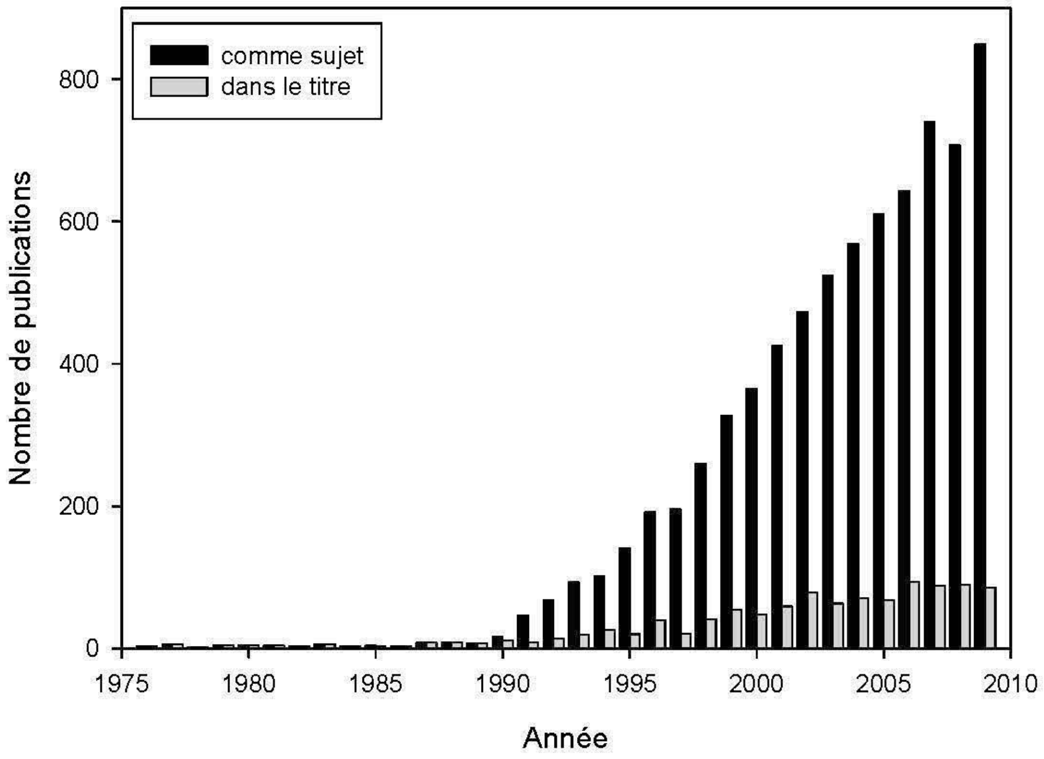

Figure 0.1. Nombre de publications référencées dans *Web of Sciences* (*ISI Web of Knowledge* v.4.8, consulté le 5 avril 2010), pour lesquelles le terme « fragmentation » apparaît dans le titre ou comme sujet dans les domaines de l'écologie, des sciences de l'environnement, de la conservation de la biodiversité et de la zoologie.

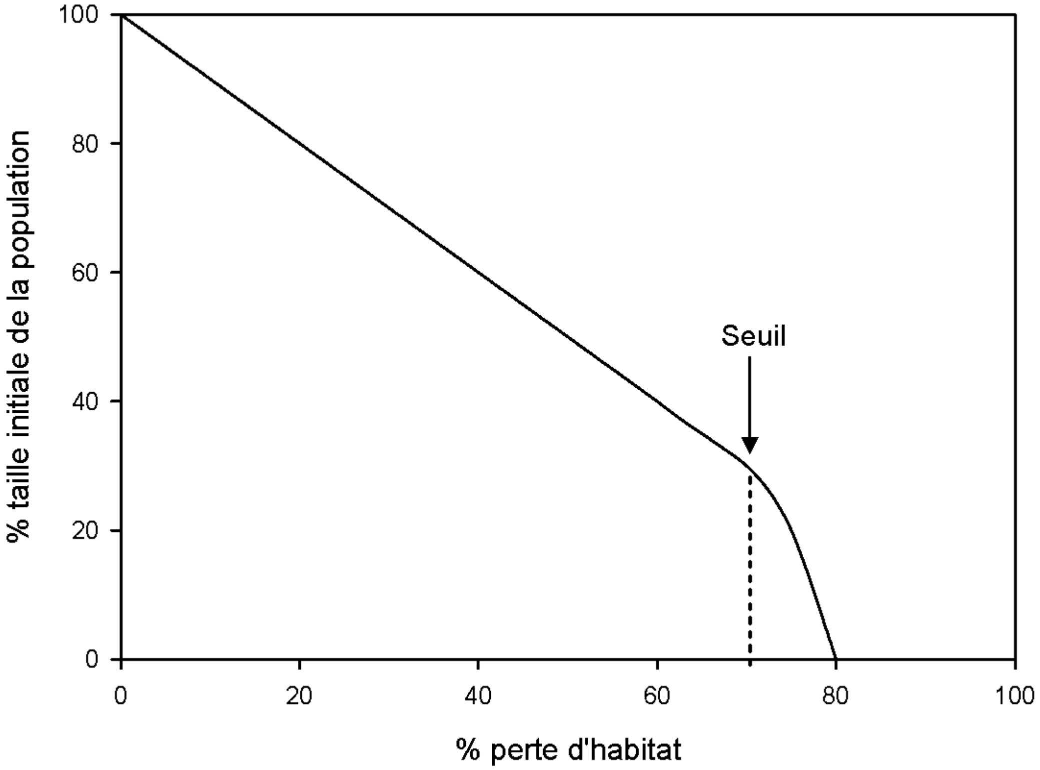

Figure 0.2. Réponse de la taille des populations animales à la perte et à la fragmentation de leur habitat (d'après Andrén 1994). Le seuil théorique où les effets de la fragmentation s'ajoutent aux effets de la seule perte d'habitat est indiqué.

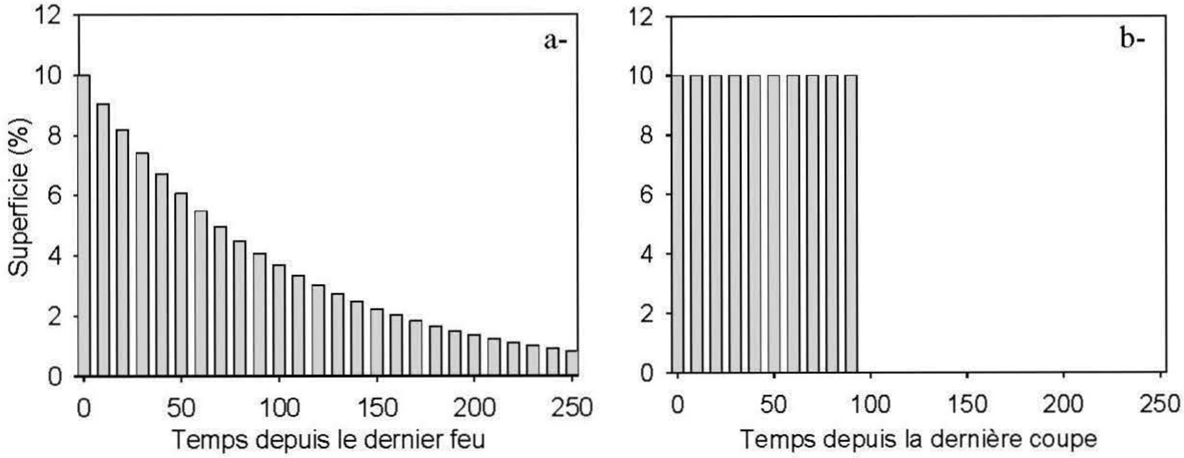

Figure 0.3. Distribution des classes d'âge des peuplements forestiers pour (a -) une forêt naturelle dynamisée par un cycle de feux de 100 ans et (b -) une forêt aménagée selon une révolution forestière de 100 ans. Tiré de Bergeron et al. (2002).

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Figure 0.4. Coupes en mosaïque (dispersées, « à 3 passes ») et coupes conventionnelles (agglomérées) sur le territoire de la Forêt Modèle Crie de Waswanipi (photos : M. Cheveau).

CHAPITRE I

REMNANT HABITAT USE BY AMERICAN MARTEN IN MANAGED LANDSCAPES OF THE EASTERN BOREAL FOREST OF CANADA SUGGESTS TOLERANCE TO HABITAT LOSS AND FRAGMENTATION

Marianne Cheveau, Louis Imbeau, Pierre Drapeau and Louis Bélanger

Résumé

L'exploitation forestière industrielle réduit la quantité des vieux peuplements forestiers dans la forêt boréale nord-américaine et ouvre actuellement la matrice forestière à un degré jamais expérimenté sous un régime de perturbations naturelles. Les espèces qui évitent les milieux ouverts sont donc de plus en plus soumises à la perte et à la fragmentation de leur habitat, ce qui peut affecter la persistance de leurs populations. Parmi les espèces de fin de succession, la martre d'Amérique (*Martes americana*) est connue comme une des plus sensibles aux perturbations d'origine anthropique. Dans la forêt boréale de l'est du Canada, où les coupes sont agglomérées ou dispersées à la manière d'un damier, nous avons évalué les effets de la perte (proportion de coupes) et de la fragmentation (configuration des habitats résiduels) de l'habitat sur les populations de martres à l'échelle du paysage (100-500 km^2) et du domaine vital (5-10 km^2). La martre est ici considérée comme une espèce focale qui peut guider l'aménagement forestier dans le futur pour des espèces qui ont les mêmes besoins en terme d'habitat, mais des domaines vitaux plus petits. Nous avons capturé 156 martres en deux automnes le long d'un gradient de perte d'habitat, dans des paysages exploités et non exploités. À l'échelle du paysage, l'abondance des martres était deux fois supérieure dans le paysage non exploité que dans les paysages exploités, et ce, malgré une proportion similaire de forêt mature. L'abondance des martres était similaire dans les deux types de paysages aménagés, malgré le fait que les proportions de forêt d'intérieur et la connectivité étaient plus élevées dans les paysages de coupes dispersées. À l'échelle des domaines vitaux, nous avons trouvé une faible relation entre la proportion de coupes dans le contexte environnant, et, l'utilisation des sites par les martres et leur abondance dans 143 sites (0 à 72% de coupes). Nous n'avons pu détecter aucun déclin marqué (seuil) dans l'occurrence ni l'abondance des martres. Au contraire, la relation montrait plutôt une réduction linéaire et inférieure à une pente 1 pour 1. Nous concluons que les populations de martres dans les paysages boréaux aménagés sont certes sensibles à l'exploitation forestière, mais pas à la configuration et peu à la quantité de coupes, lorsque ~50% des forêts sont maintenues dans le paysage environnant, et ce, en terme d'utilisation des sites et d'abondance. Alors que ces résultats suggèrent que la martre est plus tolérante à la perte d'habitat que ce que l'on assumait précédemment, la condition physique était affectée négativement par la perte d'habitat et sa configuration, montrant que les effets négatifs étaient détectés d'abord au niveau des individus plutôt qu'au niveau des populations.

Mots-clés

Martre d'Amérique, dispersion des coupes, fragmentation, échelle du paysage, échelle du domaine vital, modèles d'occupation de sites, Premières Nations

Abstract

Industrial timber harvesting is reducing old forest stands in the North American boreal forest, and currently opening up the forest matrix to a degree not previously experienced under natural disturbance regimes. Species that avoid open areas are thus increasingly subjected to habitat loss and fragmentation, potentially affecting population persistence. Among late-successional species, American marten (*Martes americana*) is reported as one of the species most sensitive to human-induced disturbances. In the boreal forest of eastern Canada, where clearcuts are clustered or dispersed in a checkerboard pattern, we evaluated the effects of habitat loss (proportion of clearcuts) and fragmentation (configuration of remnant habitats) on marten populations at landscape (100-500 km^2) and home-range (5-10 km^2) scales. Marten was considered a focal species, which could guide future forest management for species having similar requirements, but smaller home ranges. We captured 156 martens over two years along a gradient of habitat loss through unharvested and harvested landscapes. At the landscape scale, marten abundance was two-fold higher in unharvested than in harvested landscapes, despite a similar proportion of mature forests. Abundances were similar in the two harvested landscape types, despite interior forests and connectivity being greater in dispersed-cut landscapes. At the home-range scale, we found a weak relationship between the proportion of clearcuts in the surrounding context and marten site use and abundance on 143 sites (0 to 72% clearcuts). Here, we failed to detect a sharp decline (threshold) in marten occurrence and abundance; rather, the relationship showed a linear, less than 1:1, decrease. We concluded that marten populations in managed boreal landscapes are sensitive to forest harvesting, but not to the configuration and only slightly to the amount of residual forest when ~50% forests are maintained in the surrounding landscape, in terms of site use and abundance. While such results suggest that marten may be more tolerant to habitat loss than previously assumed, body condition was negatively affected by habitat loss and its configuration, showing that negative effects are first detected at an individual- rather than at a population-level.

Key words

American marten, clearcut dispersion, fragmentation, landscape scale, home-range scale, site-occupancy models, First Nation

1.1. Introduction

Boreal forest ecosystems have historically experienced dynamic changes in forest cover (due to forest wildfires, Bergeron et al. 2001) that reflect both local and regional losses as well as the fragmentation of late-seral habitats, but ongoing industrial timber harvesting is opening up the forest matrix to a degree not previously experienced in the eastern boreal forest of North America (Cyr et al. 2009). In fact, over the last several decades, timber harvesting has become the main agent of disturbance at the landscape scale in many parts of the boreal forest (Schroeder and Perera 2002, Perron 2003, Drapeau et al. 2009). Theoretical models considering effects of habitat loss and fragmentation on organisms have generally shown that the effects of habitat loss are much stronger than those related to habitat fragmentation (Fahrig 1997, 2003). Indeed, continuity of the original habitat is broken down into several patches only when around 60% of the original habitat still remains (percolation theory, Gardner et al. 1987, Andrén 1994). However, patches do not become isolated until the remaining original habitat is < 40%. When < 20% of the original habitat remains, the distance between patches increases exponentially, amplifying isolation effects. While a decline in population size of a species living in the original habitat will be linear and proportional to habitat loss in landscapes with a high proportion of suitable original habitat, the number, size, and degree of isolation of patches will also begin to influence population size of sensitive organisms (an additive effect) at some threshold (extinction threshold, between 40 and 20% of original habitat), thereby yielding a non-linear (faster than expected) response (Andrén 1994). This theoretic response of population size should be considered in light of factors such as habitat specificity of a species (Andrén et al. 1997), together with its movement abilities and spatial scales of perception (With and Crist 1995, Mönkkönen and Reunanen 1999). Consequently, threshold responses are also likely to vary between species, because functional connectivity of a landscape (see With and Crist 1995) is species-specific (D'Eon et al. 2002).

Boutin and Hebert (2002) suggested that future research on habitat loss and fragmentation effects should aim at identifying thresholds for sensitive focal species, which could afterward guide forest management or biodiversity conservation. Nevertheless, in reviewing the literature, Fahrig (2003) found no convincing empirical evidence for extinction thresholds and showed that effects of fragmentation *per se* are as likely to be positive as negative. Yet the shape of species response curves is not always as drastic as a “break-point” threshold, but could also include a “zone-type threshold” (Huggett 2005), or a gradual decline without a real threshold (Drapeau et al. 2009). In addition, the same species may exhibit different thresholds at different spatial scales within a landscape (Huggett 2005).

American marten (*Martes americana*) is considered to be one of the species most sensitive to disturbances induced by timber harvesting (Thompson 1991). Martens clearly avoid open areas (Koehler and Hornocker 1977, Hargis and McCullough 1984, Buskirk and Powell 1994, Drew 1995) and cannot tolerate more than 30-40% of its home range in regenerating areas (Chapin et al. 1998, Payer and Harrison 1999, Potvin et al. 2000). Martens are known to prefer forests with complex vertical and horizontal structure (Chapin et al. 1997), attributes that are found mainly but not exclusively in late-seral stands. This landscape sensitivity to clearcuts and the resultant loss of late-seral forests is consistent over a large part of the species’ geographic range that has been subjected to timber harvesting (Bissonette et al. 1997) and may be related to the size of its home range, which is proportionally larger than that predicted by its body size (Lindstedt et al. 1986, Buskirk and McDonald 1989). Therefore, marten is generally considered a forest specialist (Buskirk and Powell 1994) that avoids not only open but also regenerating areas (Potvin et al. 2000). It is thus viewed as a good indicator species of the functional role of late-seral forest habitat at the landscape scale (Bissonette et al. 1989, Buskirk and Ruggiero 1994).

Marten response to habitat loss is more frequently inferred from studies conducted on individuals within their home range (Chapin et al. 1998, Payer and Harrison 1999, Potvin et al. 2000, Fuller and Harrison 2005, Dumyahn et al. 2007) than from studies examining population level responses over larger landscapes. Indeed, a few studies have measured marten responses to habitat loss and fragmentation at the landscape scale. For example, Soutière (1979) mentioned a 60% decrease in marten density in a territory where habitat loss attained 67% in Nova Scotia, and Thompson (1994) noticed a 90% decrease in marten abundance in a territory where only small residual patches were maintained (~ 90% habitat loss) in Ontario. Otherwise, Bissonette et al. (1997) suggested that marten was influenced by both loss and fragmentation of habitat because its response was more drastic than a one-to-one decline expected from a net loss of habitat.

We investigated marten population responses to habitat loss and fragmentation generated by extensive forest harvesting in the eastern boreal forest of Canada. We conducted capture-recapture studies of martens in unharvested and harvested forest landscapes (several hundred km^2). In this region, forest was harvested with clearcuts under two dispersion patterns: clustered versus dispersed. The latter pattern is now promoted through an agreement between the Quebec provincial government and the Cree First Nation, which aims to mitigate impacts of forest harvesting on activities of indigenous peoples. We predicted that overall abundance patterns of marten would be higher in unharvested than in harvested landscapes, and that abundance would be higher in dispersed-cut than in clustered-cut landscapes, where the configuration of remnant forest is made of linear habitats instead of patches. Given that the spatial distribution of remnant habitats is not uniform within both harvested landscapes, we further investigated marten responses (in terms of occurrence, abundance, and body condition) along a gradient of varying amounts of clear-cutting in the neighbourhood of remnant habitats at a spatial scale that corresponds to an individual home range (5-10 km^2) in our study area (Cheveau et al. chapter 2). Finally, we determined whether marten occurrence and abundance patterns showed a threshold response with respect to habitat loss and fragmentation in these managed landscapes.

1.2. Methods

1.2.1. Study area and sampled landscapes

Our study took place in northwestern Quebec (49°45’N, 76°00’W), in the territory of the Waswanipi Cree Model Forest (WCMF), an area covering 36,505 km^2 (Fig. 1.1a). The WCMF was part of the Canadian Model Forest Network, which has aimed to develop, test and share solutions in sustainable forest management. The WCMF was the only Aboriginal-led Model Forest, and was specifically established to minimize conflicts between Cree land users and forest managers through community involvement, participation, and knowledge sharing. Our study area was also within the boundaries of the *Agreement concerning a new relationship between le Gouvernement du Québec and the Crees of Québec* (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002), where forest management is subjected to forest cover retention ("mosaic cutting" strategy) that is linked to aboriginal wildlife management. This agreement also imposes a maximum percentage of regenerating areas (40% from natural and human origin) within each management unit, corresponding to familial hunting territories, in which case timber harvesting is temporarily halted.

The study was located in the James Bay lowlands, which are characterized by mostly flat topography (with abundant poorly drained soils) with sparse hills. Mature forests are mainly black spruce (*Picea mariana* [Mill.] BSP) stands, with an understory of feather moss, sphagnum and dwarf Ericaceous shrubs (mature forest is hereafter defined as forest ≥ 7 m in height, as opposed to late-seral forests, which are hereafter defined > 120 years, as estimated from forestry maps). Forest fires are the major natural disturbance in this region (Le Goff et al. 2007). Human disturbances (logging) have affected the landscape, mainly since 1988. The spatial pattern of clearcuts was either clustered (i.e., large cutover areas several hundred km^2 in size) or dispersed in a checkerboard pattern (i.e., "mosaic cutting" strategy) (Fig. 1.1b and 1.1c), following the two cutting strategies used in Quebec's boreal forest over the past 10 years. In the clustered-cut landscapes, 50-200 ha cut-blocks were separated by separator strips. Hence, remnant forest in clustered cutting landscapes was composed of narrow corridors, 20 to 100 m wide, along streams and lakes or between cut-blocks. In the dispersed-cut landscapes (10-80 ha cut-blocks), harvesting was limited to 50 % of the mature forest. Uncut forest blocks that remain were equivalent in size to adjacent cut-blocks. Most of the clearcuts on the study area are not replanted and natural regeneration was generally < 1 m in height.

We surveyed five landscapes covering 173 to 480 km^2 each (Fig. 1.1a). Each landscape was composed of mature forests (intact or residual forests left after logging, ≥ 7 m height, Appendix 1.1), regenerating and recently cut stands (< 3 m), regenerating stands after fire (mainly a large one in 1986), forested bogs, water, and non-forested categories (roads, open barrens, alder swamps, total blowdowns) (Table 1.1). Two landscapes were harvested using clustered clearcuts (C1 and C2), two landscapes were harvested using dispersed clearcuts (D1 and D2), and one landscape was mostly unharvested (UH, Fig. 1.1b and 1.1c). Given that access to forests relies on roads, which are built only in managed landscapes, sampling unharvested landscapes was more difficult. However, some roads were built to access an old fire (1986) for salvage logging, which occurred in a small section of that wildfire. These roads crossed a lightly harvested zone, which represented our unharvested landscape (Fig. 1.1c). This sector was thus composed of 3 m-high post-burned regenerating areas and mature forests (Table 1.1). Despite high local variability in the proportion of mature forests, the overall percentage was quite similar across landscapes (~50%), including the unharvested one. The percentage of clearcuts, however, greatly contrasted between landscapes, ranging from 9% in the unharvested area, to over 20% in dispersed-cut landscapes, and upwards of 30-45% in clustered-cut landscapes (Table 1.1).

Forest cover types included a lower proportion of mixedwood forests in the northern (4-8% of all mature forests) than in the southern part (up to 22% of all mature forests) of the study area (Fig. 1.1). These stands were mostly composed of small patches of trembling aspen (*Populus tremuloides* Michx.), paper birch (*Betula papyrifera* Marsh.) and balsam fir (*Abies balsamea* (L.) Mill.), interspersed with pure black spruce stands (Appendix 1.1).

1.2.2. Marten capture and prey availability

We live-trapped martens during two autumn seasons (3 October to 8 November 2005, and 27 September to 6 November 2006) at 470 capture stations, each separated from the others by a minimum 1 km distance. In 2005, we sampled 59, 94 and 53 capture stations in C1, D1, and UH, respectively. In 2006, we sampled 153 and 111 capture stations in C2 and D2, respectively. At each capture station, we installed a *Tomahawk 202* trap (Tomahawk live traps, Tomahawk, WI) for an average of 7 nights. Trapping effort for the entire study totaled 3309 trap-nights. Traps were baited with meat (mostly beaver), covered by fir or spruce branches, and a commercial lure for marten (Makwa, Stoneham, QC, Canada) was also added. Each captured animal was anaesthetized using isoflurane gas (see methodology in Desmarchelier et al. 2007, available in the appendix A2 in this thesis), tagged, weighed, measured, and sexed, and a tooth (mainly PM1) was removed for cementum analysis to obtain the animal's age (conducted by Matson's Laboratory LLC, Milltown, MT; adult ≥ 1-y-old, juvenile < 1-y-old). As an index of general health, we used body index (mass/body length). We also classified marten according to their fur color (dark brown, reddish-brown, and pale), which is a concern for Cree trappers because such coloration types generally affect pelt prices, with dark pelts being preferred.

Because small mammals are prey for marten and because they exhibit strongly cyclic population dynamics in our region (Cheveau et al. 2004), annual variation in prey abundance patterns could influence marten habitat use over the two years of our study (a potential confounding effect). Consequently, we captured small mammals in the same area for two summers (July 2005 and 2006) prior to the marten trapping sessions. We visited the same sites during the two summers. We installed two parallel lines of *SFG Sherman* traps (H.B. Sherman traps, Tallahassee, FL) at 20-m intervals (6 traps in the clearcut and 16 in the residual forest) in 17 transects, perpendicular to the forest edge, located in D1 (N = 10 transects, 720 trap-nights) and D2 (N = 7 transects, 432 trap-nights). We also installed traps (6 in the clearcut, 6 to 12 in the residual corridor depending on its width, and 6 others in the opposing clearcut) in 17 transects located in C1 (N = 6 transects, 372 trap-nights) and C2 (N = 11 transects, 588 trap-nights), for a grand total of 2112 trap-nights each year. Small mammal traps were baited with peanut butter and apple, and were set for 3 consecutive nights. Each animal was tagged, weighed, sexed, identified to species and immediately released. The trapping protocols were approved by the Institutional Committee of the Canadian Council for Animal Care (CCAC) at UQAT (Université du Québec en Abitibi-Témiscamingue).

1.2.3. Landscape connectivity

We compared landscape connectivity between the three landscape types (unharvested versus harvested: clustered and dispersed clearcuts) using the probability of connectivity index (PC) developed by Saura and Pascual-Hortal (2007), using *Conefor Sensinode 2.2* (Pascual-Hortal and Saura 2006, Saura and Pascual-Hortal 2007). This index considers connectivity from a functional perspective (With et al. 1997) by accounting for the focal species movement abilities. It combines habitat loss and spatial arrangement of the residual habitat, within a graph theory framework (Saura and Pascual-Hortal 2007). The area of each landscape was delineated with a 2-km buffer around each capture station (see Fig. 1.1), which described the immediate habitat context around the stations. Here, we defined “non-habitat” as clearcuts, bogs, non-forested areas and water (< 4-m), whereas “potential habitat” was the intact or residual forest (≥ 7-m). We also included regenerating stands after fire as potential habitat for marten, given that Potvin et al. (2006) found in a nearby area that stands 4 to 7-m height were habitats of intermediate quality for marten. Consequently, we considered that 20-year-old post-fire stands (~ 3-m high) represent habitat more permeable to marten movements than clearcut regeneration blocks < 1-m in height. We also calculated % interior habitat in each landscape as the core area of potential habitat (after removing a 50-m interior buffer). To compute PC of a landscape from a marten’s perspective, *CS 2.2* requires three input values. First, we provided species dispersal ability, which is the maximum distance traveled during dispersion. In our study area, we found that some unsettled martens dispersed over a distance of > 100 km (see Results section below). Dispersal abilities were thus not a restricting factor at the scale of our analysis. Second, the probability of connectivity requires the maximum distance that the species can cross in non-habitat conditions, which was set at 600 m for marten, as determined from the available literature (see Snyder and Bissonette 1987). Third, we provided an associated probability of crossing non-habitat with respect to distance between suitable patches. We extrapolated these values from Snyder and Bissonette (1987), who showed that 87% of crossings were over distances < 250 m. As an example based on these reference values, the probability of crossing a 150-m gap is estimated to be only 30%, using a negative exponential distribution. Last, Chapin et al. (1998) found that marten did not use habitat patches less than 2.7 ha in area. Consequently, these small patch sizes were not considered in this analysis.

1.2.4. Scale definition

We analyzed marten responses to habitat loss and fragmentation at two scales: the landscape scale and the home-range scale. At the landscape scale (100-500 km^2), we considered our five landscapes (described above) as the sampling units. Because we investigated only five landscapes, not so contrasted in terms of % residual forests, we also analyzed marten response inside sites (sampling units) showing the same size as an average home range in the region (5-10 km^2, Cheveau et al. chapter 2). While we did not determine individual home ranges in the present study, we hereafter refer to the home-range scale (5-10 km^2) for these analyses.

1.2.5. Site definition and habitat context variables

To examine relationships between marten occupancy and habitat context variables at the home-range scale, we grouped capture stations (< 2.5 km apart, N = 2 to 7 capture stations) and created a 1-km buffer around these grouped stations with *ArcGIS 9.3* (ESRI 2008). This process created 143 *ad hoc* "sites" (4.8 to 12.5 km^2) for which we characterized seven metrics describing habitat composition and configuration. Habitat categories were derived from digital forest maps (1:20 000) created by the Ministry of Natural Resources and Wildlife of Quebec. We selected variables that are known to influence marten habitat use at this scale on the basis of available literature. Habitat composition was defined by % clearcuts (%CC), % mature forests (%MF), % late-seral forests (%LSF), and % mixedwood forests (%MIX):

1) %CC – Clearcuts represent net habitat loss for martens, given that they avoid open and low regeneration areas (Buskirk and Powell 1994);
2) %MF – Marten is a forest-dwelling species (Powell et al. 2003), which tends to establish its home range in areas with < 40% early-successional forests (Chapin et al. 1998);
3) %LSF – Martens prefer forests with complex vertical and horizontal structure (Chapin et al. 1997), attributes that are frequently found in late-seral forests;
4) %MIX – Mixed-wood forests were selected by martens at the home-range scale, as demonstrated in a previous study in northwestern Quebec (Potvin et al. 2000). Such stands potentially present higher prey richness, as shown for other vertebrates (Hobson and Bayne 2000). Because marten are opportunistic in their food habits, a locally more diverse prey community could influence marten habitat use.

Habitat configuration was described using edge density (clearcut/forest edges, m/ha), % late-seral forest core area (after removing a 50-m interior buffer, as proposed by Potvin et al. 2000, %CALSF), and % mature forest core area (%CAMF):

5) EDGE – Edge density is an index of the shape and size of clearcuts. Martens tend to minimize clearcut/forest edges inside their home range in the boreal forest (Potvin et al. 2000);

6) %CAMF – Potvin et al. (2000) also found that marten home range had larger proportions of core area of mature forest than available in the landscape;

7) %CALSF – If martens prefer late-seral forests and core area of forest habitat, we assume that a core area of late-seral forests would also be preferred.

These configuration variables were highly correlated with associated composition variables (EDGE vs. %CC : correlation = 0.809, %CAMF vs. %MF: corr. = 0.961, %CALSF vs. %LSF: corr. = 0.975).

1.2.6. Data analyses

1.2.6.1. Marten broad abundance patterns across landscapes

Overall marten abundances (captures per 100 trap-nights) were calculated in each of the five sampled landscapes. As mature forest cover was similar across landscapes, a linear regression was performed on observed marten abundance versus the percentage of non-habitat in each landscape.

1.2.6.2. Movement patterns and food resources

To detect potential dispersion movements in the study area, recapture data were analyzed for martens caught more than once, in the same trap and in different traps. Movement distances of recaptured martens were compared between adults and juveniles, for the two years, using two-way ANOVA on the log-transformed data.

We calculated small mammal abundance (number per 100 trap-nights) of the different species for each year. We performed hierarchical linear mixed models (GLMM, generalized linear mixed models) to compare abundance between years and harvested landscape types (fixed effect: year and landscape type; random effect: transects included in landscapes C1, C2, D1, and D2) for all small mammals and for red-backed voles (*Myodes gapperi*), the most abundant species in the region.

1.2.6.3. Marten site-occupancy analyses

Because each site was sampled for several nights, the probability of marten detection may vary from one visit to another due to variable environmental conditions. We used site-occupancy models to assess the importance of habitat context variables with regards to time-varying covariables on marten occurrence (MacKenzie et al. 2002, 2006). This analysis was conducted at the home-range scale. We modeled marten site-occupancy (ψ) using *PRESENCE 2.1* (MacKenzie et al. 2006), by analyzing capture histories at each site (presence/absence) for all martens, adults, and juveniles, in 2005 and 2006 separately (different sites were sampled each year). We included potential factors influencing capture (i.e., detection) probability (p), including weather conditions: rain (RAIN, in mm) and minimal daily temperature (TEMP). Julian day (JDAY) was also considered because autumn is a dispersal season for juvenile martens. Because we sampled martens during the dispersal season, we are aware that we may violate the site closure condition (especially for juveniles) in site-occupancy models, which assumes that occupancy status at each site does not change over the surveys (MacKenzie et al. 2006). In this case, estimators may be biased and inference about factors may be incorrect. However, if individuals randomly moved in and out of a sampling site, the occupancy estimator should be unbiased, if the proportion of sites are considered "used" rather than "occupied" (MacKenzie et al. 2006). Hereafter, we refer to site use instead of site occupancy when the results of such models are reported. Additional detection covariates consisted of sampling effort (EFFORT), which is the number of active traps per site per day, and the area of each site (AREA). Correlated variables ($> |0.7|$) were never entered into the same model to avoid collinearity.

We compared 35 models (see Appendix 1.2): a null model (only the constant was included for occupancy and detection); 20 models that used habitat composition variables (including 4 global models); and 14 models that used habitat configuration variables (including 2 global models). Models were ranked based on the second-order Akaike Information Criterion (AIC_c) or $QAIC_c$ when global models suggested overdispersion. Multimodel inference was performed to obtain estimates and associated standard errors of each parameter of interest (Burnham and Anderson 2002). We also calculated predictions of the probability of site use as a function of important explanatory variables, using the delta method (calculated with all models, MacKenzie et al. 2006). The delta method provide predicted probabilities of site use for determined values of a variable of interest based on the knowledge of the variance of that variable.

1.2.6.4. Marten abundance and body index models

We analyzed the relationship between marten abundance (2 years pooled) and habitat context composition and configuration using multiple regressions at the home-range scale (same sites as for site-occupancy analyses). We related the number of captured martens and the proportion of different habitat types, using GLM (generalized linear models) with a Poisson distribution (with log-EFFORT (i.e., number of trap-nights) as an offset), together with model selection and multimodel inference. We tested the effect of the same context variables that were used for the site-occupancy analyses (%CC, %MIX, %MF, %LSF, EDGE, %CAMF, %CALSF) and included AREA as a covariate. Using the delta method (using all models), we calculated predictions of marten abundance as a function of important explanatory variables.

Multiple regressions (GLM), model selection and multimodel inference were also used to evaluate, at the home-range scale, the relationship between marten body index and the proportion of clearcuts (in the same sites), age, sex and year. We also tested the effect (and possible interaction) of the proportion of clearcuts and the landscape type for a subsample of martens from dispersed- vs. clustered-cut landscapes only, to verify whether body index was influenced by clearcut dispersal patterns.

Multinomial (or ordinal, if the condition of parallelism was respected) regression, model selection and multimodel inference were used to evaluate the relationship between marten fur color and the proportion of clearcuts (in the same sites), age, sex, body index, year and landscape type, for all martens and for the subsample of martens from the dispersed- and clustered-cut landscapes only. Dark brown fur color was chosen as the reference level. All estimates were obtained using *R 2.8.1.*

1.3. Results

1.3.1. Overall landscape patterns

The unharvested landscape was composed of a largely continuous mature forest tract that had been crossed by a large wildfire 20 years previously. Consequently, connectivity was high in this landscape (PC = 0.47). The proportion of non-habitat was low (30%), although this included large lakes (10%). Interior habitat represented 55% of the landscape.

The two dispersed-cut landscapes presented a higher probability of connectivity (D1: PC = 0.31; D2: 0.25) than the two clustered-cut landscapes (C1: PC = 0.08; C2: 0.10). Differences in connectivity between these two managed landscapes were proportionally larger than differences in the proportion of non-habitat alone, which was, respectively, 41% (D1) and 44% (D2) in dispersed-cut landscapes and 58% (C1) and 52% (C2) in clustered-cut landscapes. Interior habitat represented 45% and 39% in D1 and D2, and 29% and 30% in C1 and C2, respectively.

1.3.2. Marten abundance patterns across landscapes

One hundred fifty-six individual martens were captured over the 2 years of this study. Overall marten abundance was similar between the two autumn seasons, with 4.99 individuals/100 trap-nights in 2005 and 4.55/100 trap-nights in 2006. However, abundance in the unharvested landscape (8.73/100 trap-nights) was twice as high as in the clustered-cut (C1: 3.67/100 trap-nights; C2: 5.02) and dispersed-cut landscapes (D1: 4.51/100 trap-nights; D2: 3.93), and similar between the two harvested landscape types. Marten abundance in these landscapes was negatively related to the proportion of non-habitat in each landscape (Fig. 1.2).

1.3.3. Movement patterns and food resources

The proportion of juveniles captured was lower in 2005 (39%) than in 2006 (64%; Table 1.2). Juvenile martens moved more frequently (i.e., were recaptured in different traps) than adults in 2005, but the proportion was the same in 2006 (for similar sex-ratio; Table 1.2). Average distance traveled between two capture stations was shorter for adults than for juveniles (two-way ANOVA: age: $F_{1,25} = 6.87$, $P = 0.015$, Fig. 1.3), and travel distance was marginally higher in 2005 than in 2006 (year: $F_{1,25} = 3.14$, $P = 0.088$, Fig. 1.3). Despite a non-significant interaction between age and year ($F_{1,25} = 2.32$, $P = 0.140$), most of this response may be attributed to juveniles moving more frequently and for longer distances in 2005 than 2006 (see Fig. 1.3). In 2005, one month after the first capture event in the study area, four of the 71 martens were caught by trappers more than 100 km south of their initial location (straight line), indicating that individuals may travel long distances in short time periods (one juvenile male: 131 km in approximately 40 days; three adult females: 130 km in approximately 35 days, 1-y-old; 117 km in approximately 17 days, 3-y-old; 109 km in unknown time, 1-y-old). No such long distance movements were recorded in 2006.

Abundance of small mammals (all species pooled) was higher in 2006 than in 2005 (GLMM, year: $t_{1,34} = 6.03$, $P < 0.0001$), but did not differ significantly between clustered-cut and dispersed-cut landscapes ($t_{1,34} = 2.22$, $P = 0.16$, Fig. 1.4). This pattern was mostly due to red-backed voles, which were almost four times more abundant in 2006 than in 2005 (GLMM: year, $t_{1,34} = 4.86$, $P < 0.0001$; landscape, $t_{1,34} = 0.50$, $P = 0.67$, Fig. 1.4).

1.3.4. Marten site use according to site-occupancy models

In 2005, the null model (i.e., constant occupancy and capture (or detection) probability) ranked as one of the best models (ΔQAICc < 2) for all martens and for adults (see Appendix 1.3). The estimated c-hat value for site-occupancy models was close to 3, indicating overdispersion or lack-of-fit for these two data sets (Burnham and Anderson 2002). Consequently, no habitat context variables were important in explaining occupancy or detection for all martens and for adult martens in 2005. Although no habitat context variables came out as explanatory factors for juveniles, Julian day had a substantial effect on the probability of capture with greater detection as the trapping season progressed (model-averaged estimate ± unconditional SE: 0.086 ± 0.027). In 2005, average site use (occupancy) on the study area (C1 + D1 + UH) was over 70% for all martens (estimate ± SE: 0.80 ± 0.11) and for adults (0.71 ± 0.17), but 55% for juveniles (0.55 ± 0.10). Average probability of capture was around 20-30% (all martens, 0.28 ± 0.04; adults, 0.18 ± 0.05; juveniles, 0.21 ± 0.04).

In 2006, none of the habitat context variables had a substantial influence on site use (occupancy) by all martens or by juveniles. Adult marten site use was marginally and negatively influenced by % clearcuts (estimate ± SE: -0.061 ± 0.034) and positively by % mixedwood forests (0.109 ± 0.066). The predicted model-averaged probability of adult marten site use decreased from 86 to 67% when the proportion of clearcuts increased from 3 to 67%, and increased from 73 to 93% when the proportion of mixedwood forests increased from 0 to 41% (Fig. 1.5). Moreover, Julian day had a substantial effect on the probability of capturing an adult, with greater detection as the trapping season progressed (0.057 ± 0.015). In 2006, average site use on the study area (C2 + D2) was over 80% (all martens, 0.86 ± 0.08; adults, 0.80 ± 0.12; juveniles, 0.85 ± 0.16). Average probability of capture was, however, around 15-20% (all martens, 0.20 ± 0.02; adults, 0.13 ± 0.02; juveniles, 0.16 ± 0.03).

1.3.5. Marten abundance and body index models

Marten abundance was negatively influenced by edge density in our sites (estimate ± SE: -0.019 ± 0.007) and, to a lesser degree, by the proportion of clearcuts (-0.011 ± 0.005). Predicted model-averaged abundance decreases from 1.62 to 0.71 martens/100 trap-nights when edge density increased from 0 to 64 m/ha, but decreased only from 1.16 to 1.02 martens/100 trap-nights when the proportion of clearcuts increased from 0 to 72% (Fig. 1.6). None of the other habitat characteristics were important in explaining marten abundance.

Body index values were calculated for 153 captured martens. Body index (from all martens) was influenced by % clearcuts, age and sex, but not by year (estimate ± SE (year): -0.514 ± 0.361). Average body index was higher for males (21.10) than females (16.39) (sex: -4.677 ± 0.372), and higher for adults (20.10) than juveniles (18.74) (age: 1.099 ± 0.367). Marten body index was lower in sites where the proportion of clearcuts was higher (%CC: -0.035 ± 0.010). This effect was more pronounced for adult males (Fig. 1.7). For the subsample of 133 martens from dispersed- and clustered-cut landscapes, body index was higher in dispersed-cut (19.93) than in clustered-cut (18.50) landscape (landscape type: -1.619 ± 0.385). Body index was also influenced by age (1.040 ± 0.349) and sex (-4.632 ± 0.351), but not by %CC (-0.021 ± 0.011; Fig. 1.7) for this subsample.

Fur color (from all martens, using multinomial regressions) was only influenced by age (reddish-brown, 1.30 ± 0.57; pale, 1.45 ± 0.57). Dark-brown fur was displayed more frequently by juveniles (79%) than was reddish-brown and pale fur colors (50% and 46%, respectively). For the subsample of martens from dispersed and clustered cutting landscapes, fur color was influenced substantially by %CC (0.02 ± 0.01) and marginally by age (0.59 ± 0.34) (using ordinal regressions). Pale martens were captured in sites with a higher proportion of clearcuts (36%) than were reddish-brown and dark brown martens (29% and 28%, respectively). Dark brown martens were also more frequent in dispersed-cut (24%) than in clustered-cut landscapes (13%), and conversely, pale martens were more frequent in clustered-cut (48%) than in dispersed-cut landscapes (37%); this difference was not statistically important (landscape: 0.46 ± 0.37).

1.4. Discussion

1.4.1. Marten broad abundance patterns across landscapes

Our prediction of abundance patterns of martens across landscapes was met for the unharvested landscape where marten abundance was twice as high as that in managed landscapes. The low proportion of clearcuts, the high connectivity of suitable habitat, and the higher proportion of interior habitat in the unharvested landscape are all concurrent factors that may explain this pattern. In managed landscapes, however, our prediction did not hold and landscapes harvested with both cutting patterns (clustered and dispersed) harbored similar abundances of martens. Given that martens are positively associated with the proportion of interior habitat (Potvin et al. 2000), we expected that clustered-cut landscapes with mainly linear remnant forest (corridors and riparian buffers) would be less frequently used by martens because of their low proportion of interior habitat. Our clustered-cut landscapes had indeed less interior habitat and the highest proportion of clearcuts across our compared landscapes. Moreover, suitable habitats were three times less connected in the clustered- than in the dispersed-cut landscapes. Hence, our results suggest that, at this level of habitat retention (41-57%), within which interior habitat constitutes 30 to 55% of the landscape, marten broad abundance was not influenced by the configuration (blocks vs. linear strips) of the remaining late-seral habitat.

Strong differences in abundance patterns of marten between unharvested and harvested landscapes and a concomitant lack of differences between harvested landscapes may also be linked to the nature of the resulting matrix and time since disturbance. In the unharvested landscape, the wildfire occurred 20 years previously and the resulting area was composed of regeneration that had reached 3 m in height. Such environments could provide suitable protection cover against predators for marten and good habitat for prey, such as snowshoe hare and small mammals (Paragi et al. 1996, Fisher and Wilkinson 2005, Jacqmain et al. 2007). In addition, natural disturbances are also known to leave legacy trees (Franklin et al. 2000) that generate habitat attributes and structure favorable for the return of marten (see Chapin et al. 1997, Potvin et al. 2000). In contrast, regeneration in most (91%) of the cut-blocks of both harvested landscape types (clustered vs. dispersed) in our study area were between 0- and 11-years-old and were < 1 m in height, which did not provide sufficient protection cover for martens (Potvin et al. 2000) and their prey (Fisher and Wilkinson 2005). Moreover, despite an absence of replication for the unharvested landscapes, our abundance results are better explained by the proportion of non-habitat for marten, rather than the proportion of mature forests, which are often considered as optimal habitats.

1.4.2. Marten response to habitat context composition and configuration variables

At the home-range scale, martens were tolerant of an increased proportion of clearcuts, whereas we found weak relationships between habitat loss (increased proportion of clearcuts) and 1) marten site use (only for adults), 2) marten abundance, and 3) marten body condition (mainly for adult males). Hence, > 70% of the studied sites were used by martens, regardless of contrasting differences in marten movements and prey availability between the 2 years of the study (discussed later). Thus, overall marten site use (adults and juveniles pooled) was not influenced by the surrounding proportion of clearcuts. Juvenile site use also was not influenced by surrounding habitat characteristics. This may not be surprising given the greater propensity for juveniles to disperse compared to adults during the autumn (Johnson 2008), i.e., at the time we conducted this study. Indeed, high rates of movement were observed for transient individuals searching for a place to settle. Moreover, during autumn dispersal, juveniles also possibly use marginal or low quality habitats. Buskirk et al. (1989) found that young martens use a broader range of habitats than do older individuals. However, when considering adults only, we found a marginally negative relationship between the probability of site use by an adult marten and the surrounding proportion of clearcuts. Adult martens were more likely to be residents, and thus, were more prone to being influenced by habitat loss.

Although the relationship between marten abundance and the proportion of clearcuts was statistically significant, its biological significance was weak (as shown in Fig. 1.6). Even if marten abundance and probability of site use were only weakly influenced by the proportion of clearcuts, marten abundance was more strongly affected by the density of edges incurred by timber harvesting. As martens do not venture into open areas (Buskirk and Powell 1994), abrupt forest/clearcut edges represent boundaries in the landscape, which act as barriers to movement (Cheveau et al. chapter 3). Hargis et al. (1999) also found that marten capture rates were negatively correlated with increasing extent of high-contrast edges.

Marten body condition was negatively affected by the proportion of clearcuts. Moreover, body condition was affected by the dispersion pattern of clearcuts. Martens were in better condition in dispersed-cut than in clustered-cut landscapes. Males, and particularly adults, were more affected by habitat loss and its configuration, possibly because they had a larger home range, and consequently, moved more frequently than females (Powell 1994). However, low body condition in females could also lead to reproductive failure during blastocyst implantation (delayed pregnancy) and spontaneous abortion, as has been shown in other mustelids (weasels and stoats; Tapper 1979, Erlinge 1981). For an animal that usually has < 5% body fat, such as the marten (Robitaille and Cobb 2003), poor body condition could have important consequences for short-term survival. Indeed, martens are very lean for their body size. Moreover, Buskirk and Harlow (1989) estimated that a marten would metabolize all its fat reserves in 92 hours, if fat was used as the sole metabolic substrate and without any other external inputs. Accordingly, our results suggest that increasing clearcut areas and their concentration decreased body condition. In turn, decreased body condition could potentially affect populations through increased mortality of males and lower productivity of females, which were not measured in the current study.

Fur color was influenced by marten age, with younger martens being dark-brown more frequently than older individuals. Moreover, fur color was affected by the proportion of clearcuts (when considering the two harvested landscape types only), where sites with more clearcuts contained paler martens. Cree trappers have already noticed that marten pelts were paler in highly harvested landscapes (P. Dixon, Cree Trapper Association, *pers. comm.*). Despite a small difference in the proportion of the different fur colors between landscapes, the livelihoods of Cree trappers can be adversely affected when they proceed to sell the pelts, since paler martens have a lower value.

The positive relationship between adult occurrence and the amount of mixedwood forest corroborates the findings of Potvin et al. (2000), who worked in an adjacent study area, regarding marten preferences for mixedwood over conifer stands (notwithstanding their age). The importance of mixedwood stands in this largely conifer-dominated matrix is quite surprising considering their rarity at the landscape scale (< 10%), but this finding was confirmed by a companion study on marten habitat selection in our study area (Cheveau et al. chapter 2). This result suggests that mixedwood stands are possibly a key habitat for marten populations in the region. Consequently, retention and conservation of mixedwood stands in a conifer matrix could play a critical role for marten populations in a harvested landscape.

Jacqmain et al. (2008) also found that these stands are critical habitat for moose (*Alces alces*) in the same study area. Cree land users already have recognized the importance of mixedwood stands for wildlife in the region and have asked for specific management guidelines for these stands (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002).

1.4.3. Movement patterns and food resources

Even though martens are opportunistic predators that can forage on a large diversity of prey (Thompson and Colgan 1990), small mammals and snowshoe hares are the most important resources in the marten winter diet in eastern North America (Martin 1994). The abundance of at least one species of small mammal, the red-backed vole, is cyclic in boreal forests of eastern North America (Cheveau et al. 2004). Red-backed vole was the most abundant rodent species in our study area. With a 4-year cycle, red-backed voles (and to a lesser degree, all other species) were rare in 2005 and reached their peak phase in 2006. Similarly, snowshoe hare populations (*Lepus americanus*) also exhibit cyclical dynamics, on the order of 10 years in the northern part of North America (Krebs et al. 2001). In our region, hares were at similar but low abundance during the 2 years of investigation (Assels et al. 2007). Therefore, in 2005, our study was conducted during a year of important food shortage, with simultaneous low densities of both small mammals and snowshoe hares. Thompson and Colgan (1990) found that, during such years, marten increased the proportion of alternative prey, such as squirrels, grouse and berries in their diet. Furthermore, Thompson and Colgan (1987) and Poole and Graf (1996) found that declines in preferred prey affected marten density, age ratio, productivity, and size of home ranges. In our study, we observed that both the frequency and length of marten movements increased in 2005, for juvenile martens as well as for adults, which usually are resident. This high rate of movement, which may be linked to a year of food shortage in our study area, likely explains the higher detection probability observed during the first year of sampling. We also recorded exceptional long-distance dispersal movements in adults (> 100 km), which was also noted by Thompson and Colgan (1987) in cases of reduced prey availability. Hence, our results suggest that, during the 2005 food shortage year, marten were less selective in habitat use and occurred in a wider range of habitats, possibly to increase their chance of encountering prey. This, in turn, may explain the lack of relationships between marten occurrence and habitat context variables in our models. Theoretically, such synchronism in small mammal and snowshoe hare low densities may occur every 20 years. However, in 2006, martens could forage on abundant small mammals, showing no long-distance movements for either adults or juveniles. Consequently, this second sampling year was not affected by high rates of movement as was observed in the 2005 season, thereby allowing us to make inferences regarding habitat context variables. Indeed, some context variables appeared as significant explanatory variables (%CC, %MIX) in our adult marten site-occupancy models. Despite differences in food availability, we found that marten body condition was not different between the 2 years of investigation, suggesting that behavioural adaptations (movement increase and habitat use change) could compensate for food shortage.

1.4.4. Effects of habitat thresholds on marten populations

Although we sampled a wide gradient of habitat loss (0 to 72% clearcuts) at the home-range scale (within 5-10 km^2), we found only a weak response of marten populations. Within this range, the decline in marten abundance and adult occurrence was almost linear with a smooth slope. In a meta-analysis examining habitat fragmentation, Andrén (1994) proposed a theoretical threshold where fragmentation effects (isolation) are added to habitat loss effects at around 20-40% of residual habitat, but he considered that such habitat thresholds should be determined specifically for each species. In contrast, Fahrig (2003) suggests that, if such a threshold exists, it should result in a statistical interaction effect between the amount of residual habitat and fragmentation *per se*; but evidence of such an interaction has yet to be identified empirically. Marten population responses did not reach any threshold within a range of 0 to ~70% of habitat loss in our study area. Rather, predicted adult marten occurrence (Fig. 1.5) showed a 22% decrease within that range, which is far less drastic than a one-to-one linear response. This result is contrary to what has been previously observed by Bissonette et al. (1997) for Maine and Utah marten populations. Our results do not corroborate previous studies on marten, which have suggested a strong negative response near 30-40% habitat loss (inside their home range, Chapin et al. 1998, Potvin et al. 2000). Yet, Fuller (2006) found that martens in Newfoundland could tolerate high levels of habitat loss (60-70%). Hence, marten could be less tolerant to habitat loss at its southern limit of distribution (as in Utah, Hargis et al. 1999) than in the boreal forest. However, the decrease in marten abundance or site use could be less than an 1:1 response if sampled residual patches were all larger than the minimal area requirement for marten establishment.

Another explanation for this tolerance to habitat loss could be related to the fact that managed landscapes in this portion of the boreal forest are embedded in an unharvested matrix, which may provide source habitats for marten populations that function as a "source-sink" metapopulation (*sensu* Pulliam 1988). Persistence of martens in habitat remnants of managed forests could be linked to their dispersal abilities, allowing them to move from continuous forest tracks of the unharvested matrix to habitat remnants (linear strips and patches) in harvested landscapes. Understanding marten population dynamics with respect to connectivity between existing unharvested forests and remnant habitats requires data on productivity of females and dispersal movements. This dynamic could, however, be disrupted if current forestry practices extend throughout the commercial timberline without consideration for conservation of large tracks of continuous late-seral forest.

1.4.5. Clearcut dispersion patterns and marten persistence

Our two scales of investigation correspond to concrete forest management scales included in the aforementioned formal agreement between the Government of Quebec and the Crees of Quebec (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002). Indeed, our landscape scale (173 to 480 km^2) could be compared to family hunting territories (several hundreds of square kilometres), which represent the management unit. Despite our limited sample size (N = 5 landscapes), our results showed that marten populations were maintained within landscapes that retain ~50% mature forests, although populations were at lower densities compared to unharvested areas. Currently, the agreement allows for a maximum of 40% in regenerating areas (< 20-years-old) within each hunting territory, which appears to be adequate to maintain marten.

At a smaller scale, clearcut dispersion patterns are determined within an “intervention sector,” ranging from 10 to 200 km^2 in the black spruce boreal forest. Our results seem to indicate that the “mosaic cutting” strategy (dispersed clearcuts) promoted by the agreement did not provide an advantage for marten populations when considering their abundance levels and site use, but this pattern maintained martens in better physical condition than in clustered-cut landscapes.

1.5. Conclusion

We found that American marten was sensitive to forest harvesting in the boreal forest of eastern Canada. Indeed, abundance was halved in harvested landscapes compared to the unharvested landscape, even if the proportion of mature forests was the same in all landscapes. However, marten abundance was similar in all harvested landscapes despite that the proportion in clearcuts ranged from 21 to 45% (and non-habitat ranged from 43 to 59%), at a scale of several hundreds of square kilometres. Hence, we could not discern whether or not dispersed clearcuts could provide an advantage over clustered clearcuts in terms of marten abundance at the landscape scale, even if the amount of interior habitat was higher in the dispersed-cut landscapes. Moreover, marten did not respond to the configuration of residual habitat and only modestly to its amount when the level of habitat retention was ~50% in the surrounding landscape. It seems that marten was more influenced by its environment at the landscape scale than at smaller scales, which could be explained by its high observed movements. Hence, at the finer scale (5-10 km^2), we found that marten site use and abundance in residual forests was not strongly influenced by the amount of habitat loss in the surrounding context. While such results suggest that marten may be more tolerant to habitat loss than previously assumed, martens from the dispersed-cut landscapes exhibited better physical condition than martens from clustered-cut landscapes, especially males, showing that negative effects are first detected at an individual- rather than at a population-level. Although we failed to detect a clear threshold in marten response even when nearly

70% of local habitat was lost, we acknowledge that we did not investigate marten response to habitat loss beyond a 70% level at the home-range scale (5-10 km^2) and beyond ~50% at the landscape scale. Because habitat loss and fragmentation created by forest harvesting is temporary, we also recognize that our results represent only one point in time and could not be considered as long-term conclusions on marten persistence. Furthermore, we suggest that future studies should verify whether this tolerance could also be detected when individuals are settled in their home range (for example, during the winter) and if their habitat selection in the region reflects this tolerance. Moreover, marten productivity (demographic data) could also be investigated to evaluate long-term persistence of a viable population in these managed landscapes.

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Table 1.1. General composition of the five studied landscapes (% of the total area) as well as the local proportion of clearcuts within sites: C1 and C2 are clustered-cut landscapes, D1 and D2 are dispersed-cut landscapes and UH is the unharvested landscape.

	Mature forests ≥ 7 m				Regenerating stands from				Clearcuts (range)
	Overall	Late-seral forests[§]	Mixed-wood forests	Forested bogs	Clearcut	Fire	Water	Non-forested	Within sites[†]
C1	42%	15%	2%	8%	45%	-	1%	4%	18-72%
C2	48%	10%	10%	10%	29%	-	9%	4%	3-67%
D1	57%	39%	2%	8%	25%	2%	4%	4%	17-52%
D2	52%	23%	4%	14%	21%	4%	7%	2%	11-41%
UH	50%	14%	4%	7%	9%	20%	10%	4%	0-37%

§ coniferous forests more than 120 years (estimated age)

[†] 4.8 to 12.5 km^2, including 2 to 7 capture stations

Table 1.2. Frequencies (and percentage) of recaptures for adult (A) and juvenile (J) martens in 2005 and 2006.

Year	Age	Recaptured in the same trap	Recaptured in a different trap	Captured one time	Total	P (Chi-square)	df
2005	A	7 (17%)	5 (12%)	30 (71%)	42	0.03	2
	J	5 (19%)	10 (37%)	12 (44%)	27		
2006	A	4 (13%)	4 (13%)	23 (74%)	31	0.79	2
	J	7 (13%)	10 (19%)	37 (68%)	54		

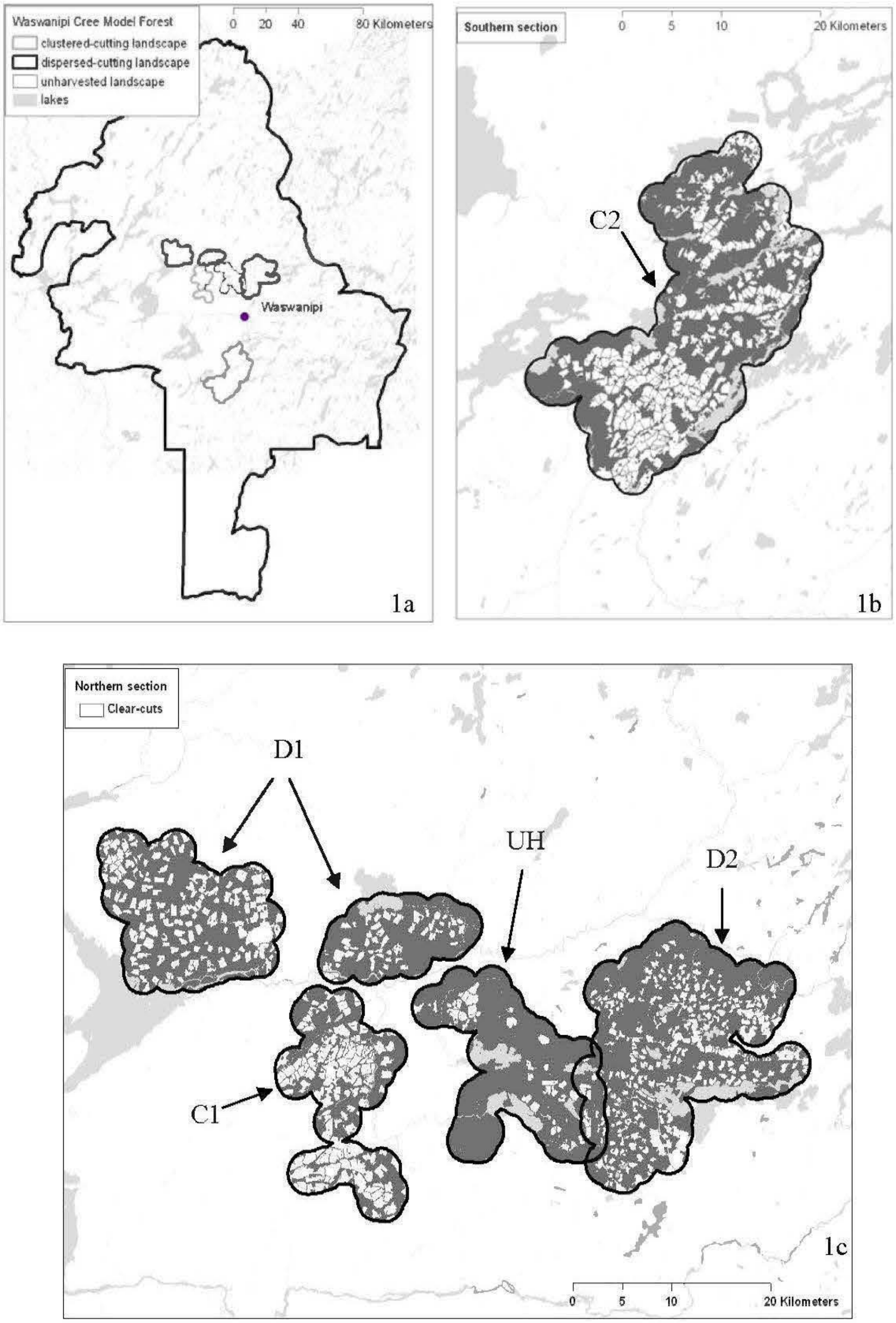

Figure 1.1. Global (1a), northern (1b), and southern (1c) study areas, where clearcuts (white) and unharvested habitat (grey) are shown. Landscapes were delineated by a 2-km buffer around capture stations. C1 (173 km²) and C2 (480 km²) were clustered-cut landscapes, D1 (314 km²) and D2 (367 km²) were dispersed-cut landscapes, and UH (213 km²) was the unharvested landscape.

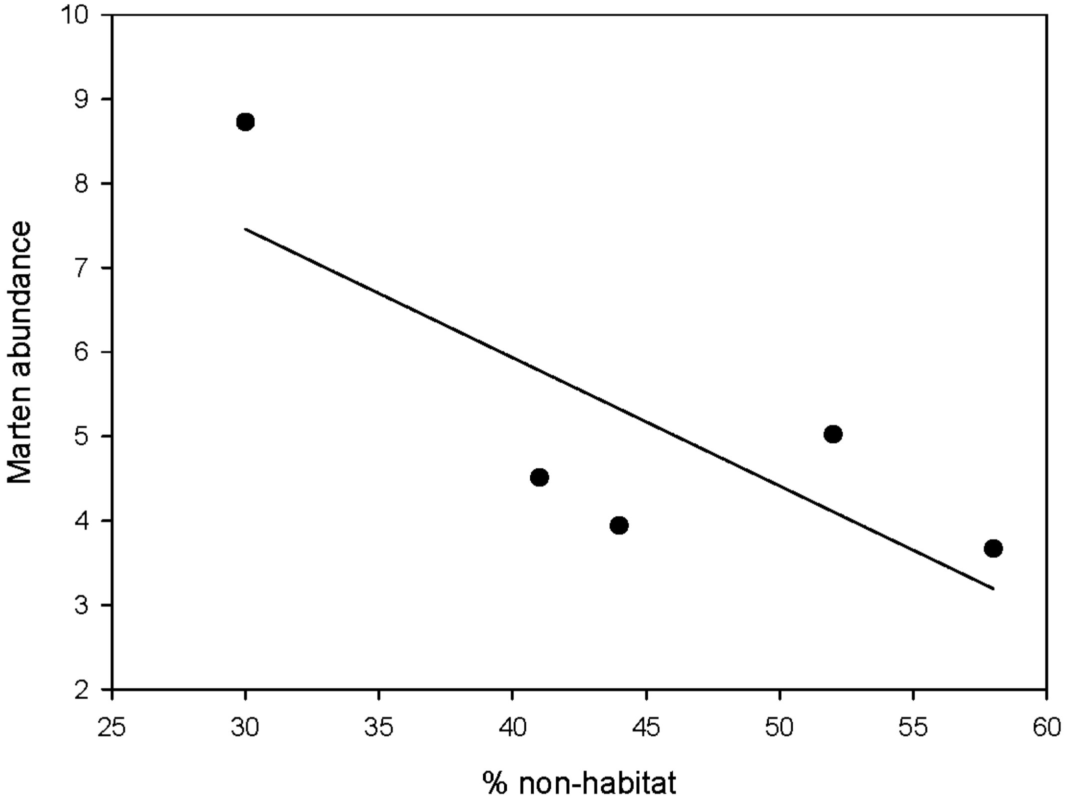

Figure 1.2. Overall marten abundance (captures/100 trap-nights) in our five landscapes as a function of the proportion of non-habitat (clearcuts, bogs and non-forested areas). Linear regression is shown ($R^2 = 0.63$).

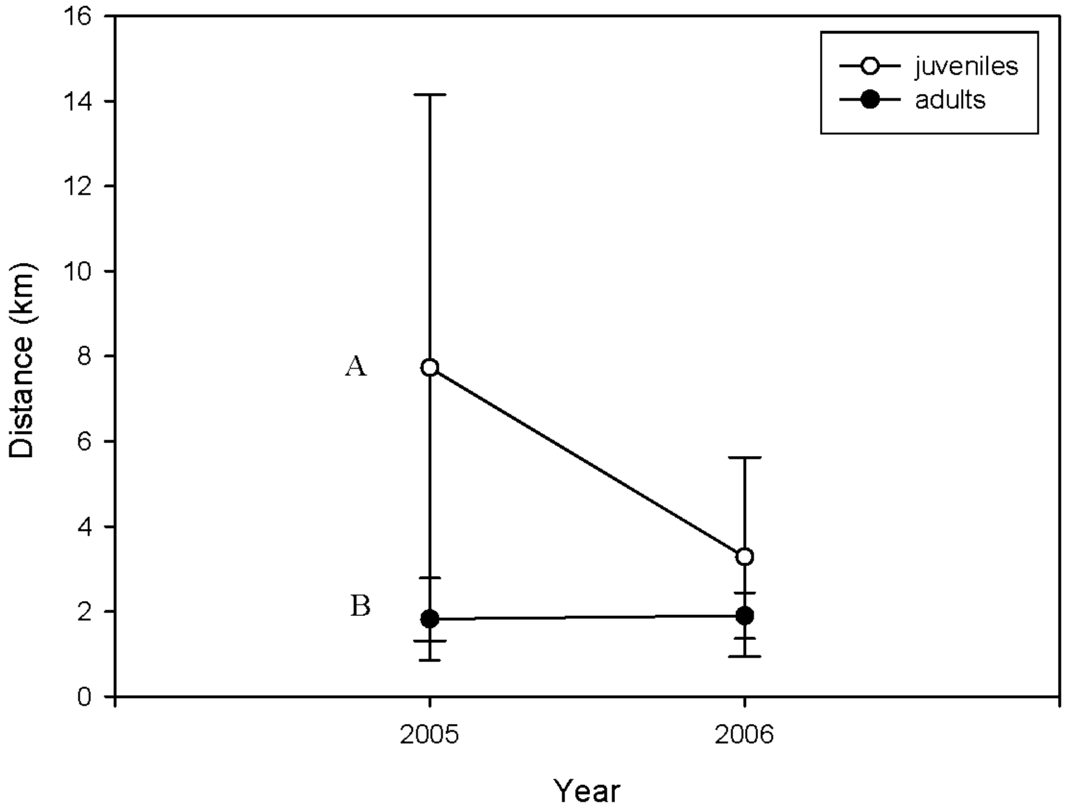

Figure 1.3. Mean distance (± S.D.) between two capture points traveled by juvenile (N = 10 in 2005 and N = 10 in 2006) and adult (N = 5 in 2005, and N = 4 in 2006) martens that were captured more than once. Different letters indicate an overall significant difference between juveniles and adults (two-way ANOVA, $p < 0.05$).

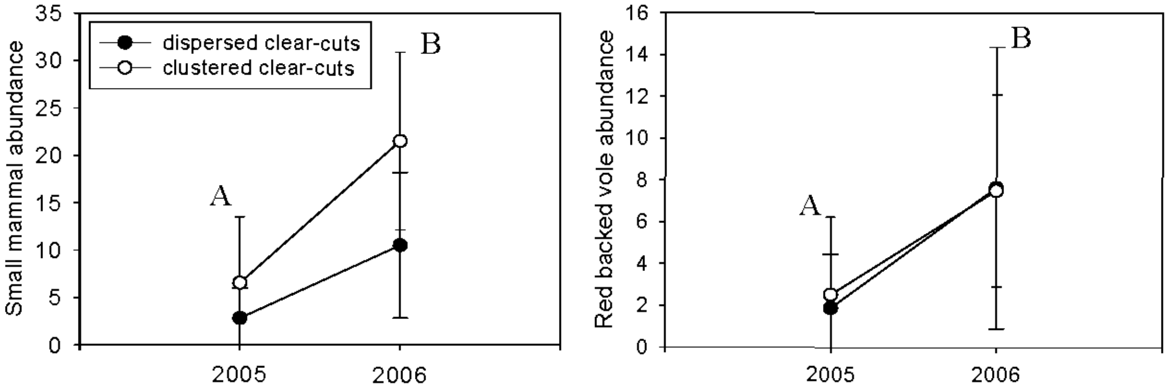

Figure 1.4. Abundance (captures/100 trap-nights ± S.D.) of small mammals (all species pooled and red-backed voles) during two summers. Results are shown for clustered-cut and dispersed-cut landscapes. Different letters indicate significant differences between years (two-way ANOVA, $p < 0.05$). There was no significant difference between landscape types.

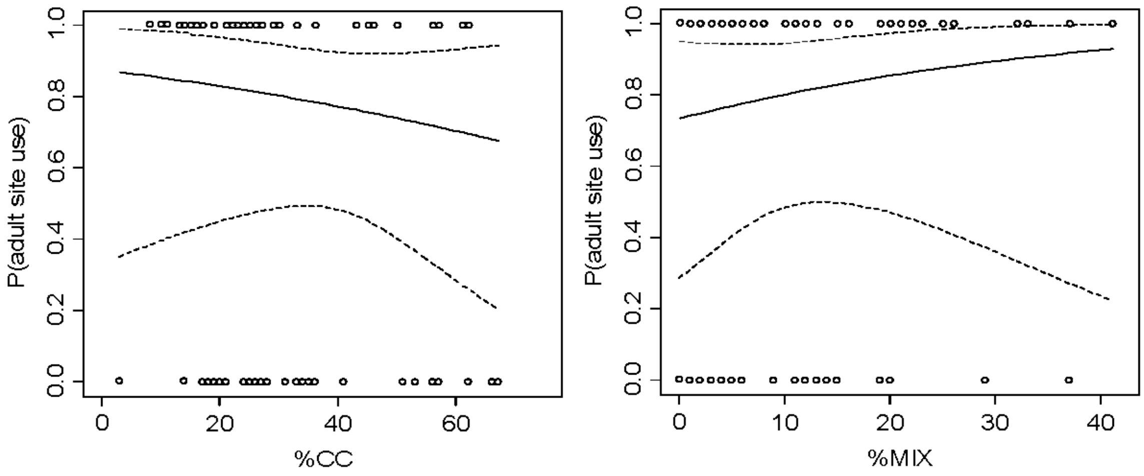

Figure 1.5. Predicted probability of adult marten site use (based on site-occupancy models) as a function of the surrounding proportions of clearcuts (%CC) and mixedwood forests (%MIX) corresponding to a home-range scale (delta method: predicted model averaged estimate with a 95% confidence interval), in 2006. Presence (1)/absence (0) status is shown for each site (N = 143), depending on the proportions of clearcuts (%CC) and mixed wood forests (%MIX).

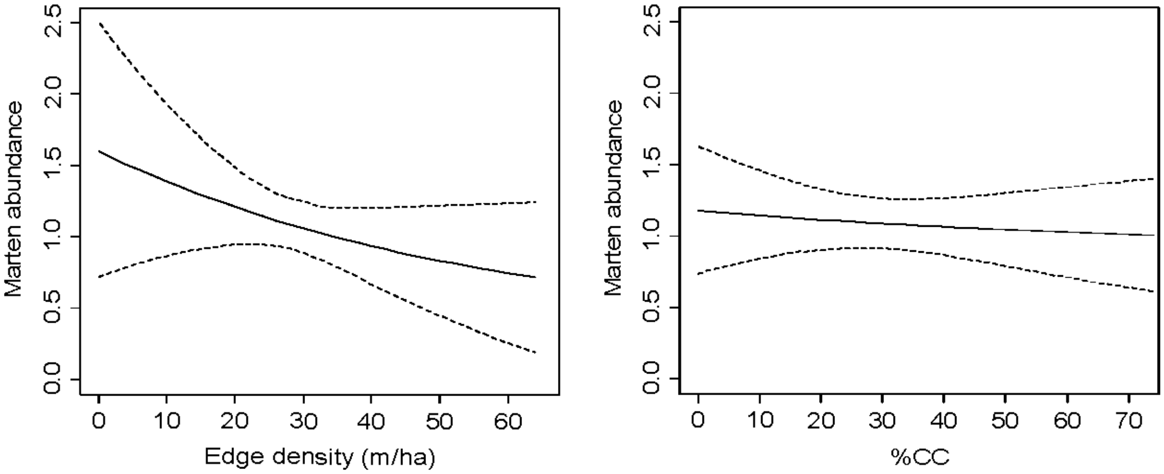

Figure 1.6. Predicted marten abundance (martens/100 trap-nights) as a function of edge density as well as the proportion of clearcuts (%CC) corresponding to a home-range scale (delta method: predicted model averaged estimate with a 95% confidence interval).

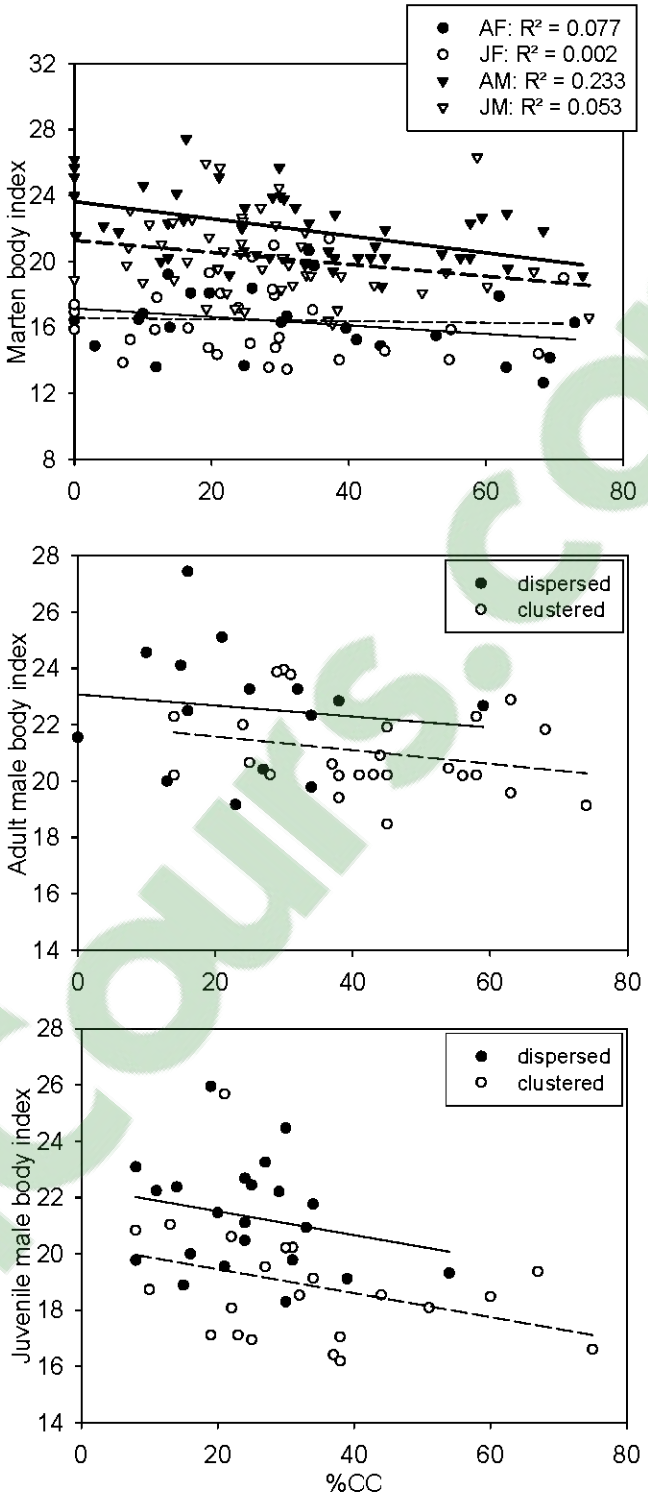

Figure 1.7. Marten body index depending on % clearcuts within sites a) for all martens (N = 153), b) for adult males and c) for juvenile males. In graph a, bold lines are for males (M) and thin lines are for females (F), while solid lines are for adults (A) and dashed lines for juveniles (J). In graphs b and c, solid lines are for dispersed-cut landscapes and dashed lines for clustered-cut landscapes.

Appendix 1.1. a- Forest types (%); b- age classes (%) of residual forest stands; and c- density classes (%) of intact and residual forest stands, in the five landscapes (D: dispersed-cut, UH: unharvested and C: clustered-cut landscapes).

a-

Forest types	D1	D2	UH	C1	C2
Conifer					
Black spruce dominated	93.4	88.7	89.1	89.7	69.2
- *EE*	*88.3*	*66.9*	*82.2*	*72.5*	*51.4*
- *EPG*	*2.2*	*14.1*	*2.6*	*12.1*	*10.6*
Jack pine dominated	2.1	6.3	1.4	3.9	8.2
Tamarack dominated	0	0.1	0	0.1	0.1
Balsam fir dominated	0.9	0.2	0.5	0	3.5
Other conifer dominated	1.6	0.4	3.4	2.2	1.4
Mixedwood and deciduous					
Trembling aspen dominated	0.3	0.7	1.3	2.4	14.2
Paper birch dominated	0.5	3.3	3.2	0.1	2.0
Intolerant deciduous dominated	0	0	0.1	0	0.1
Undetermined mixedwood	0.4	0.1	0.3	0.5	0
Others	0.7	0.2	0.8	1.1	1.3

b-

Age class (years)	D1	D2	UH	C1	C2
10	0.1	0.1	1.3	1.9	0.7
30	1.7	0.1	1.2	0.9	2.8
50	2.9	0.4	3.2	2.7	1.3
70	17.0	12.9	45.0	35.7	50.2
90	9.9	40.9	21.4	21.9	23.2
120	65.0	37.2	23.5	32.8	20.7
YUA*	0.3	0.8	0.4	0.9	0.3
OUA*	2.9	7.4	3.9	3.0	0.8

* YUA: young uneven-aged, OUA: old uneven-aged

c-

Density class	D1	D2	UH	C1	C2
>80% (A)	3.9	1.2	2.4	1.9	23.5
60-80% (B)	21.5	26.2	22.9	41.3	33.1
40-60% (C)	44.2	51.7	50.5	36.9	32.4
25-40% (D)	30.4	20.9	24.1	19.9	10.9

Appendix 1.2. Models tested for site-occupancy analyses including variables that influence site use (ψ) and detection (p).

Model	ψ	p
Null	-	-
<u>Context composition</u>		
Global:	%CC, %LSF, %MIX	EFFORT, AREA, JDAY
	%MF, %LSF, %MIX	EFFORT, AREA, JDAY
	%CC, %LSF, %MIX	EFFORT, AREA, RAIN, TEMP
	%MF, %LSF, %MIX	EFFORT, AREA, RAIN TEMP
Individual:	%CC	-
	%LSF	-
	%MF	-
	%MIX	-
	%CC	EFFORT, AREA
	%LSF	EFFORT, AREA
	%MF	EFFORT, AREA
	%MIX	EFFORT, AREA
	%CC	EFFORT, AREA, JDAY
	%LSF	EFFORT, AREA, JDAY
	%MF	EFFORT, AREA, JDAY
	%MIX	EFFORT, AREA, JDAY
	%CC	EFFORT, AREA, RAIN, TEMP
	%LSF	EFFORT, AREA, RAIN, TEMP
	%MF	EFFORT, AREA, RAIN, TEMP
	%MIX	EFFORT, AREA, RAIN, TEMP
<u>Context configuration</u>		
Global:	%CAMF, %CALSF, EDGE	EFFORT, AREA, JDAY
	%CAMF, %CALSF, EDGE	EFFORT, AREA, RAIN, TEMP
Individual:	%CAMF	-
	%CALSF	-
	EDGE	-
	%CAMF	EFFORT, AREA
	%CALSF	EFFORT, AREA
	EDGE	EFFORT, AREA
	%CAMF	EFFORT, AREA, JDAY
	%CALSF	EFFORT, AREA, JDAY
	EDGE	EFFORT, AREA, JDAY
	%CAMF	EFFORT, AREA, RAIN, TEMP
	%CALSF	EFFORT, AREA, RAIN, TEMP
	EDGE	EFFORT, AREA, RAIN, TEMP

* - indicates that no variable is included, only the constant.

Appendix 1.3. Highest-ranked occupancy models for marten in fragmented landscapes. Models were ranked by ΔAIC_c (only those < 2 are shown). Occupancy (ψ) and detection (p) variables are indicated.

	Model structure			
Data set	ψ	p	ΔAIC_c*	Akaike weight
2005	- **	- **	0.00	0.19
	%MIX	-	1.77	0.08
	%LSF	-	1.85	0.07
	%CALSF	-	1.87	0.07
2005 adults	-	-	0.00	0.19
2005 juveniles	Edge	EFFORT, AREA, JDAY	0.00	0.15
	%MIX	EFFORT, AREA, JDAY	0.10	0.14
	%CC	EFFORT, AREA, JDAY	0.12	0.14
	%MF	EFFORT, AREA, JDAY	0.37	0.12
	%CAMF	EFFORT, AREA, JDAY	0.42	0.12
	%LSF	EFFORT, AREA, JDAY	0.99	0.09
	%CALSF	EFFORT, AREA, JDAY	1.15	0.08
2006	%CC, %LSF, %MIX	EFFORT, AREA, JDAY	0.00	0.47
2006 adults	%MIX	EFFORT, AREA, JDAY	0.00	0.32
	%CC	EFFORT, AREA, JDAY	0.15	0.29
	EDGE	EFFORT, AREA, JDAY	1.22	0.17
2006 juveniles	%MIX	EFFORT, AREA	0.00	0.32

* QAICc for the two first data sets (2005 and 2005 adults) and AICc for all other data sets.

** - indicates that no variable is included, only the constant.

CHAPITRE II

MARTEN SPACE USE AND HABITAT SELECTION IN MANAGED CONIFEROUS BOREAL FORESTS OF EASTERN CANADA

Marianne Cheveau, Louis Imbeau, Pierre Drapeau and Louis Bélanger

Résumé

Les effets de la perte et de la fragmentation de l'habitat sur le comportement individuel des organismes fauniques peuvent avoir des conséquences directes sur la viabilité des populations dans les paysages forestiers altérés. La martre d'Amérique (*Martes americana*) est une des espèces spécialistes de la forêt parmi les plus sensibles aux perturbations d'origine anthropique. Des études ont montré que la martre ne peut tolérer plus de 30 à 40% de coupes à l'intérieur de leur domaine vital. Cependant, dans une étude conjointe, nous avons trouvé que les martres étaient plus tolérantes à la perte et à la fragmentation de l'habitat, à l'échelle du paysage, en forêt boréale de l'est du Canada que ce à quoi on s'attendait. Nous avons testé si cette plus grande tolérance se reflétait aussi au niveau des domaines vitaux des individus, en analysant l'utilisation de l'espace par les martres (superficie des domaines vitaux et degré de superposition) ainsi que leur sélection de l'habitat dans des paysages fragmentés par les coupes forestières. Nous avons installé des colliers émetteurs sur des martres femelles et délimité 20 domaines vitaux hivernaux (polygones minimums convexes 100% et kernels 60%) dans des paysages de coupes dispersées (N = 8) et agglomérées (N = 12). La superficie des domaines vitaux n'était pas reliée à la proportion de coupes (c'est-à-dire à la perte d'habitat), mais plutôt à la proportion de forêts mixtes, qui représentent des habitats de haute qualité. À l'échelle des domaines vitaux, nous avons déterminé à l'aide de fonctions de sélection de ressources que les habitats les plus utilisés étaient les forêts mixtes et que les moins utilisés étaient les coupes récentes et les tourbières. À l'échelle du paysage, les domaines vitaux étaient composés de plus de forêts mixtes que les polygones aléatoires, et les zones d'activité plus intense des martres étaient composées de plus de forêts résiduelles et de moins de milieux perturbés par l'homme, tels que les coupes, les lisières et les routes. Ces résultats suggèrent que les forêts mixtes, qui n'occupent pourtant que ~10% de l'aire d'étude, jouent un rôle critique pour les martres dans les paysages boréaux dominés par les forêts de conifères. Nous recommandons donc que ces peuplements isolés soient conservés, d'autant plus qu'ils ne sont actuellement pas recherchés par l'industrie forestière. Nous suggérons que l'exploitation forestière, qui cible plutôt les peuplements résineux, serait moins dommageable pour les martres, car elles n'affectent pas les habitats les plus importants. Ceci pourrait donc expliquer cette plus grande tolérance des martres à la perte et à la fragmentation de l'habitat en forêt boréale de l'est du Canada.

Mots-clés

Sélection des ressources, martre d'Amérique, aménagement forestier, forêts mixtes, utilisation de l'espace, fragmentation

Abstract

Effects of habitat loss and fragmentation on individual behaviour of organisms may have direct consequences on population viability in altered forest ecosystems. American marten (*Martes americana*) is a forest specialist considered as one of the most sensitive species to human-induced disturbances. Some studies have shown that marten cannot tolerate more than 30-40% clearcuts within their home range. In a companion study we found, however, that martens were more tolerant than expected to habitat loss and fragmentation at the landscape scale in the eastern boreal forest of Canada. We tested whether this greater tolerance is also reflected at the level of individual home ranges, by investigating marten space use (home range size and overlap) and habitat selection in landscapes fragmented by timber harvesting. We installed radio-collars on female martens and determined 20 winter home ranges (100% minimum convex polygons and 60% kernels) in a dispersed-cut (N = 8) and a clustered-cut (N = 12) landscapes. Home range size was not related to the proportion of clearcuts (i.e., habitat loss), but rather to the proportion of mixedwood stands, which represent high quality habitat. At the home range scale, we determined, using resource selection functions, that the most used habitats were mixedwood forests and the least used were recent clearcuts and forested bogs. At the landscape scale, home ranges were composed by more mixedwood forests that random polygons and marten high activity zones were composed by more residual forest and less human disturbances like clearcuts, edges and roads. These results suggest that mixedwood forests, which occupy ~10% of the study area, play a critical role for marten in this conifer dominated boreal landscape. We thus recommend the retention of these isolated stands, which were not the focus of timber harvesting. We suggest that forest harvesting, that focuses on conifer stands, could be less harmful as long as they do not affect the most important habitat for marten. This could explain greater tolerance of marten to habitat loss and fragmentation in the boreal forest of eastern Canada.

Key words

Resource selection, American marten, forest management, mixedwood forests, space use, fragmentation

2.1. Introduction

Habitat loss and fragmentation are widely seen as a major threat for wildlife populations (Wiens et al. 1993, Fahrig and Merriam 1994). Nevertheless, to understand effects of fragmentation at the population level (Wiens 1995, Fahrig 2003), proximate mechanisms underlying population responses need to be understood first at the level of individuals (Ims 1995, Andreassen et al. 1998). Indeed, fine-scale effects like changes in individual behaviour (for example in space use, Ims et al. 1993) may have consequences at higher scales. Previous studies have been conducted mostly on birds (but see Bowman and Fahrig 2002) and have shown that habitat loss and fragmentation could induce an increase in individual movement costs (Bélisle et al. 2001, Gobeil and Villard 2002) due to constrained movements (Bélisle and Desrochers 2002), an increase in predation risk, and a decrease in foraging efficiency (Turcotte and Desrochers 2003). Hence, this may reduce reproductive success of individuals and in turn influence population viability (Hinsley et al. 1999, Hinsley 2000).

Ims et al. (1993) predicted three potential types of effects on individual space use in response to habitat loss and fragmentation that are usually based on modifications of home range size and overlap between individuals. The first type of response may be expected only for "social" species that could tolerate high home range overlap and high population densities. In this case, a fusion response is possible and implies an increase in home range overlap which could be associated with a decrease in home range size as habitat becomes more fragmented. This response was found, for example, in vole populations (Wolff et al. 1997). When habitat fragment size approaches the minimum individual requirement, a second type of response known as a fission response can be observed, which is characterized by less overlap between individual home ranges, eventually associated with a decrease in home range size. Indeed, individuals in this case protect a smaller territory but more intensively to fulfill their needs. This type of response has been shown in some species of birds and mammals, such as the northern saw-whet owl (Hinam and St Clair 2008) and European red squirrel (Wauters et al. 1994). If fragment size becomes less than individual minimum habitat requirements, an expansion response could be induced. Individuals will expand their home range to include more suitable habitats and eventually several fragments. This scenario implies an increase in movements between fragments depending on their connectivity (distance between patches, matrix hostility and presence of corridors). This response, which seems the most common, was found for different species of birds (Gjerde and Wegge 1989, Carey et al. 1992, Redpath 1995, Siffczyk et al. 2003) and mammals (Selonen et al. 2001, Riley et al. 2003, Irwin 2008).

American marten (*Martes americana*) is considered to be one of the most sensitive species to disturbances induced by timber harvesting (Thompson 1991). It could thus be a good candidate for examining the effects of habitat loss and fragmentation on individual space use. Marten is a predator but also a prey to both terrestrial and avian predators (Hodgman et al. 1997). This species is clearly a forest specialist (Buskirk and Powell 1994) that avoids open areas (Koehler and Hornocker 1977, Hargis and McCullough 1984, Drew 1995). However, across its geographic range, American martens are now recognized to use a wide variety of forest habitats, contrary to the previous views where marten was considered a strict specialist of old conifer forests (Strickland and Douglas 1987). Factors that guide marten habitat choice could be summarized in three categories (Thompson and Harestad 1994): predator avoidance, special habitat features, and prey availability. Martens prefer to live under a dense canopy cover which provides protection against avian predators (Hargis and McCullough 1984, Drew 1995), and avoid stands with less than 30-40% overhead tree cover (Koehler and Hornocker 1977, Spencer et al. 1983, Smith and Schaefer 2002). Coarse woody debris provide accesses under the snow cover during the winter for resting sites, maternal dens (Chapin et al. 1997), and to hunt small mammals (Sherburne and Bissonette 1994). Large trees and snags also provide resting and denning sites (Wynne and Sherburne 1983, Chapin et al. 1997) when they reach sufficient size. Finally, marten could be opportunistic in diet (Martin 1994). Consequently, availability and vulnerability of prey influence its foraging success, and thus affect its habitat selection (Coffin et al. 1997, Andruskiw et al. 2008).

Although key habitat elements could be found in a diversity of forest stands (Chapin and Harrison 1996), martens select forest stands with a complex vertical and horizontal structure (Payer and Harrison 2003). While such structural complexity may be associated with forest age and tree species composition in some regions, these associations are often site-specific and cannot be generalised throughout its range (Chapin and Harrison 1996). From the northern to southern part of its geographic range in eastern North America, different forest types fulfill marten needs. In the coniferous boreal forest, martens usually selected dense conifer and mixedwood stands in Newfoundland (Bateman 1986), or showed no selection for any specific composition in Labrador (Smith and Schaefer 2002). In the mixedwood boreal forest of Quebec, martens sometimes selected conifer forests (Alvarez 1998, Godbout and Ouellet 2008), but may also selected mixedwood and deciduous stands (Potvin et al. 2000). In the mixedwood forests of Maine, martens selected mature forests (> 9 m high, Fuller and Harrison 2000), forests with substantial spruce budworm (*Choristoneura fumiferana*) mortality (Chapin et al. 1997) or mixedwood forests (Katnik 1992). However, marten also used partial cuts in Maine (Fuller and Harrison 2005), second growth forests in Ontario (35-50 yrs, Thompson et al. 2008; 50-90 yrs, Bowman and Robitaille 1997) and old plantations in New Brunswick (20-40 yrs, Pelletier 2005), at least during the summer. Such various patterns of habitat selection across regions clearly suggests that “habitat loss” caused by timber harvesting may lead to different space use strategies following forest management, depending perhaps on which stand types are targeted by the timber industry.

While selected habitats may differ among regions, martens clearly avoid open areas with low canopy cover across its geographic range. These habitats include clearcuts (Potvin et al. 2000), meadows and burns (Koehler and Hornocker 1977), and low productivity stands (Smith and Schaefer 2002) or bogs (Bateman 1986). At the level of the individual, it is generally believed that marten cannot tolerate more than 30-40% open areas inside their home range (Chapin et al. 1998, Fuller and Harrison 2000, Payer and Harrison 2000, Potvin et al. 2000). Despite convergence in marten sensitivity to habitat loss at the home range scale over different regions, in a companion study we found that marten populations sampled over large landscapes (several hundreds of km^2) were more tolerant to habitat loss than previously expected (Cheveau et al. chapter 1). This study was, however, conducted during the dispersal season for juvenile martens. Hencethere is a need to assess whether this greater tolerance persisted in seasons where martens are more territorial. In this study, we evaluated marten response to habitat loss and fragmentation generated by extensive forest harvesting, at the individual level (i.e., inside their home range) during winter, in the same study area. The forest was harvested with clearcuts using a clustered pattern, following an agreement between the provincial government and a First Nation government, which aimed to mitigate the impacts of forest harvesting on indigenous activities, including wildlife harvesting. A dispersed strategy of clearcutting was also used. We thus investigated the effect of these two clearcut dispersion patterns (i.e., habitat loss configuration) on marten space use (i.e., home range size and overlap). Because martens do not tolerate intrasexual overlap in their territory (Katnik et al. 1994), we predicted that marten should not respond to fragmentation induced by forest harvesting with a fusion response, as suggested by Payer et al. (2004)'s results. Similarly, because martens require a home range size proportionally larger than predicted by its body size (Lindstedt et al. 1986, Buskirk and McDonald 1989), we hypothesized that martens would not adopt a fission response. Consequently, we predicted that martens would more likely respond to fragmentation by expanding their home range, as already shown by Potvin and Breton (1997). In parallel, habitat quality of the remaining forest cover could also influence home range size and act as a confounding effect in the analysis of the response of individuals to habitat loss and fragmentation. Hence, as response to forest harvesting may also be mediated by the habitat quality of the remaining habitat, we investigated marten habitat selection at the landscape and at the home range scale. Because home range size is mainly driven by habitat productivity at population and individual scales (McLoughlin and Ferguson 2000), the relationship between home range size and the proportion of different habitat types was used as a predictor to detect suitable and unsuitable habitats. Hence, we predicted that late-seral forests, which often have complex structure, will represent suitable habitat for marten and reduce home range size.

2.2. Methods

2.2.1. Study area

Our study area was located in northwestern Quebec, eastern Canada (49°45'N, 76°00'W), within the territory of the Waswanipi Cree Model Forest (WCMF, 36 506 km^2), and was composed mainly of black spruce (*Picea mariana* [Mill.] B.S.P.) stands with an understory of feather moss, sphagnum and dwarf Ericaceous shrubs. The WCMF was part of the Canadian Model Forest Network, which has aimed to develop, test and share solutions in sustainable forest management. The WCMF was the only Aboriginal-led Model Forest, and was specifically established to minimize conflicts between Cree land users and forest managers through community involvement, participation, and knowledge sharing. The area is located in the James Bay lowlands, which are characterized by mostly flat topography (with abundant poorly drained soils) with sparse hills. Despite the fact that forest fires are the major natural disturbance in this landscape (Bergeron et al. 2001, Le Goff et al. 2007), forest harvesting has been the main source of disturbances for the past decade. Since 1988, harvesting has been conducted using clearcuts in a clustered or a dispersed pattern. In the clustered-cut landscapes (50-200 ha cut-blocks), the residual forest was composed of narrow corridors, 20-100 m wide, along streams, or between cut-blocks. In the dispersed-cut landscapes (10-80 ha cut-blocks), the residual forest was composed of uncut forest blocks equivalent in size to adjacent cut-blocks. Except for regenerating stands from a recent fire (1986), and forest harvesting (< 20 yr old), the landscape was mainly composed of stands > 70 yr old. Indeed, the region was characterized by a gap in forest succession, as 30 and 50 yr old age classes were nearly absent.

For the present study, we focussed on two landscapes (delineation defined below): one with clustered clearcuts (518 km^2, 49°22'N 76°06W) and one with dispersed clearcuts (341 km^2, 49°54'N 75°55W, see Fig. 2.1). Habitat composition (Table 2.1) varied slightly between the two studied landscapes. The major difference was related to the proportion of mixedwood and coniferous stands. The clustered-cut landscape was composed of more mixedwood stands, 25% of all residual mature forests compared to 8% in the dispersed-cut landscape. On the contrary, the dispersed-cut landscape was composed of more coniferous stands; 92% of all remnant mature forests compared to 75% in the clustered-cut landscape. The overall proportion of clearcuts was 29% and 20% in the clustered-cut and the dispersed-cut landscapes, respectively.

2.2.2. Marten telemetry data

We captured martens during the fall season (26 September - 6 November 2006, see Cheveau et al. chapter 1 for methods). Age (cementum analyses, Matson's Laboratory LLC, Milltown, Montana, USA, adult = 1-yr and older, juvenile = young-of-the-year) and body index (mass/body length, as an index of general health) was determined for each individual. Radio-collars (Holohil MI-2, Carp, Ontario, Canada, 24 g with mortality signal) were installed on 27 female martens (> 500 g). Only 22 martens were followed for > 10 locations. We conducted a telemetry survey using a Cessna-337 airplane and concentrated our survey during the early winter (20 November - 19 December 2006; only one location per day). Locations were plotted on 1:20 000 maps and coordinates were transferred to *ArcGIS 9.3* (ESRI Institute). We estimated our precision in locating marten from the airplane at < 50 m (mean = 33 m, median = 12 m), using known locations (N = 14). One marten seemed to travel without establishing a territory and another died after only 11 locations before its home range size had stabilized. We decided to exclude these two individuals from home range analysis. We finally obtained 20 home ranges (> 15 locations / marten), eight in the dispersed-cut landscape and 12 in the clustered-cut landscape. The trapping protocol and all manipulations were approved by the Institutional Commitee of the Canadian Council for Animal Care (CCAC) at UQAT (Université du Québec en Abitibi-Témiscamingue).

Two types of home ranges, computed with *RANGES 6* (Kenward et al. 2003), were used for habitat selection analyses. The 100% minimum convex polygon (MCP100) delineated the general winter home range and the 60% kernel (K60) located high activity zones for each marten (Fig. 2.1). We used a fixed kernel with LSCV (least square cross validation) estimator for the smoothing parameter (Seaman and Powell 1996), which seemed to produce the lowest bias and the lowest surface fit error (Seaman et al. 1999). Because the reference smoothing parameter tended to overestimate range areas (Worton 1989), we multiplied it by a fractional value. We first estimated this fractional multiplier by LSCV which reached an inflection point for each home range and then used the median of these multipliers as a fixed multiplier for all the home ranges, as suggested by Kenward et al. (2003). Despite the fact that LSCV tended not to perform well with < 30 locations (Seaman et al. 1999), all our home ranges, except one, reached an inflection, indicating that a local minimum had been found.

We analyzed the relationship between marten home range size (MCP100 and K60) and landscape type (dispersed vs clustered clearcuts), age, body mass, and body index of martens using generalized linear models (GLM, 10 models, see Appendix 2.1), model selection (using Akaike information criterion (AIC)) and multimodel inference (Burnham and Anderson 2002). We also tested the relationship between home range size and its composition (% of each habitat, described later, 16 models, see Appendix 2.1). We conducted the same analysis for marten body condition and related this metric to home range composition (same as for the previous analysis), landscape type and age (18 models).

2.2.3. Scales and habitat types

Habitat selection was analysed at two scales: 1) at the home range scale, we compared habitat use and availability within each home range, and 2) at the landscape scale, we compared home range composition and habitat availability within each landscape (3rd and 2nd order selection respectively, Johnson 1980; design III and II respectively, Thomas and Taylor 1990).

Using digital forest cover maps, we defined 9 habitat types (fine-grained habitats) in our study area (see Table 2.1): recent clearcuts (< 11 years, RC), old clearcuts (≥ 11 years, OC), regenerating stands after fire (FI), lakes and large rivers (WA), forested bogs (BO), mature and late-seral mixedwood forests > 70-year-old (either coniferous or deciduous dominated, mainly 70-90 and 120 yr age classes, M70120), late-seral conifer forests (> 120-year-old, C120), conifer forests 70-90-year-old (which are mainly black spruce forests, C7090), and non-forested areas (NF). Late-seral mixedwood forests were classified as mixedwood forests (M70120) because we assumed that the mixedwood component could be important habitat for marten since it was preferred in the southern portion of the eastern boreal forest of Canada (Potvin et al. 2000). Age classes were all derived from forest cover maps (Quebec Ministry of Natural Resources and Wildlife 1994). We also grouped habitat types in coarse-grained habitats: clearcuts (CC) merging RC and OC, remnant forests (RF) merging C120, C7090 and M70120 and open areas (OA) merging RC, OC, FI and BO. Habitat maps were extracted from digitized forest maps (1:20 000), rasterized and imported in ASCII format in *R 2.9.2*, where habitat selection analyses were performed. We used 10 x 10 m pixels for the home range scale analysis and 100 x 100 m pixels for the landscape scale analysis.

2.2.4. Habitat selection

2.2.4.1. Home range scale

We analyzed habitat selection using resource selection functions (RSF) for a separate sampling of available and used units (SP-A protocol, Manly et al. 2002). RSFs were based on the comparison between habitat composition (fine-grained habitats only) at marten telemetry locations (habitat used) and at 1000 random points (habitat available) created within each individual home range (MCP100). All analyses were conducted in *R 2.9.2* (packages *adehabitat, maptools* and *sp*). We defined as available units all pixels (10 x 10 m) inside a 50-m buffer around random points, and as used units all the pixels inside a 50-m buffer around marten locations. No overlap was tolerated between used and available units, as suggested by Thomas and Taylor (2006). Habitat types which represented < 1.5% of the home range were discarded as suggested by Erickson et al. (1998). For each landscape type, we performed a logistic regression using a generalized linear mixed model (GLMM, Gillies et al. 2006; package *lme4*) with a random intercept for each individual. We predicted RSF values using the particular exponential form suggested by Manly et al. (2002). Then, we estimated relative probabilities of use for each habitat type derived from RSF values and scaled between 0 and 1.

2.2.4.2. Landscape scale

At the landscape scale, habitat selection was assessed by comparing the composition of marten home ranges (habitat used) to random polygons (habitat available) within each landscape. The delineation of the landscape is critical at this level (Johnson 1980). We defined the two landscapes as the minimum concave polygon around all capture points where traps were set and marten telemetry locations to which a 1-km influence zone was added (Fig. 2.1), which represented the sampled study area. We generated random polygons with the same distribution of size as marten home ranges (Gaussian distribution defined by mean and standard deviation, bounded by minimum and maximum) within each landscape, using *SELES* (Spatially Explicit Landscape Event Simulator, Fall and Fall 1996). We conducted 1000 iterations of 8 (for the dispersed-cut landscape) or 12 (for the clustered-cut landscape) random polygons for the distribution in size of MCP100 and K60 in each landscape. Available units corresponded to all pixels (100 x 100 m) within random polygons, whereas used units were all pixels within marten home ranges. We characterized the available resources using both composition (fine- and coarse-grained habitats) and configuration variables: edge density (m/ha), road density (m/ha) and the proportion of residual forest core area (%), which excluded a 50-m interior buffer (CARF). We calculated the proportion of each habitat type from all iterations of random polygons generated (mean percentage for 8 or 12 polygons). We then compared the mean percentage of each habitat type in marten home ranges with the distribution of mean percentage from the 1000 iterations, for each landscape, using *R*. Analyses of edge and road densities were conducted in *ArcGIS 9.3*.

2.3. Results

2.3.1. Home range size

Mean female marten home range size (MCP100) was 5.69 km^2 (2.12-11.39 km^2) in the dispersed-cut landscape and 4.23 km^2 (0.85-9.82 km^2) in the clustered-cut landscape. Mean kernel (60%) size (higher activity zone) was 1.72 km^2 (0.30-3.91 km^2) in the dispersed-cut landscape and 1.16 km^2 (0.17-3.00 km^2) in the clustered-cut landscape.

Home range size was not related to landscape type (clustered- vs. dispersed-cut), female age, body mass, or body index (Table 2.2). Home range size was not related to the proportion of different clearcuts (RC, OC, nor CC) in the home ranges for both MCP100 and K60, nor was it related to the proportion of late-seral forests (C120, Fig. 2.2 and Table 2.2). Home range size was best predicted by the proportion of mixedwood forests (M70120) and conifer forests 70-90-year-old (C7090) in the home range, for both MCP100 and K60 (best models, Appendix 2.1). Moreover, these two variables had a substantial effect in explaining home range size (MCP100 and K60, 95% confidence interval excludes 0, see Table 2.2). However, home range size decreased with increased proportion of M70120 and increased with increased proportion of C7090 (Fig. 2.2).

We found no overlap in any female home ranges (see Fig. 2.1). Of the 20 home ranges, seven were composed of ≥ 40% CC (4-53%), including five with ≥ 40% RC (0-53%).

Female body index was positively related to the overall proportion of remnant forest (RF = C120 + C7090 + M70120) in the home range (best model, Akaike weight = 0.37; RF: 0.08 ± 0.05, but 95% confidence interval includes 0).

2.3.2. Habitat selection

2.3.2.1. Home range scale

Within home ranges, all habitat types (except forested bogs in the dispersed-cut landscape) were selected more by marten than recent clearcuts (Table 2.3). The most used habitat in the two landscape types was mixedwood forest (M70120), followed closely by late-seral conifer forests (C120) and conifer forests 70-90-year-old (C7090, Table 2.4). In contrast, marten avoided forested bogs (BO) and recent clearcuts (RC) in the two landscape types (Table 4). In the clustered-cut landscape, old clearcuts (OC) were highly used (Table 2.4), but this effect was almost entirely driven by one individual (#462) who established its home range in a 30-year-old post-clearcut regenerating area. When excluding this individual, old clearcuts were less used by other martens (Table 2.4).

2.3.2.2. Landscape scale

In the dispersed-cut landscape, marten home ranges (MCP100) included significantly more mixedwood forests (M70120), overall remnant forests and core area of remnant forests than did random polygons. Higher activity zones (K60) also had significantly more mixedwood forests, residual forests, core area of residual forests and conifer forests 70-90-

year-old (C7090). Likewise, home ranges had significantly less recent clearcut area, overall clearcut area, edge densities, road densities, and marginally less open area (Table 2.5). In the clustered-cut landscape, marten home ranges were comprised of more mixedwood forests (MCP100: marginally and K60: significantly) than random polygons (Table 2.5).

2.4. Discussion

2.4.1. Marten space use in fragmented landscapes

Our prediction that home ranges would expand as a response to habitat loss (i.e. proportion of clearcuts) was not supported. Despite a large difference in proportion of clearcuts within marten home ranges between the two landscape types (dispersed: 19% (range: 4-43%); clustered: 35% (range: 15-53%)), we found no relationship between home range size and the proportion of clearcuts, nor between home range size and landscape type. Hence, contrary to our prediction, marten home range size did not increase in human-induced fragmented landscapes in this area of the boreal forest. The dispersion pattern of clearcuts did not influence marten home range size either. We suggest that at this level of remnant forest habitat within home ranges (dispersed: 60% (range: 48-70%); clustered: 53% (range: 35-81%)), configuration effects were possibly still unpronounced. However, we found that home ranges were smaller in the clustered-cut landscape than in the dispersed-cut landscape which could be explained by the higher proportion of residual mixedwood stands (three times higher in the clustered-cut landscape). The overwhelming importance of this habitat for martens is likely to dampen other effects related to habitat loss and fragmentation. Despite the fact that we cannot detect individual marten response to habitat loss and fragmentation (as defined by Ims et al. 1993), we suggest that a fusion response is unlikely, considering that we found no overlap between female home ranges. Our landscapes were only moderately fragmented, presenting no isolation between patches. Hence, we also suggest that we did not reach a situation sufficiently extreme to observe a fission response, as found for European marten (*Martes martes*) in France (Mergey 2007).

Contrary to what was expected, late-seral conifer forests (C120) did not influence marten home range size. Paludification, a process characteristic of late-seral forests in this region, causes a decrease in stand productivity. This phenomenon leads to a decrease in tree density (canopy opening) and tree size. Consequently, late-seral forests in the region, despite a complex structure, lack the large trees, large snags and large coarse woody debris that are characteristic of other old-growth forests (Harper et al. 2003). This could partly explain why late-seral forests in our study area were less important than expected for marten. On the other hand, home range size was reduced as the proportion of mixedwood forests increased whereas surprisingly home range size increased when the proportion of conifer forests 70-90-year-old increased. The importance of mixedwood stands in the region was consistent throughout the study. Marten selected such stands at the home range and at the landscape scale (discussed later). The counter intuitive effect of conifer forests 70-90-year-old, which are nevertheless used by marten at the home range scale, could partly be explained by their simple vertical structure. Harper et al. (2003) showed that this age class is characterized by dense and tall stands with poor understory richness and diversity. Indeed, black spruce forests reach their peak in stand-level structural diversity including logs and snags near 150-200–year-old (Harper et al. 2003). Moreover, windthrow affects mostly black spruce stands > 100-year-old (Harper et al. 2002). We suggest that this lack of understory complexity could influence local prey abundance or vulnerability (Andruskiw et al. 2008), forcing marten to travel more to find food.

2.4.2. Marten habitat selection in boreal landscapes: the unexpected importance of mixedwood forests

Despite the fact that mixedwood forests are a rare habitat type (only 4 and 12%) in our dominant coniferous landscapes, they were the most selected habitat. Marten selected mixedwood forests both at the home range and the landscape scales of investigation, being located more frequently in this habitat type than expected by chance and maximizing the proportion of this habitat type within their home range. Moreover, the negative relationship found between home range size and the proportion of mature and late-seral mixedwood forests indicated that this habitat type was likely to be optimal habitat in terms of food or other essential resources (McLoughlin and Ferguson 2000). With regards to habitat factors influencing marten habitat choice (Thompson and Harestad 1994), we suggest that mixedwood stands are key habitats, presenting a complex forest structure that reduced the risk of predation and provided increased prey availability and denning sites. According to Hély et al. (2000), coarse woody debris accumulation in boreal mixedwood stands is influenced by time since fire and canopy composition. Mixedwood stands dominated by trembling aspen produce a large amount of coarse woody debris mainly between 80 and 130-year-old (Hély et al. 2000), as dominance of trembling aspen in the canopy declines near 80 years after fire in our study region (Bergeron et al. 1998). Consequently, our 70-120-year-old mixedwood forests are at their peak in structural complexity. In parallel, snowshoe hares, who browse almost exclusively on deciduous species in our region (Jacqmain 2003), are known to be at higher density in mixedwood stands in the same study area (Jacqmain et al. 2007). In winter, it is well known that snowshoe hare contributes significantly to marten diet in eastern North American regions (Thompson and Colgan 1990, Cumberland et al. 2001). Moreover, the relative richness of mixedwood stands compared to the more simple conifer stands could in general explain the attractive value of mixedwood stands for the wildlife community.

Potvin et al. (2000) also found that martens select mixedwood stands in the mixedwood boreal forest, a more southern ecoregion located under 49°N (Gauthier et al. 2000). Our results thus confirm the importance of mixedwood stands in the boreal regions, as these habitats were still selected by marten in the conifer boreal forest where they are much rarer. Moreover, these stands also play a critical role for other species such as moose (*Alces alces*) in our study area by providing late winter habitats (Jacqmain et al. 2008). Unexpected importance of mixedwood stands in this largely conifer-dominated region will require a better understanding of the role of such rare stands in the boreal forest. Some could argue that the best way to make new mixedwood stands in the conifer boreal forest is to cut conifer stands composed with a certain percentage of deciduous trees (trembling aspen and/or paper birch). However, potential similarities and differences between after-fire and after-cut regenerated mixedwood stands should be investigated before using this strategy.

2.4.3. Avoidance of open areas

Marten avoid open areas like recent clearcuts and forested bogs in our study area, corroborating patterns already shown in other regions (open bogs, Raine 1983; recent clearcuts, Thompson and Harestad 1994, Buskirk and Powell 1994, Poole et al. 2004). At the home range scale, these two habitats were the least used by martens in the two studied landscapes. Thompson and Colgan (1994) found that martens had lower densities in old clearcut zones for at least 40 years, to limit predation risk (Thompson 1994, Drew 1995) and because their preferred prey were absent or at low density in these habitats (Thompson and Colgan 1994). At the landscape scale, we also found that marten avoided human disturbances such as clearcuts, especially recent ones, as well as edges and roads, within their higher activity zones (defined by kernels) in the dispersed-cut landscape. Such avoidance was not detected by our analyses in the clustered-cut landscape, likely because delineated kernels included clearcuts around remnant forest strips in the activity zone, even though marten did not use clearcuts. In addition, we used coarse scale maps (100 x 100 m pixels) for habitat selection analysis at the landscape scale, due mainly to computation limitation. The resolution of pixels could locally omit narrow linear structures like forest strips, which were usually 20-100 m wide.

2.4.4. Tolerance to habitat loss

Previous studies have shown that American marten cannot tolerate more than 20-40% of their home ranges in open and regenerating areas (Chapin et al. 1998, Payer and Harrison 1999, Fuller and Harrison 2000, Potvin et al. 2000, Dumyahn et al. 2007). Surprisingly, we found that 35% of martens had more than 40% clearcuts within their home ranges in our study area. This result corroborates previous findings at the population level that martens are probably more tolerant to habitat loss and fragmentation in this part of the boreal forest (Cheveau et al. chapter 1) than are southern populations. In this study, we found only a weak relationship between the proportion of clearcuts in the surrounding context and marten site use, abundance, and body condition, and we failed to detect a sharp decline in marten occurrence and abundance when the surrounding proportion of clearcuts increase. Fuller

(2006) also found that Newfoundland martens (*Martes americana atrata*) could tolerate high levels of habitat loss (60-70%) at similar latitudes. This may suggest that martens in boreal ecosystems could be preadapted to natural forest fragmentation, as they evolved in landscapes where large disturbances such as fire events of several hundreds km^2 co-occur with large continuous tracks of late-seral forests (Bergeron et al. 2001, 2004, 2006, Le Goff et al. 2007). Marten population tolerance to habitat fragmentation in this part of the boreal forest could have evolved in the context where unburned forests in post-fire landscapes (7-37% of the area, Perron et al. 2009) are surrounded by an unburned forest matrix that is dominated by late-seral cover (Cyr et al. 2010).

2.5. Conclusion

Despite the fact that marten spatial use did not reflect any specific strategy in response to habitat loss and fragmentation caused by forest harvesting, marten avoided clearcuts at the home range scale and human disturbances (clearcuts, edges and roads) at the landscape scale. We have shown that a marten population in eastern boreal forests of Canada is more tolerant to habitat loss and fragmentation than southern populations, and this was concordant at different scales of investigation (population level, Cheveau et al. chapter 1, and individual level, this study).

Mixedwood forests embedded within coniferous landscapes seem to play an important role for marten, even if they are rare at the landscape scale. Cree land users already recognized the importance of mixedwood forests for wildlife and asked for specific management guidelines for these stands (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002). Our results provide compelling evidence that mature and late-seral mixedwood stands should be included as permanent retention blocks in forest management planning as they represent critical habitat for many wildlife species in the boreal forest of eastern Canada. Considering the importance of mixedwood stands for marten populations in this region, we recommend that forest harvesting keeps focusing on conifer stands (and then, retaining mixedwood stands), mitigating habitat loss and fragmentation effects for this animal species.

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Table 2.1. Composition of the two studied landscapes (% of the total area of each landscape): C is the clustered-cut landscape and D is the dispersed-cut landscape.

	Residual mature forest					Regenerating stands from				
	Overall	C120[1]	C7090[2]	M70120[3]	Forested bogs	Recent clearcuts[4]	Old clearcuts[4]	Fire	Water	Non-forested
C	*47%*	9%	26%	12%	10%	23%	6%	-	10%	5%
D	*52%*	23%	25%	4%	15%	12%	8%	4%	7%	3%

[1] conifer stands > 120 years (estimated age from forest maps).

[2] conifer stands 70 and 90 years (estimated age classes from forest maps).

[3] mixedwood stands 70 to 120 years (estimated age classes from forest maps).

[4] recent clearcuts were < 11 years; old clearcuts were ≥ 11 years.

Table 2.2. Model averaged estimates (beta), unconditional S.E. and 95% confidence interval associated with variables explaining female marten home range size.

	K60			MCP100		
	Beta	Uncond. S.E.	Confidence interval	Beta	Uncond. S.E.	Confidence interval
Landscape	0.59	0.53	(-0.45, 1.62)	1.54	1.54	(-1.48, 4.56)
Body index	-0.04	0.19	(-0.41, 0.33)	-0.19	0.54	(-1.24, 0.87)
Body mass	0.001	0.01	(-0.01, 0.01)	0.004	0.02	(-0.02, 0.03)
Age	0.07	0.63	(-1.17, 1.31)	0.28	1.80	(-3.25, 3.80)
M70120	-0.03	0.01	**(-0.06, -0.01)***	-0.12	0.04	**(-0.20, -0.04)***
C120	0.01	0.02	(-0.03, 0.06)	0.01	0.07	(-0.12, 0.14)
C7090	0.02	0.02	(-0.01, 0.05)	0.13	0.05	**(0.03, 0.23)***
Recent clearcuts	-0.004	0.02	(-0.04, 0.04)	-0.04	0.05	(-0.13, 0.05)
Old clearcuts	0.003	0.02	(-0.03, 0.04)	0.02	0.06	(-0.10, 0.14)
Forested bogs	0.05	0.03	(-0.001, 0.10)	0.17	0.11	(-0.04, 0.38)
Regenerating stands from fire	-0.02	0.04	(-0.10, 0.06)	-0.01	0.12	(-0.25, 0.22)
Non forested	-0.01	0.03	(-0.07, 0.06)	0.21	0.22	(-0.21, 0.64)
Water	0.01	0.03	(-0.06, 0.07)	-0.17	0.21	(-0.59, 0.25)
Residual forest	-0.01	0.01	(-0.04, 0.02)	0.02	0.12	(-0.21, 0.26)
Clearcuts	-0.004	0.02	(-0.04, 0.03)	-0.05	0.06	(-0.16, 0.07)
Open areas	0.01	0.02	(-0.03, 0.04)	0.04	0.11	(-0.17, 0.24)

* Bolded values indicate where confidence interval excludes 0, which means that the variable has a substantial effect in explaining home range size.

Table 2.3. Mixed-effects logistic regression model of resource selection by martens with their selection coefficients and standard errors, in the two landscapes.

Habitat type*	Dispersed-cut landscape	Clustered-cut landscape
Old clearcuts	0.889 ± 0.078	2.537 ± 0.048†
Forested bogs	-0.157 ± 0.086	0.798 ± 0.057
M70120	2.286 ± 0.061	2.530 ± 0.035
C120	1.952 ± 0.058	2.418 ± 0.041
C7090	1.936 ± 0.057	2.320 ± 0.033
Regenerating stands from fire	1.347 ± 0.072	-
Water	-	1.189 ± 0.070

*Reference category was recent clearcuts.

†1.520 ± 0.070 when marten #462 is ignored.

Table 2.4. Predicted resource selection functions (RSF) and relative probabilities of use (RPU) of the different habitat types, in the two landscapes.

	Dispersed-cut landscape		Clustered-cut landscape	
Habitat type	RSF	RPU	RSF*	RPU*
Recent clearcuts	0.008	0.014	0.009 (0.010)	0.000 (0.000)
Old clearcuts	0.019	0.169	0.114 (0.044)	1.000 (0.291)
Forested bogs	0.007	0.000	0.020 (0.021)	0.105 (0.102)
M70120	0.078	1.000	0.113 (0.127)	0.990 (1.000)
C120	0.056	0.690	0.101 (0.117)	0.876 (0.915)
C7090	0.055	0.676	0.092 (0.095)	0.790 (0.732)
Regenerating stands from fire	0.031	0.338	-	-
Water	-	-	0.030 (0.031)	0.200 (0.180)

*Numbers in parentheses are when marten #462 is ignored.

Table 2.5. Mean ± S.E. of the proportion of the different habitat variables for random polygons (N = 1000 simulations of 8 or 12 polygons) compared to the mean proportion of habitat variables within marten home ranges (MCP100 and K60) in the two landscapes. Bolded values showed significant (or nearly significant) differences between random polygon and home range composition ($P < 0.05$).

	Dispersed-cut landscape						Clustered-cut landscape					
	MCP100			K60			MCP100			K60		
	Mean ± SE*	Home range mean	p†	Mean ± SE*	Home range mean	p†	Mean ± SE*	Home range mean	p†	Mean ± SE*	Home range mean	p†
<u>Fine-grained habitats</u>												
Recent clearcuts	12.8 ± 3.0	13.5	0.587	12.4 ± 4.7	4.5	**0.032**	23.4 ± 7.9	31.5	0.842	23.5 ± 9.3	19.1	0.324
Old clearcuts	8.1 ± 4.5	5.8	0.339	8.3 ± 5.7	6.6	0.445	6.2 ± 4.8	6.3	0.548	6.5 ± 5.8	4.5	0.438
Forested bogs	14.5 ± 3.6	10.4	0.128	14.5 ± 5.2	10.9	0.257	9.9 ± 3.4	5.8	0.097	10.2 ± 4.9	4.9	0.129
M70120	4.4 ± 2.2	8.6	**0.036**	4.3 ± 2.9	13.2	**0.004**	11.8 ± 4.1	18.9	**0.056**	11.4 ± 5.5	23.6	**0.020**
C120	22.6 ± 4.3	18.9	0.781	22.5 ± 6.0	17.4	0.790	8.6 ± 2.9	5.3	0.881	8.4 ± 4.0	5.0	0.794
C7090	25.4 ± 5.4	33.5	0.072	25.1 ± 7.1	38.1	**0.040**	26.0 ± 5.0	25.1	0.554	25.9 ± 7.0	27.8	0.366
Regenerating stands from fire	3.5 ± 4.0	5.8	0.720	3.7 ± 4.9	5.0	0.682	-	-	-	-	-	-
Non forested	2.4 ± 1.2	3.0	0.747	2.5 ± 1.7	3.5	0.774	4.8 ± 1.8	3.5	0.240	4.9 ± 2.6	5.1	0.585
Water	6.2 ± 4.7	0.4	**0.021**	6.8 ± 6.3	0.8	0.139	9.4 ± 5.7	3.7	0.171	9.4 ± 7.3	10.1	0.579
<u>Coarse-grained habitats</u>												
Clearcuts	21.0 ± 4.5	19.3	0.372	20.7 ± 6.0	11.1	**0.044**	29.9 ± 7.8	37.8	0.846	29.9 ± 9.7	23.6	0.284
Residual forests	52.4 ± 4.6	61.0	**0.023**	52.3 ± 6.6	68.6	**0.003**	46.1 ± 6.7	49.2	0.319	45.5 ± 9.0	56.4	0.117
Open areas	39.1 ± 5.4	35.5	0.259	38.8 ± 7.3	27.1	**0.051**	39.7 ± 8.0	43.6	0.692	39.9 ± 10.0	28.5	0.139
<u>Habitat configuration</u>												
Edge density	22.5 ± 3.7	20.0	0.254	25.1 ± 5.4	13.5	**0.013**	26.9 ± 5.8	29.3	0.673	33.3 ± 8.3	26.2	0.195
Road density	9.6 ± 1.6	7.5	0.111	11.2 ± 2.6	4.8	**0.005**	10.8 ± 2.7	12.8	0.791	14.0 ± 4.2	12.3	0.392
Core area of residual forests	29.9 ± 4.2	37.4	**0.045**	29.4 ± 6.5	44.1	**0.014**	28.3 ± 6.7	30.0	0.374	27.6 ± 8.9	31.8	0.292

* Mean % for all habitat variables except edge and road densities (m/ha).

† Probabilities associated to the rejection of the hypothesis. We hypothesized that forested areas (M70120, C120, C7090, RF and CARF) were more abundant in marten home ranges than in random polygons and the opposite for all other variables.

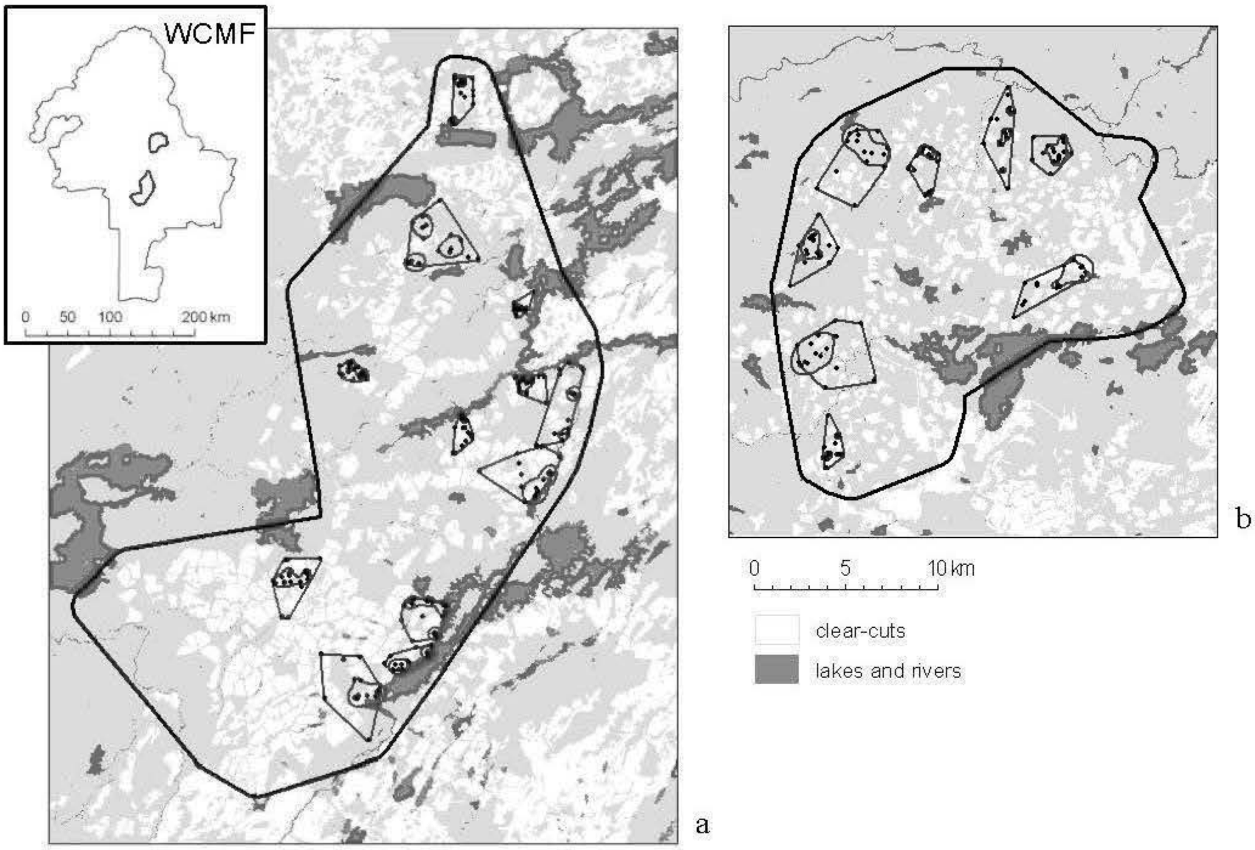

Figure 2.1. Location of the two study areas within the Waswanipi Cree Model Forest (WCMF, 49°45'N, 76°00'W). Study areas were defined as the minimum concave polygon around capture stations and marten telemetry locations (shown) in a- the clustered-cut and b- the dispersed-cut landscapes. Marten home ranges (MCP100 and K60) were shown.

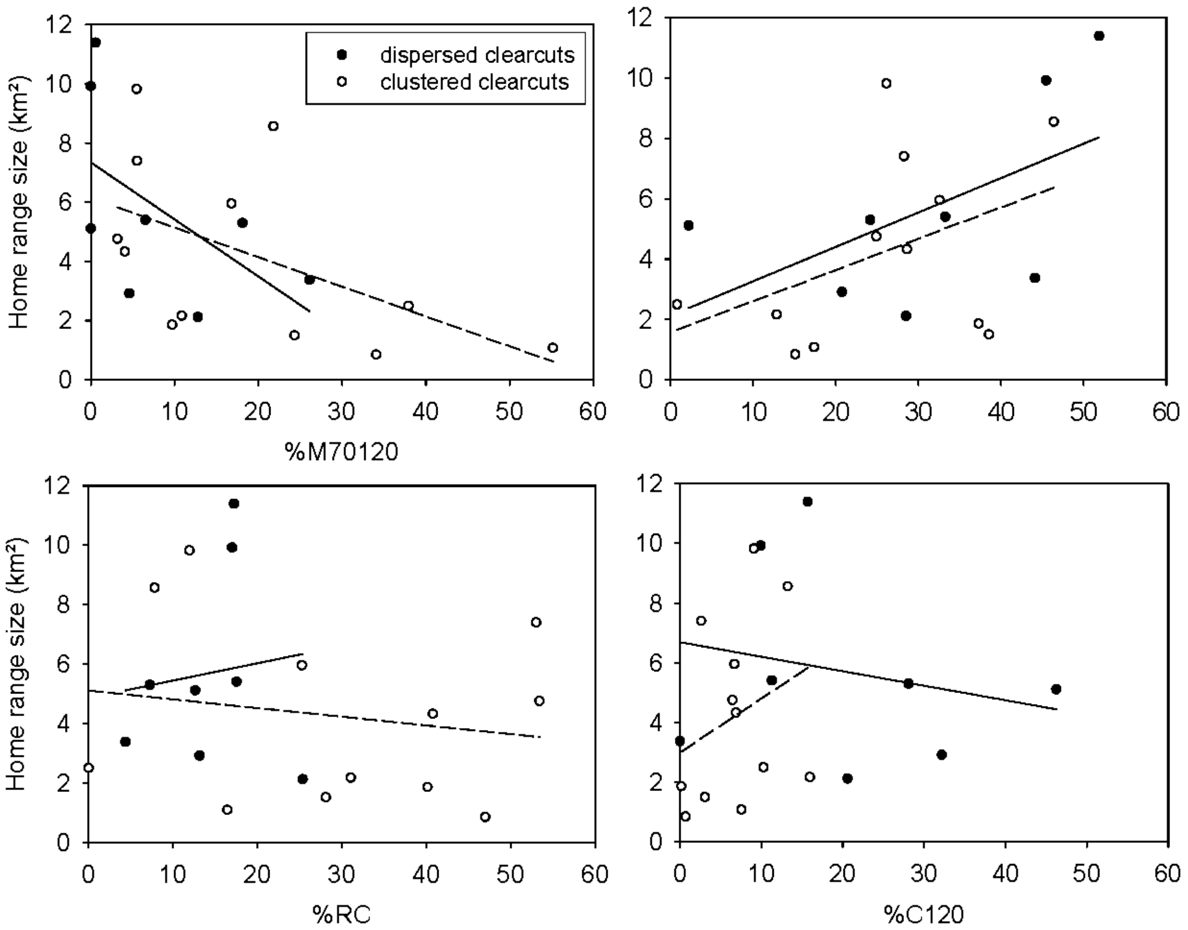

Figure 2.2. Home range (MCP100) size was related to the proportion of mixedwood forests (M70120), conifer forests 70-90 yr old (C7090), late-seral conifer forests (C120) and recent clearcuts (RC) in the dispersed-cut (solid line) and the clustered-cut (dashed line) landscapes.

Appendix 2.1. Model selection for female marten home range size (K60: kernel 60%; MCP100: Minimum Convex Polygon 100%).

K60					MCP100				
Models	K	AICc	ΔAICc	weight	Models	K	AICc	ΔAICc	weight
Landscape	3	66.26	0.00	0.29	Landscape	3	108.39	0.00	0.27
Body mass	3	67.47	1.21	0.16	Body mass	3	109.38	0.98	0.17
Age	3	67.51	1.25	0.16	Age	3	109.39	1.00	0.16
Body index	3	67.57	1.31	0.15	Body index	3	109.41	1.01	0.16
Landscape + body index	4	69.22	2.96	0.07	Landscape + body index	4	111.24	2.84	0.07
Landscape + age	4	69.34	3.08	0.06	Landscape + age	4	111.52	3.12	0.06
Landscape + body mass	4	69.41	3.15	0.06	Landscape + body mass	4	111.55	3.15	0.06
Age + body index	4	70.63	4.37	0.03	Age + body index	4	112.43	4.04	0.04
Landscape + age + body index	5	72.78	6.52	0.01	Landscape + age + body index	5	114.83	6.44	0.01
Landscape + body mass + age	5	72.94	6.68	0.01	Landscape + body mass + age	5	115.12	6.72	0.01
M70120	3	60.52	0.00	0.55	M70120	3	102.14	0.00	0.40
M70120 + C120	4	63.56	3.05	0.12	C7090	3	103.75	1.61	0.18
Forested bogs	3	63.86	3.34	0.10	C120 + C7090	4	103.98	1.85	0.16
C7090	3	66.24	5.72	0.03	M70120 + C120	4	105.09	2.95	0.09
C120	3	66.38	5.86	0.03	Forested bogs	3	106.74	4.61	0.04
Residual forest	3	67.13	6.62	0.02	Non forested	3	108.43	6.29	0.02
Open areas	3	67.16	6.64	0.02	Recent clear-cuts	3	108.70	6.57	0.02
C120 + C7090	4	67.22	6.70	0.02	Water	3	108.76	6.62	0.01
Regenerating stands from fire	3	67.23	6.72	0.02	Clear-cuts	3	108.94	6.81	0.01
Non forested	3	67.53	7.01	0.02	C120	3	109.28	7.14	0.01
Water	3	67.53	7.02	0.02	Old clear-cuts	3	109.39	7.25	0.01
Recent clear-cuts	3	67.54	7.02	0.02	Open areas	3	109.45	7.31	0.01
Old clear-cuts	3	67.56	7.04	0.02	Regenerating stands from fire	3	109.45	7.32	0.01
Clear-cuts	3	67.59	7.07	0.02	Residual forest	3	109.47	7.33	0.01
Residual forest + clear-cuts	4	69.80	9.28	0.005	Residual forest + clear-cuts	4	111.20	9.06	0.004
Residual forest + open areas	4	70.29	9.78	0.004	Residual forest + open areas	4	112.24	10.10	0.002

CHAPITRE III

AMERICAN MARTEN RESPONSE TO FOREST/CLEARCUT EDGES AND CORRIDOR USE IN THE BOREAL FOREST OF EASTERN CANADA

Marianne Cheveau, Pierre Drapeau, André Desrochers, Louis Imbeau and Louis Bélanger

Résumé

La création de lisières est une conséquence fréquente de la fragmentation de l'habitat. Les prédateurs sont souvent attirés par les lisières forestières où les opportunités de quête alimentaire sont supposées être meilleures qu'à l'intérieur des forêts. Malgré la sensibilité de la martre d'Amérique (*Martes americana*) à la fragmentation de l'habitat, les effets des lisières sur leur distribution et leurs mouvements n'ont que rarement été étudiés, de même que leur utilisation potentielle des corridors forestiers. Nous avons analysé la réponse des martres aux lisières forêt/coupe et aux bandes forestières, en forêt boréale de l'est du Canada. Tel qu'escompté, les martres évitaient les coupes, où l'abondance des proies était faible. De même, les pistes de martres étaient plus nombreuses près des lisières forêt/coupe. La distribution des martres par rapport aux lisières avait tendance à suivre celle des proies, qui étaient aussi plus abondantes près des lisières, tout comme les accès sous-nivaux. Les martres se déplaçaient plus fréquemment parallèlement aux lisières lorsqu'elles en étaient proches comparativement à ce qu'elles faisaient à l'intérieur des forêts. Ceci suggère que les lisières agissent comme barrière aux déplacements. Les étroites bandes forestières étaient intensivement utilisées par les martres et leurs mouvements y étaient plus linéaires. Malgré le fait que ces bandes forestières ne soient pas destinées à fournir de la connectivité dans le paysage, elles peuvent jouer le rôle de corridors de déplacement pour les martres et les autres espèces fauniques, qui évitent ainsi les milieux ouverts dans les paysages fragmentés de manière extensive par l'exploitation forestière.

Mots-clés

Martre d'Amérique, effet de lisière, corridors, mouvements, lisières multiples, sinuosité, lièvre d'Amérique, écureuils, petits mammifères

Abstract

Edge creation is a frequent by-product of habitat fragmentation. Predators are often attracted to forest edges where foraging opportunities are presumed to be better than in forest interiors. Despite the sensitivity of American marten (*Martes americana*) to habitat fragmentation, the effects of edges on its distribution and movements have been rarely investigated, as has its potential use of forest corridors. We analyzed marten response to forest/clearcut edges and forest strips in the boreal forest of eastern Canada. As expected, martens avoided clearcuts, where prey abundance was low. According to expectation, marten tracks were at higher densities near forest/clearcut edges. The distribution of martens with respect to edges tended to follow that of the prey, which were more abundant near edges, as well as subnivean access points. Marten more frequently moved in parallel with edges when near the edge compared to the forest interior, suggesting that edges act as barriers to movement. Narrow forest strips were highly used by marten and movements were more linear in such habitats. Although these forest strips were not designed to provide connectivity, they could act as movement corridors for martens and other wildlife species, thereby avoiding open areas in landscapes extensively fragmented by forest harvesting.

Key words

American marten, edge effect, corridors, movements, multiple edges, sinuosity, snowshoe hare, squirrels, small mammals

3.1. Introduction

Habitat fragmentation induced by human activities (agriculture, timber harvesting) is associated with the creation of sharp edges (Saunders et al. 1991). These non-natural edges generally modify several properties of remnant habitat patches. Edge effects are well-documented and affect not only microclimate, including light, wind, temperature and humidity (Chen et al. 1995), but also vegetation (Rheault et al. 2003, Marozas et al. 2005, Boudreault et al. 2008), and forest composition and structure (Harper et al. 2005), including coarse woody debris, fallen trees and canopy cover (Mascarùa-Lopez et al. 2006). These biophysical impacts can influence animal distributions (invertebrates, Haddad and Baum 1999; rodents, Mills 1995), as well as predation risk and nest parasitism, which is well-documented for birds (Paton 1994, Donovan et al. 1997, Batary and Baldi 2004). Effects of edges on animal movements have been less frequently investigated (but see Desrochers and Fortin 2000, Desrochers et al. 2003), except perhaps to evaluate their permeability (Stamps et al. 1987, Gibbs 1998, St Clair et al. 1999). The width of edge effect on animals depends on its abruptness and the contrast between the two habitats, together with the perceptual abilities of each species (Larivière 2003).

The use of corridors by animals is related to their response to edge (Lidicker 1999). Indeed, the study of corridors could be considered a specific case of the study of edge effects, as they are made of two edges in a close proximity, the interaction of which could create a response greater than that incurred from the influence of either edge alone (Harper et al. 2007). Advantages and disadvantages of corridors for conservation strategies have been highly debated (Noss 1987, Simberloff and Cox 1987, Simberloff et al. 1992, Beier and Noss 1998, Noss and Beier 2000, Haddad et al. 2000). The main points on each side have been summarized by Bennett (2003), and include the facilitation of movements, the maintenance of the continuity of natural ecological processes, and habitat creation. However, movement facilitation could be either an advantage for isolated subpopulations (Hilty et al. 2006), in enhancing gene flow (Aars and Ims 1999) or a disadvantage in spreading exotic species (Downes et al. 1997) or disease (Johnson and Cully 2005), and facilitating movements of generalist predators (Gehring and Swihart 2003). Evidence of corridor effectiveness in providing connectivity for animals is accumulating from field studies on various taxa, including insects (Haddad 1999, Tewksbury et al. 2002, Webb et al. 2008), birds (Machtans et al. 1996, St Clair et al. 1999, Darveau et al. 2001) and mammals (La Polla and Barrett 1993, Beier 1993, Potvin and Bertrand 2004). Conversely, corridors are sometimes not effective in increasing connectivity (see Beier and Noss 1998, Hannon and Schmiegelow 2002), for example because focal species do not use them (reported in Hilty et al. 2006).

Mammalian predators are usually assumed to be attracted by edges and corridors where foraging opportunities are considered to be better than in interior habitats (Robinson et al. 1995, King et al. 1998), although empirical evidence is often not that clear (Hansson 1994, Heske 1995). Among such predators, American marten (*Martes americana*) is a forest specialist, but it is also a prey species at the same time; both characteristics likely affect its selection of habitats as well as fine-scale movement behavior. While a few studies have investigated the effects of edges on marten, in most cases at home range- or landscape-scales (Chapin et al. 1998), to the best of our knowledge, none were conducted directly on individual movements along edges, although such movements are known to be affected by other habitat characteristics (Nams and Bourgeois 2004). In managed landscapes, marten is reported to be sensitive to habitat fragmentation (Hargis et al. 1999) and to avoid clearcuts, and its large habitat requirements are likely to be affected by the many edges that occur in these managed landscapes (Buskirk and Powell 1994). Consequently, marten movements may be channelled by these edge boundaries.

In the eastern boreal forest of Canada, timber is currently harvested under two dispersion patterns of clearcuts over large areas: clustered, where the configuration of residual mature forest is made of narrow forest strips, or dispersed in a checkerboard pattern. These two landscape types offer the opportunity to study the effect of single edges as well as multiple edges (i.e., corridors) on marten. To investigate marten response to forest/clearcut edges and its use of corridors, we used snow tracking, which is considered an efficient means of monitoring marten movements (Raphael 1994, Proulx and O'Doherty 2006). Considering its avoidance of open areas, we predicted that 1) marten would avoid clearcuts, 2) marten would use forest/clearcut edge zones more than expected, 3) marten movements would be more parallel near the edge, which should act as a barrier, and 4) marten would use narrow forest strips as effective movement corridors. To explain marten distribution patterns with respect to edges, we investigated two non-exclusive hypotheses regarding underlying processes: 1) prey that are supranivean (i.e., who live above the snowpack) are distributed in a non-random way, selecting edges and 2) forest composition and structure change near edges would create more subnivean access points to hunt small mammals (who live beneath the snowpack).

3.2. Methods

3.2.1. Study area

Our study area was located in north-western Quebec, eastern Canada (49°45’N, 76°00’W), within the territory of the Waswanipi Cree Model Forest (WCMF). The WCMF was part of the Canadian Model Forest Network, which had aimed to develop, test and share solutions in sustainable forest management. The WCMF was the only Aboriginal-led Model Forest, and was specifically established to minimize conflicts between Cree land users and forest managers through community involvement, participation and knowledge sharing. The study area was located in the James Bay lowlands which are characterized by mostly flat topography (with abundant poorly drained soils) with sparse hills. Mature forests are mainly black spruce stands (*Picea mariana* [Mill.] B.S.P.), with an understory of feather moss, sphagnum and dwarf shrubs in the Ericaceae. Forest fire is the major natural disturbance in this region (Bergeron et al. 2001, Le Goff et al. 2007). Logging has been active mainly since 1988. The spatial pattern of clearcuts was either clustered (i.e., large cutovers areas of several hundreds of km^2 in size) or dispersed in a checkerboard pattern. Most of the clearcuts in the study area are not planted and natural regeneration is generally < 1 m in height. In the dispersed-cut landscapes (10-80 ha cut-blocks), the residual forest was composed of uncut forest blocks equivalent in size to adjacent cut-blocks. In the clustered-cut landscapes (50-200 ha cut-blocks), the residual forest was composed of narrow corridors, 20 to 100 m wide, along streams or between cut-blocks. We investigated marten response to single edges in two dispersed-cut landscapes (314 and 367 km^2), and marten response to the close proximity of two edges (in narrow forest strips) in two clustered-cut landscapes (173 and 480 km^2).

3.2.2. *Effects of single edges*

3.2.2.1. *Field sampling*

We conducted snow tracking surveys during two winters, 2005 and 2007 (February-March) in the dispersed-cut landscapes. We sampled 51 transects perpendicular to forest/clearcut edges, systematically located at 3-km intervals. This distance was chosen to assure independency between transects, as marten home ranges during the winter were around 5-10 km^2 in the study area (Cheveau et al. chapter 2). We consequently assumed that each crossed marten path was from a different individual. Sampled transects were different during the two winters but located in the same two landscapes. Transects were 350-m long (100-m in the clearcut and 250-m in the residual forest). Sampled edges were all located along ≤ 11-year-old clearcuts. All marten tracks crossed by the transect line were noted with a high-precision GPS (Trimble Geo XM) and distance to the edge was measured. We did not restrict our field sampling to a particular time after a snow fall, but we never visited transects earlier than 6-h following snowfall.

When a recent marten track was crossed, we followed it for a distance representing a minimum net displacement of 500-m, whenever this was possible. Over the two winters, we sampled snow tracks of 21 martens in two landscapes, each with an average path length of 625-m (230 to 925-m), and a grand total of 13.1 km. We used continuous real-time GPS sampling (a point was recorded every second) to map exact marten paths.

3.2.2.2. *Data analyses on the effects of simple edges*

First, we tested the expected avoidance of clearcuts by marten. Using a chi-square test, we compared the distribution of marten tracks between clearcuts and residual forests. The expected distribution would be 71% (250-m) in the forests and 29% (100-m) in the clearcuts if there is no selection for any type of habitat. We also calculated marten track density (number of tracks per km) in the residual forests and the clearcuts.

Secondly, we analyzed the distribution of marten tracks with regards to the distance to the forest/clearcut edge. For this analysis, we considered only tracks along the forested part of each transect and discarded those in the clearcut. Using *ArcGIS 9.2* (ESRI), we calculated the distance between each track to the nearest forest/clearcut edge. We used a randomization approach to test the difference between paired distances to edge (random distance – real distance). Indeed, for each real track distance to edge, we generated an associated random distance to edge (between 0-250 m). Positive differences suggested that tracks were nearer to the edge than expected for a random uniform distribution and negative differences suggested that tracks are further to the edge than expected. We repeated the process 1000 times and calculated the proportion of the randomized differences that were higher or smaller than observed difference. All statistical analyses were conducted in *SAS/STAT software 9.1* (SAS Institute).

Third, we analyzed the effect of edges on marten movements. In order to limit redundancy, we selected only one recorded point per five seconds along the marten paths. Using *SAS 9.1*, we filtered out points that were > 1 m/sec from the previous point, as they exceeded the maximum speed that a field observer could walk using snowshoes in the snow. We also removed redundant points that are < 2-m from the previous point. This procedure cleaned marten paths from redundant and outlier points (due to bad GPS precision) and smoothed them. We generated equidistant points along smoothed paths that split movement paths into steps of 10-m, 20-m or 40-m, which represent three spatial scales. Using *ArcGIS*, we calculated the distance between these equidistant points to the nearest forest/clearcut edge. Using *SAS*, we then calculated absolute angles (from 0 to 90°) between each step and the nearest edge (see Desrochers and Fortin 2000). Only steps that were located in the forest were considered in the following analyses. For each marten path, we conducted linear regressions to predict movement angle from the distance (log-transformed) to the nearest edge. We then performed a Student *t*-test on parameter estimates from the regressions, with a null hypothesis that they did not differ from 0, after weighting each estimate with the inverse of the variance for each path. We also performed piecewise regressions (two-piece regression, Proc NLIN) to detect a potential breakpoint in the movement angle of marten paths when distance to edge increased.

3.2.3. Effects of multiple edges in narrow corridors

3.2.3.1. Field sampling

We replicated the previously described methods in forest strips of the two clustered-cut landscapes. We sampled 50 transects that exhibited variable length due to the variable width of the forest strips retained between clearcut blocks (50-m in the clearcut, 35 to 185-m in the forest strip (81% between 50 and 100-m) and 50-m in the second clearcut). We also followed 17 marten paths, each with an average length of 541-m (80 to 960-m), and a grand total of 9.2 km.

3.2.3.2. Data analyses on the effects of multiple edges

To test the avoidance of clearcuts in clustered-cut landscapes, we adjusted the number of tracks for a 100-m strip width. Then, the expected distribution of tracks would be 50% (100-m) in the forest and 50% (100-m) in the clearcuts, if there is no selection for any type of habitat. We used the same approach to test the distribution of tracks with respect to the distance to edge along the forested part of the transect, where random distances were generated between 0 and half of the forest strip width. We used the same procedure to smooth and split marten paths into step lengths, and we performed linear regressions and Student t-tests as described for single edges.

3.2.4. Movement sinuosity

Sinuosity was calculated for sections composed of five steps (5 x 10-m, 20-m or 40-m) using the index developed by Whittington et al. (2004) (log (L/R^2), with L being the path length and R net displacement). We then performed linear mixed models to evaluate the effect of single vs. multiple edges (dispersed vs. clustered clearcuts) on marten path sinuosity, with individual marten path as a random effect, for each scale (50-m, 100-m and 200-m). We also tested the difference in sinuosity for sections near the edge (based on the previously determined threshold point value) vs. deep within the forest, in the dispersed-cut landscapes, at the three scales (one-way ANOVA). We log-transformed the absolute value of the sinuosity index to achieve normality.

3.2.5. Prey distribution

All prey tracks crossed by the transect line were also noted with the GPS. However, only squirrel (red squirrel, *Tamiasciurus hudsonicus,* and northern flying squirrel, *Glaucomys sabrinus* pooled) and snowshoe hare (*Lepus americanus*) tracks were considered for the following analyses. Superimposed tracks were given an arbitrary value of three tracks. Tetraonids (spruce grouse, *Falcipennis canadensis*, ruffed grouse, *Bonasa umbellus,* and sharp-tailed grouse, *Tympanuchus phasianellus*) were not considered because snow tracks are not good indices for bird occurrence and because our sample was small (N = 76 tracks for the two landscape types and the two years pooled). Other rarely encountered species (weasels, fox, wolf, moose, mice and porcupine) were also ignored. As for marten tracks, we performed a chi-square test to analyze the distribution of prey tracks in the residual forests vs. the clearcuts, and randomisations to evaluate if snowshoe hare and squirrel tracks were more frequent near edges. We also calculated prey track density (number of tracks per km) in the residual forests and in the clearcuts.

We also captured small mammals during two summers, one following the first winter of tracking (July 2005) and one preceding the second winter of tracking (July 2006) in the same study area. During the two summers, we visited the same sites, which were a sub-sample of the snow-tracked transects. We installed two parallel lines of SFG Sherman traps at 20-m intervals (six in the clearcut and 16 in the residual forest) along 17 transects, perpendicular to the forest edge, located in the two dispersed-cut landscapes (1152 trap-nights). We also installed traps (six in the clearcut, six to 12 in the residual corridor, depending on its width, and six others in the opposite clearcut) in 17 transects located in the two clustered-cut landscapes (960 trap-nights), for a grand total of 2112 trap-nights each year. Traps were baited with peanut butter and apple, and were set for three consecutive nights. Each animal was tagged, weighed, sexed, identified to species and released. We tested the difference in small mammal abundance (number per 100 trap-nights) between the two landscape types, and between the clearcut and the forest using a hierarchical linear mixed model (GLMM, generalized linear mixed model; fixed effect: landscape type and habitat type (clearcut vs. forest); random effect: 34 transects nested in four landscapes).

3.2.6. Forest structure

During the second winter (2007), we recorded all potential subnivean access points, which were naturally created under logs and fallen trees or under low branches of young trees, within a distance of < 2-m each side of the transect line. Snow depth was ~ 50 cm during that survey (February 2007). We used the same approach as previously described for marten and prey tracks to analyze the distribution of potential subnivean access points in the residual forests vs. in the clearcuts, and their distribution with respect to the distance to the edges in the residual forests.

3.3. Results

3.3.1. Avoidance of clearcuts

Of 146 marten tracks that were crossed by a transect, six were found in clearcuts, including four at < 20 m from the edge. Marten tracks were more frequent in the residual forests than in the clearcuts, in the two landscape types (dispersed-cut: chi-square = 19.57, $P < 0.001$, df = 1; clustered-cut: chi-square = 63.48, $P < 0.001$, df = 1). Track density was similar in clearcuts in the two landscape types, but was three times higher in the residual forest of the clustered-cut landscapes (strips) than in the dispersed-cut landscapes (blocks, Table 3.1).

3.3.2. Effects of single edges

Marten tracks were nearer the edge (mean ± SD = 95.9 ± 67.0 m) than expected from a uniform random distribution (N pairs = 64, $P = 0.001$ for 1000 randomisations, Fig. 3.1).

Movement angles of marten paths were smaller than an absolute random angle (|45°|) between 0 and ~100 m from a forest/clearcut edge, for the three spatial scales (10, 20 and 40-m steps, see Fig. 3.2). Parameter estimates from linear regression were significantly > 0 for the three spatial scales (Table 3.2), suggesting that marten paths were more parallel (smaller angles) to edges when distance to edge decreased. Piecewise regressions revealed that movement angles decreased when marten were about 75-100 m from an edge (threshold point estimates: 76.6 m for 10-m segments:,101.3 m for 20-m segments, regression did not converge upon a solution for 40-m segments).

3.3.3. Effects of multiple edges

Marten tracks were uniformly distributed inside forest strips in the clustered-cut landscapes (N pairs = 53, P = 0.192 for 1000 randomisations). In contrast to the dispersed-cut landscapes, distance to edge was not a good predictor for movement angles, except at the coarsest scale (40-m), where we observed a pattern opposite to that observed for the dispersed-cut landscapes; marten paths were more parallel (smaller angles) to edges when distance to an edge increased (Table 3.2).

3.3.4. Movement sinuosity

Marten paths were more linear in the clustered-cut than in the dispersed-cut landscapes for the three spatial scales. This response was significant only at the intermediate scale (100-m: p = 0.009). The same response also occurred at the largest scale but was not significant (200-m: p = 0.06) (Table 3.3). We chose 75-m as the threshold point value (see piecewise regressions) to compare movement sinuosity near the edges vs. deep within the forest. Marten paths were more linear near the edges (< 75-m from an edge) than deeper within the residual forests in the dispersed-cut landscapes, at the highest scale only (200-m, ANOVA: F = 5.29, P = 0.027, df = 1).

3.3.5. Prey distribution

Snowshoe hare tracks were more frequent in the residual forests compared to the clearcuts, in the two landscape types (dispersed-cut: chi-square = 86.02, $P < 0.001$, df = 1; clustered-cut: chi-square = 97.30, $P < 0.001$, df = 1). The same distribution was observed for squirrel tracks (dispersed-cut: chi-square = 32.75, $P < 0.001$, df = 1; clustered-cut: chi-square = 30.08, $P < 0.001$, df = 1). Small mammals were significantly more abundant in the forest than in the clearcut in 2006 ($P < 0.001$), and marginally in 2005 (P = 0.068) (Fig. 3.1).

In the dispersed-cut landscapes, snowshoe hare tracks were nearer the edge (mean ± SD = 107.4 ± 54.3 m) than expected from a uniform random distribution (N pairs = 319, P = 0 from 1000 randomisations, Fig. 3.1), but they showed a uniform distribution inside strips in the clustered-cut landscapes (N pairs = 104, P = 0.341). Squirrel tracks were also nearer the edge (mean ± SD = 101.7 ± 61.4 m) than expected from a uniform random distribution along the forested part of transects in the dispersed-cut (N pairs = 107, P = 0 for 1000 randomisations, Fig. 3.1) but further the edge inside forest strips in the clustered-cut landscapes (N pairs = 32, P = 0.001).

Snowshoe hare and squirrel track density in the forest was the same in the two landscape types (Table 3.1). Small mammals were at similar abundance in the two landscape types (GLMM; $t_{1,34} = 2.22$, P = 0.157).

3.3.6. Forest structure

Potential subnivean access points were more frequent in residual forests than in clearcuts, in the two landscape types (dispersed-cut: chi-square = 113.66, $P < 0.001$, df = 1; clustered-cut: chi-square = 50.00, $P < 0.001$, df = 1). Potential subnivean access points were more frequent near the edge along forested parts of transects in the dispersed-cut (N pairs = 656, P = 0 for 1000 randomisations) but uniformly distributed inside forest strips in the clustered-cut landscapes (N pairs = 154, P = 0.708).

3.4. Discussion

3.4.1. Avoidance of clearcuts

As expected, martens avoided recent clearcuts in our study area. This behavior has already been documented for martens, which avoid open areas like meadows and burnt areas (Koehler and Hornocker 1977), open bogs (Raine 1983), recent clearcuts (Thompson and Harestad 1994, Buskirk and Powell 1994, Poole et al. 2004) as well as regenerating clearcuts (Thompson 1994, Payer and Harrison 2003) and open forests showing low productivity (Smith and Schaefer 2002). Thompson and Colgan (1994) found that martens were at low densities in old clearcuts areas for at least 40 years, to limit predation risk (Thompson 1994, Drew 1995) and because their preferred prey were absent or at low density in these habitats, which reduces their chance of capture (Thompson and Colgan 1994). Indeed, marten prey (small mammals, squirrels and snowshoe hares) also avoided clearcuts in our study area. Snowshoe hares avoid clearcuts for several years and relocate immediately after logging to adjacent residual forest (Ferron et al. 1998). Snowshoe hare returns occurred mostly in regenerating stands 4-m in height in our study area, which corresponds to clearcuts aged between 13 and 27 years (Jacqmain et al. 2007). This coincides with the regeneration of a dense shrub layer, which represents browse and cover against predators, especially during the winter (Wolff 1980). Red squirrels and northern flying squirrels also avoid recent clearcuts (Thompson et al. 1989, Lomolino and Perault 2001), and following disturbance, will relocate in residual forests adjacent to recent clearcuts (Côté and Ferron 2001). Red squirrels showed abundances in these adjacent residual forests similar to those of large uncut forests, whereas northern flying squirrels were present, at least in the short term, at low density in residual stands (Côté and Ferron 2001). Their habitat preferences include presence of snags, decaying stumps, complex canopies and coarse woody debris, which provide dens and food (seeds and hypogeous fungi) that are typically found in mature forests (Carey et al. 1999, Holloway and Malcolm 2006). Small mammal responses to clearcuts differ greatly from one region to another (Kirkland 1990). In the boreal forest of eastern Canada, the relative abundance of the various species changes more after logging compared to overall small mammal abundance. Indeed, Cheveau (2003) found an increase in the abundance of deer mouse (*Peromyscus maniculatus*) and meadow vole (*Microtus pennsylvanicus*), and a decrease in abundance of red-backed vole (*Myodes gapperi*), in clearcuts compared to undisturbed forest. We hypothesized that low regeneration (< 1-m height) in the clearcuts explains the lower overall abundance of small mammals in this habitat.

3.4.2. Edge zone use (single edges)

Our second prediction was confirmed; marten selected generally edge zones (0-125 m) more than would be expected by chance in our study area, but tend to be less frequent in the first 20-m than in the section from 20 to 80-m from the edge. This strategy could be interpreted as a compromise between the avoidance of the more risky zone for predation (0-20 m) and the advantage to use the edge zone where prey were at higher densities. However, we found in a companion study that marten abundance was negatively correlated to edge density at a scale corresponding to a mean home range size (Cheveau et al. chapter 1). This suggests that marten response to edge is scale-dependent. Potvin et al. (2000) found fewer forest/clearcut edge zones (defined as a 50-m buffer zone around residual forests) within marten home ranges when compared to the available landscape in the boreal forest. In a companion study, we also found that marten installed their high activity zones where forest/clearcut edges were at lower densities (Cheveau et al. chapter 2). Marten abundance was also negatively correlated to sharp edge density in Utah (Hargis et al. 1999). In contrast, Chapin et al. (1998) detected no difference between observed distance to the edge (from marten locations by telemetry) and expected distance (from random points) within marten home ranges in Maine. Moreover, edge density within marten home ranges was not different from the available landscape in their region. Buskirk and Powell (1994) have suggested that marten use of edges may depend on the habitat composition on either side of these boundaries. For example, martens showed an intensive use of forest/meadow edges in California (Spencer et al. 1983), which could be related to high prey availability in the meadows.

In our study area, we found that snowshoe hare tracks were at higher densities in the edge zone (0-125 m). Snowshoe hares are known to concentrate their activities near edges (forest/clearcut, Conroy et al. 1979) and within ecotones (the transition zone between two different habitats, Ferron and Ouellet 1992). However, they were also frequent around 150-m from the edge, and martens tend to follow their distribution (see Fig. 3.1). This result could be explained by the fact that marten feed preferentially on snowshoe hares in winter in the eastern part of North America (Martin 1994). Marten hunting strategy during this season was reported to be characterized by frequent investigations of logs, roots and stumps (Raine 1987) and other potential prey locations (Spencer and Zielinski 1983), especially searching for resting sites of hares (Thompson and Colgan 1994). Similarly, squirrels were at higher densities in the edge zone in our study area, but tend to avoid the first 20-m. Besides, squirrels do not seem to have a clear response to edge in other regions. Indeed, King et al. (1998) found marginal selection of forest/clearcut edges by red squirrel in deciduous forests of New Hampshire. In Finland, Siberian flying squirrels (*Pteromys volans*) were attracted by young forest/clearcut edges, but did not respond in the same fashion to mature forest/clearcut edges (Desrochers et al. 2003).

Forest structure is affected by forest/clearcut edges in the boreal forest, showing more coarse woody debris and fallen trees near the edge (Mascarùa-Lopez et al. 2006). These habitat elements could create subnivean access points for marten to hunt small mammals or provide thermal resting sites, especially during prolonged periods of low temperatures (Bissonette and Broekhuizen 1995). Indeed, we found that subnivean access points were at higher densities near the edge in our study area, but stayed abundant along the whole transect (> 30 access points per 20-m sections). We suspected that subnivean access points were probably not limiting marten foraging behavior in our study area. Moreover, small mammal abundances did not seem higher near the edges during the summer. Additionally, we noted very few events of marten traveling under the snow, as already mentioned by Potvin (1998) and Thompson and Colgan (1994) in the eastern boreal forest of Canada. Our results corroborate the hypothesis of Thompson and Colgan (1990) that marten preferentially hunt for large prey, especially snowshoe hares, taking small mammals opportunistically between larger catches.

3.4.3. Edge effect on marten movements

We found that marten moved more parallel to an edge when they approached an edge. This behavior could be interpreted as a barrier effect created by forest/clearcut edges (prediction 3), but also as an attraction effect related to foraging activities, considering that marten distribution followed prey distribution. These two hypotheses potentially create a confounding or a synergetic effect; marten are following edges to avoid predation risk associated to the clearcut or following prey tracks to increase their chances of capturing prey. However, such parallel movements along edges, which were not related to food availability, have already been shown for other forest specialists, like black-capped chickadees (*Poecile atricapillus*, Desrochers and Fortin 2000) and three-toed woodpeckers (*Picoides dorsalis,* Gagné et al. 2007). Consequently, we suggest that the strongest effect explaining marten movements near edges was created by a barrier effect. Desrochers and Fortin (2000) predicted that this behavioral response would appear when boundaries (i.e., edges) act as movement conduits. Marten movements were constrained at a fine scale, and this effect was recorded as far as 75-100 m from edges. When marten were at > 75-100 m from any edge, movement angles were random compared to estimated values close to the edge, suggesting individuals were no longer influenced by edge location. The importance of this "edge effect" is similar to those found for other species along forest/clearcut edges (chickadee: 125 m, Desrochers and Fortin 2000; woodpeckers: 80 m, Gagné et al. 2007) and could be related to the size of adjacent open areas. Indeed, smaller openings, like unpaved forest roads, may create a thinner "edge effect" on marten movements as shown by Frouin (*unpublished data*, ~30 m).

3.4.4. Corridor use

As expected, martens used narrow forest strips in our study area as movement corridors. Marten track density was three times higher in corridors than in residual blocks, which was not the case for snowshoe hares and squirrels. Machtans et al. (1996) had hypothesized that, if a species avoids clearcuts, individuals will be concentrated in residual forests, showing higher activity than in control habitats (i.e. large unharvested areas). We suggest that the higher intensity use of narrow forest strips by marten could be explained by this behavior (concentration of activities), rather than by a higher abundance. On the other hand, the smaller size of snowshoe hare and squirrel home ranges could explain the lack of activity concentration observed in forest strips for these species.

Contrary to what was found along single edges, marten moved in a more parallel fashion when distances to edge increase inside corridors, at our highest scale (40-m step lengths). This could be explained by the width of sampled forest strips (median width = 71 m), which provide a maximum distance to edge of ~35 m. At this scale, a marten located in the middle of a strip was unable to move with a 40-m step length without entering the clearcut. If we consider that marten avoid clearcuts, a marten located in the middle of a strip was more likely to move in a more parallel fashion with respect to its edges. When located near the edge of such a strip, however, a marten could move parallel or perpendicular to this edge, i.e., to go back deeper inside the forest strip. Given that the extent of edge effects on marten movement was 75-100 m, forest strips presented no forest interior, which created a canalization effect on movements. Hence, movements were more linear in corridor landscapes (clustered clearcuts) than in landscapes where single edges were present (dispersed clearcuts). Imbeau and Desrochers (2002) predicted that movement paths should be less sinuous at a scale larger than the maximum width of strips than in continuous landscapes. In our case, we found that the sinuosity of marten paths was lower in linear landscapes, at the two highest scales only (100-m and 200-m), which seems to confirm this hypothesis.

3.5. Conclusion

Overall distribution patterns of marten with regards to edge tended to follow the distribution of supranivean prey, especially snowshoe hare. Despite our main result showing that martens select edges, we showed that a barrier effect modifies their movements at least within the first 75-100 m of an edge. Therefore, cutting patterns (dispersed vs. clustered) may have a strong effect on the amount of forest interior conditions left for this species. Linear forest strips < 100-m wide provide highly used movement corridors in regenerating landscapes, but canalized marten movements. From an ecosystem management perspective inspired by natural disturbances (mainly wildfires in the region), such linear structures are not promoted (Perron et al. 2009). Our results suggests that corridors, even narrow ones, could represent an important source of connectivity for medium to large forest-dwelling species, like the marten. Consequently, these structures should be included in future management plans.

3.6. Acknowledgements

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Table 3.1. Overall density of marten, snowshoe hare and squirrel tracks as well as potential subnivean access points in the residual forests and clearcuts of the dispersed-cut (DISP) and clustered-cut (CLUST) landscapes.

		Tracks / km	
		DISP landscapes	CLUST landscapes
Marten	Forest	5.0	14.4
	Clearcut	0.6	0.6
Snowshoe hare	Forest	25.0	26.8
	Clearcut	4.1	2.8
Squirrels	Forest	8.4	8.6
	Clearcut	1.0	1.0
Potential subnivean access points	Forest	109.5	70.4
	Clearcut	16.3	13.2

Table 3.2. Mean parameter estimates obtained from linear regressions between movement angles relative to the nearest clearcut/forest edge of each marten paths and distance (log-transformed) to the edge (for the 0-100 m section only), in the dispersed-cut (DISP, N = 18 martens) and the clustered-cut (CLUST, N = 17 martens) landscapes. P-values were obtained with a one-sample Student t-test.

	DISP landscapes			CLUST landscapes		
Spatial scale	Parameter estimate ± S.E.	p	N steps	Parameter estimate ± S.E.	p	N steps
10-m steps	7.88 ± 2.18	0.002	703	-0.79 ± 1.75	0.659	877
20-m steps	9.87 ± 3.09	0.005	330	-0.28 ± 1.25	0.824	417
40-m steps	7.86 ± 2.17	0.002	154	-3.54 ± 0.96	0.002	194

Table 3.3. Mean sinuosity calculated for sections composed by five steps (5 x 10-m, 20-m and 40-m) and p-value of the linear mixed models, in the dispersed-cut (DISP) and the clustered-cut (CLUST) landscapes. The sinuosity of a straight line is given for each spatial scale as reference.

	DISP landscapes			CLUST landscapes			
	Mean sinuosity ± S.E.	No. sections	No. marten paths	Mean sinuosity ± S.E.	No. sections	No. marten paths	p
5 x 10-m	-3.72 ± 0.02	205	21	-3.75 ± 0.02	144	17	0.310
Straight line	*-3.91*						
5 x 20-m	-4.36 ± 0.03	94	21	-4.49 ± 0.04	65	16	0.009
Straight line	*-4.61*						
5 x 40-m	-5.08 ± 0.03	39	20	-5.18 ± 0.04	28	16	0.060
Straight line	*-5.30*						

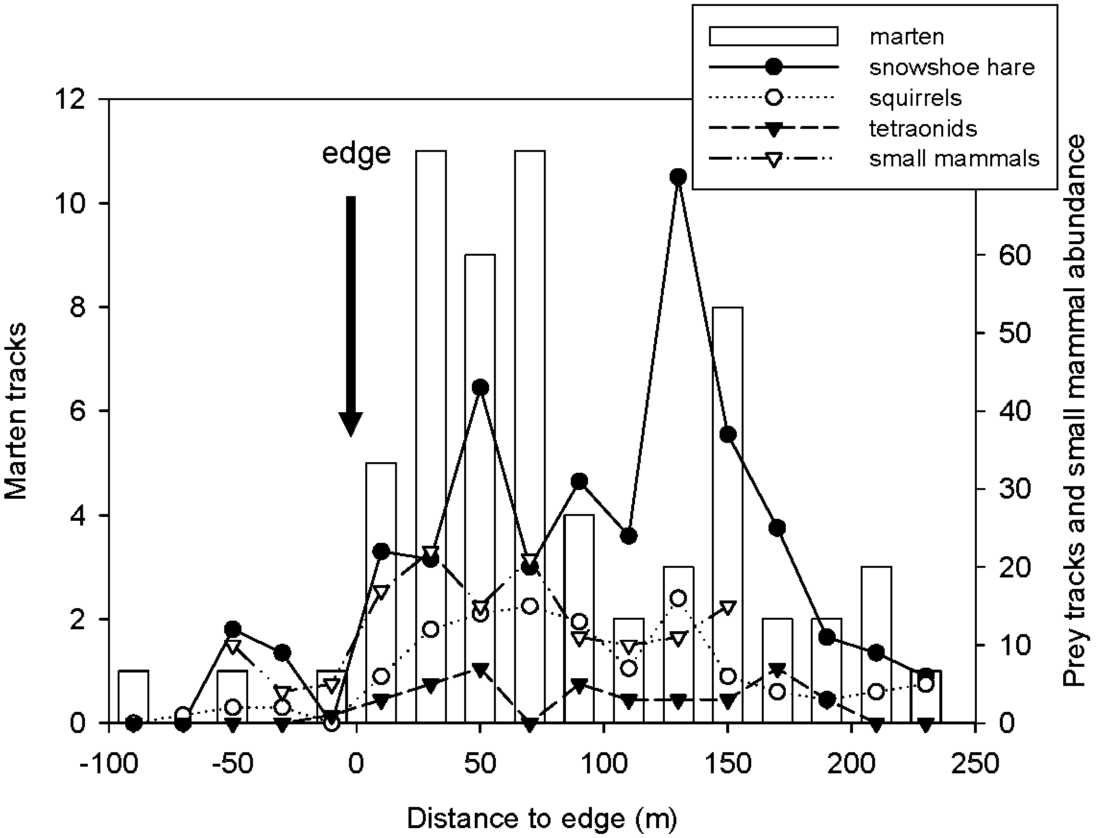

Figure 3.1. Relationship between the number of marten tracks and the distance to the nearest forest/clearcut edge, in dispersed-cut landscapes. Number of snowshoe hare, squirrel and tetraonid tracks are also shown (two winters pooled), as well as the small mammal abundance (number per 100 trap-nights, two summers pooled) as a function of the distance to the edge. Negative distances were within the clearcut and positive distances were within the residual forest.

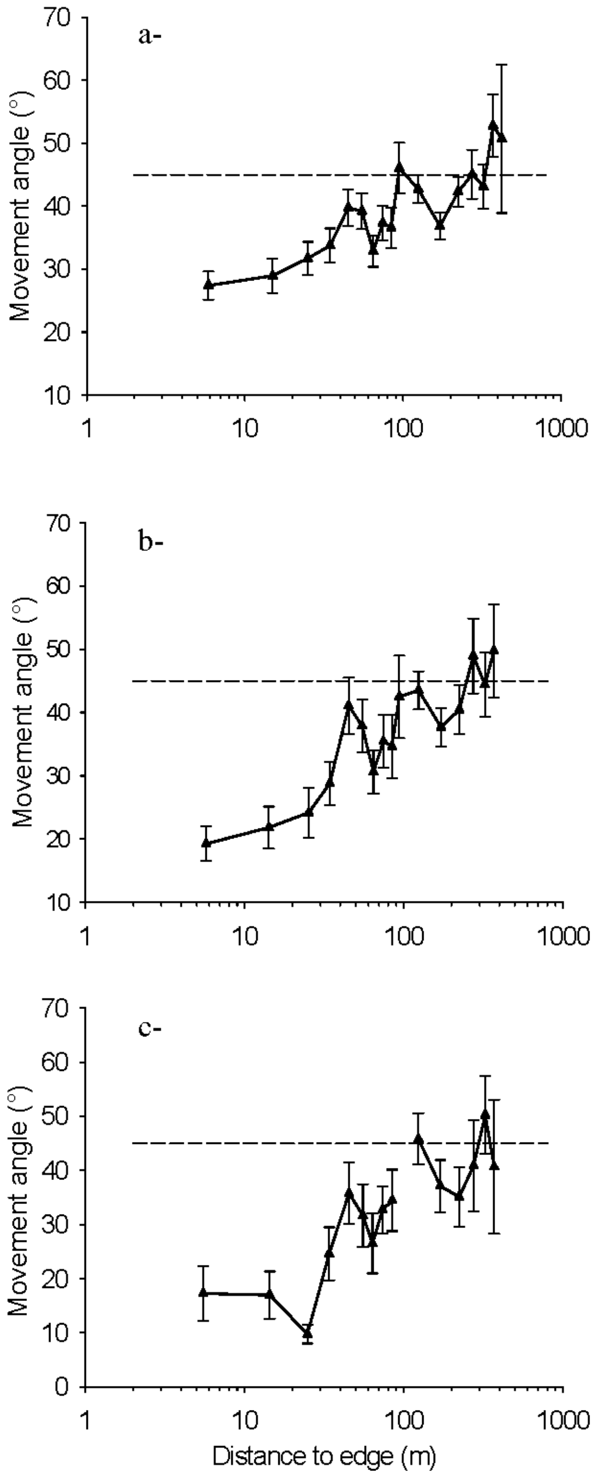

Figure 3.2. Mean movement angles of equidistant steps to the nearest forest/clearcut edge in relation to distance to the edge, for a- 10-m step lengths, b- 20-m step lengths and c- 40-m step lengths (mean ± S.E.), in dispersed-cut landscapes. The dashed line represents mean absolute random angle (|45°|).

CHAPITRE IV

BUILDING A COMMON VISION ON AMERICAN MARTEN BY INTEGRATING CREE KNOWLEDGE AND SCIENTIFIC FINDINGS IN MANAGED FORESTS OF NORTHERN QUEBEC

Marianne Cheveau, Louis Bélanger, Pierre Drapeau and Louis Imbeau

Résumé

L'implication des communautés autochtones dans l'aménagement forestier est maintenant un incitatif international de même qu'une obligation au Québec et au Canada. Pourtant, les processus ne sont actuellement pas encore très développés, et des adaptations et des améliorations sont encore nécessaires. Un des obstacles majeurs reste l'incompréhension entre deux visions du monde différentes parmi les intervenants. Dans ce contexte, intégrer les connaissances autochtones et les connaissances scientifiques pour atteindre un point de convergence entre ces deux perspectives peut mener à l'élaboration d'une vision commune, qui contribuera à établir des stratégies d'aménagement plus adaptées socio-écologiquement.

La martre d'Amérique (*Martes americana*) représente un symbole du mode de vie traditionnel Cri et est reconnue par les scientifiques comme une des espèces les plus sensibles aux perturbations d'origine anthropique en forêt boréale. Nous avons interrogé des trappeurs Cris de Waswanipi (à travers un groupe focus composé d'aînés, et de 16 entrevues individuelles semi-dirigées) dans le but de documenter leur vision de la martre et leurs préoccupations au sujet de l'aménagement forestier. Les participants ont décrit l'utilisation culturelle de la martre de même que leurs méthodes de piégeage. Ils ont aussi identifié les proies et les prédateurs des martres, ainsi que leur comportement reproducteur et leur préférence d'habitat. Finalement, ils ont rapporté les impacts de l'aménagement forestier sur l'abondance, la santé et la qualité de la fourrure des martres. Nous avons ensuite identifié les convergences et divergences entre les connaissances des Cris et nos résultats scientifiques concernant l'écologie de la martre, les préférences d'habitat et les impacts de l'aménagement forestier. Nous concluons que les deux sources d'information sont très concordantes, ce qui atteste que les efforts mis à développer une vision intégrée des connaissances Cries et scientifiques peuvent augmenter la compréhension mutuelle au-delà de la simple collection des savoirs écologiques traditionnels.

Mots-clés

Cris de Waswanipi, aménagement forestier, martre d'Amérique, connaissances autochtones

Abstract

The involvement of indigenous communities in forest management is now an international incentive as well as a national obligation in Quebec and Canada. However, current processes are not well developed, and adaptations and improvements are still needed. One of the major obstacles remains misunderstandings between two different worldviews between stakeholders. In this context, integrating indigenous and scientific knowledge to reach a converging point between the two perspectives could lead to the development of a common vision that will contribute to better socio-ecological adapted management strategies.

American marten (*Martes americana*) represents a symbol of the traditional Cree way of life and is recognized by scientists as one of the most sensitive species to human-induced disturbances in boreal forest ecosystems. We interviewed Cree trappers from Waswanipi (through a focus group, composed of elders, and 16 individual semi-directive interviews) in order to document their vision of marten and their concerns about forest management. Participants described cultural use of the marten as well as their trapping methods. They also identified marten prey, predators, reproductive behaviour and habitat preference. Finally, they reported impacts of forest harvesting on marten abundance, health and fur quality. We then identified convergences and divergences between Cree knowledge and scientific findings related to marten ecology, habitat preference and impacts of forest management. We conclude that the two data sources were highly concordant, attesting that efforts in developing an integrated Cree and scientific knowledge could increase mutual understanding beyond the single collection of traditional ecological knowledge.

Key words

Waswanipi Crees, forest management, American marten, indigenous knowledge

4.1. Introduction

National and international accords, such as the Rio Declaration and the Convention on Biological Diversity, encourage an active participation of local communities, including First Nations, in sustainable resource management (Cheveau et al. 2008, available as an appendix in this thesis). In forest management, the Forest Stewardship Council (FSC) and other certification agencies especially encourage participation of indigenous communities in co-management committees to obtain sustainable forest management labels. However, concrete community involvement and active participation of indigenous people to resource management is still mainly at the stage of pilot projects throughout the world (Cheveau et al. 2008).

In Northern Quebec, Cree peoples have legal rights in the local collaborative forestry regime (Adapted Forestry Regime) in their traditional territory with specific participation mechanisms (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002). However, Pelletier (2003) showed that the current process is not satisfying at two levels: recognition of Cree knowledge and encouragement of Cree stewardship. One of the major problems remains the conciliation of two fundamentally different worldviews (Stevenson and Webb 2004). Traditional ecological knowledge and community values should not only be gathered, but also incorporated into the forest planning process, which is rarely done currently (Stevenson and Webb 2003), except for some local initiatives (see Karjala and Dewhurst 2003, Waswanipi Cree Model Forest 2007 for examples). In order to harmonize forest management and the Cree way of life, it is essential to integrate Cree and scientific knowledge to reach a converging understanding between the two perspectives and then propose better socio-ecological adapted management strategies that will be recognized by both groups.

Cree people identify wildlife species of primary interest as they play an important cultural role, namely moose (*Alces alces*), beaver (*Castor canadensis*), marten (*Martes americana*), snowshoe hare (*Lepus americanus*), different species of fish, caribou (*Rangifer tarandus*) and grouse (*Falcipennis canadensis*, *Bonasa umbellus*) (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002). Moose represents the most important species for southern Cree communities (Feit 1987), whereas caribou occupies the same place for northern communities (Berkes 1999). Moose and caribou compose the largest part of food intake from wildlife harvests. Fish, hare and grouse complement food intake between the harvest of larger animals (Berkes 1999, Feit 2000). Marten, and in a larger part beaver, were historically a complement to subsistence hunting (Feit 2000). Moose was already the focus of a study in the Waswanipi traditional territory (Jacqmain 2008). The cultural importance of this species and related hunting strategies are widely documented by Feit (1987, 1992, 2000). Similarly, the importance of beaver in historical relationships between Crees and non-natives has generated considerable literature (Berkes 1998, Feit 1986, 2004, 2007). In contrast, the Cree view of marten has not been well documented.

In parallel, American marten also represents a focal species for forest biologists and managers, as the species is recognized as one of the most sensitive to human disturbances in forest ecosystems (Thompson 1991). Martens are known to prefer forests with a complex structure (Chapin et al. 1997), an attribute that is found mainly in old stands, which are also the focus of forest industry. Martens clearly avoid open areas and clearcuts, and cannot generally tolerate more than 30-40% of its home range in regenerating stands (Chapin et al. 1998, Payer and Harrison 1999, Potvin et al. 2000). Therefore, marten is considered to be a good indicator species of the functional role of older forest habitat at the landscape scale (Bissonette et al. 1989).

In this chapter, our first objective was to document the Cree vision of marten relative to its ecology and cultural use, and concerns on how this species responds to forest management. The second objective was to identify the degree of concordance (or discordance) between Cree knowledge and scientific findings to develop common knowledge on the impacts of forest management on marten in the eastern boreal forests of Canada.

4.2. Methods

4.2.1. Study area

Our study area was located in northwestern Quebec, eastern Canada (49°45'N 76°00'W), within the territory of the Waswanipi Cree Model Forest (WCMF, Fig. 4.1). The WCMF, created in 1997, was part of the Canadian Model Forest Network, which has aimed to develop, test and share solutions in sustainable forest management. The WCMF was the only Aboriginal-led Model Forest, and has been specifically established to minimize conflicts between Cree land users and forest managers through community involvement, participation, and knowledge sharing. Our study area was also within the boundaries of the *Agreement concerning a new relationship between le Gouvernement du Québec and the Crees of Québec* (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002), where forest management is subjected to an Adapted Forest Regime with forest cover retention ("mosaic cutting" strategy, described later) that is linked to Cree considerations for wildlife habitat conservation.

The area is located in the James Bay lowlands, which are characterized by mostly flat topography (with abundant poorly drained soils) with sparse hills. Natural forests are composed mainly of black spruce (*Picea mariana*) stands with an understory of feather moss, sphagnum and dwarf ericaceae shrubs. Isolated stands of mixedwood and deciduous forests are dispersed in the region, but represent < 10% of the landscape. Despite the fact that forest fires were the major natural disturbance on this landscape (Le Goff et al. 2007), forest harvesting has been the main source of disturbances over recent decades. Clearcuts have occurred since the early 1970's but extensive forest harvesting started mainly in the mid 1980's. The spatial pattern of clearcuts was either clustered (i.e., large cutover areas, several hundred km^2 in size) or dispersed in a checkerboard pattern (i.e., "mosaic cutting" strategy), following the two cutting strategies used in Quebec's boreal forest. The first pattern has been used since 1986 and the second since 1995 in the region. More precisely, the Waswanipi territory contains two types of dispersed clearcuts. The first is a "two-pass cut" which is now promoted through the agreement between the Quebec provincial government and the Cree First Nation (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002).

It aims to mitigate impacts of forest harvesting on the traditional activities of Cree peoples. The second is a “three-pass cut”, based on a ~ 30-yr rotation. This specific pattern was chosen by the community, as it conserves at all times a larger proportion of forest than occurs as clearcut. In this project, we did not consider the difference between the two dispersed clearcut types, however, Crees referred mostly to the second pattern during discussions. In both cases, only the first pass was realized. In the clustered-cut landscapes (50-200 ha cut-blocks), the residual forest was composed of narrow corridors 20-100 m wide along streams or between cut-blocks. In the dispersed-cut landscapes (10-80 ha cut-blocks), the residual forest was composed of uncut forest blocks equivalent in size to adjacent cut-blocks. The majority of the harvested part of the Waswanipi territory was clearcut using clustered cuts and two large zones (> 300 km^2) were harvested using dispersed cuts.

4.2.2. Waswanipi community

Crees have occupied the region for > 5000 years (Feit 1999). Except for the last 30 to 40 yrs, Crees have been mostly nomads, living in the forest from hunting, fishing and trapping (Marshall 1987). Waswanipi Crees are the most southerly inland Cree community in Quebec. The Hudson Bay Company installed a fur trading post on an island in Waswanipi lake, in 1819, which closed in 1965 (Marshall 1987). The village of Waswanipi was founded in 1976-1978 near the Waswanipi river (Marshall 1987) following signature of the *James Bay and Northern Quebec Agreement* in 1975 (Government of Quebec 1975). This agreement covered economic development and property issues (including land claims) in Northern Quebec, and established a number of cultural, social and governmental institutions for members of the communities involved in the agreement. The agreement also established an Income Security Program (ISP) for Cree Hunters and Trappers to encourage persistence of traditional activities. Currently, 1450 persons live in the village, where ~ 40% are < 15 yrs old and ~ 30% are > 35 yrs (Statistics Canada 2006), including ~15% still practicing subsistence hunting and trapping (CRA 1997). The WCMF territory is divided into 62 family hunting territories (named *traplines*) that are under the supervision of a steward (named *tallyman*) who decides whether the hunting territory should be harvested for big game and fur-bearers during the coming year, by whom and when (Feit 1992). The steward is usually an active elder, to whom the community recognizes as having an intimate knowledge of the land.

4.2.3. Community and researcher involvement

The overall project (named the *Wapistan project*, literally Marten project), including both scientific data acquisition and Cree knowledge collection, was approved by the community and represented a priority for the WCMF. The project was presented to the community in English and Cree translation during general assemblies, meetings and using posters in the Cree Trapper Association (CTA) office. During the project, the main researcher lived in the community and the territory for 235 days between June 2004 and June 2008, for field work, integration meetings and other community activities. Three Cree co-researchers were involved at different stages of the project, including agreements to obtain access to the territory for field work, cabin rental, public presentations and interviews. The ecological part of the project aimed to evaluate impacts of habitat loss and fragmentation created by forest harvesting on marten use of residual forests at the landscape scale (Cheveau et al. chapter 1), to characterize space use and habitat selection of marten (Cheveau et al. chapter 2) and to estimate marten response to forest/clearcut edges and use of forest strips (Cheveau et al. chapter 3). All scientific data were collected in the Waswanipi Cree Model Forest territory, the same territory where Cree knowledge came from.

4.2.4. Data collection

We collected data on topics related to the importance of marten in Cree culture, trapping methods, marten natural history (prey and predators, reproduction) and population trends, marten habitat preferences and effects of forestry (see Appendix 4.1). We first assembled a focus group (in the presence of an interpreter) with elders (N = 7; four men and three women) focusing principally on trapping methods, cultural use and natural history of marten (September 2007). Individual semi-directive interviews (N = 16) were also held in

September 2007 and in January-February 2008. Some of the participants were interviewed depending on their availability and willingness to participate but some were selected as recognized marten trappers in the community. We are aware that we do not have a representative sample of the Waswanipi community but instead we have a sample that more accurately reflects the Waswanipi marten trappers. Participants were mostly men of different age classes (<30 yr, 30-60 yr, >60 yr old, see Table 4.1). Because Cree trappers and hunters have benefited from ISP since 1975, if they spend most of their time in the forest, we classified participants using this status as an index of their land experience and wether they are the steward for their family hunting territory (i.e., tallyman) or not (Table 4.1). Their status as a trapper was also noted (no longer trapping, part-time or full-time trapper). Finally, participants were divided into three categories depending on the length of time their family hunting territory has been subjected to forest harvesting and silvicultural treatments (>20 yrs, 10-20 yrs, <10 yrs, Table 4.1) and the clearcut dispersion pattern they experienced (dispersed vs. clustered clearcuts). This information was mostly used to characterize our sample. Considering our small sample size, we did not compare information between the different attribute groups.

All participants voluntarily accepted to answer our questions and signed a consent form. Interviews were audio-taped when participants agreed and manuscript notes were taken. We used pictures of the different habitat types as support for discussions on marten habitat, as well as for perceptions on the different harvesting patterns (dispersed-cut and clustered-cut landscapes). The research protocol was approved by the institutional Ethics Committee for Research involving humans at UQAT (Université du Québec en Abitibi-Témiscamingue).

4.2.5. Data analysis

We first reviewed audio-tapes to verify that manuscript notes were complete. Secondly, notes collected during interviews and the focus group workshop were analyzed using the content analysis approach (Sabourin 2003). We used *NVivo 2* to code participant responses using the list of topics covered during individual interviews and the focus group workshop. We finally compiled the different answers for each topic. We observed that, during the last interviews, information reached a saturation level (Miles and Huberman 2003), when no new information was collected. Information collected during the focus group was treated separately and used as a validation source (triangulation process, Miles and Huberman 2003) for individual interviews, as elders represent an authority within the community. Degree of convergence between the two sets of information ensures the validity of data.

4.3. Results

4.3.1. Cultural use of marten

Martens are mainly used for commercial fur trade, as a source of income. Participants reported that in the old days they used to eat marten meat (N = 10), but this practice has now been abandoned (N = 5). Martens are trapped approximately from November to March (N = 5), but dates are flexible depending on weather conditions, such as first snowfall (October to December) and snow melt (March-April). Respondents mentioned that, usually, they stop trapping when the temperature is too warm because furs start peeling off (poorer quality, N = 5). Different traps were used over time. Traditional traps were made with logs. One design was based on the principle of a suspended snare (N = 1) and a second one was composed of different logs ("wooden trap") which fall on the animal when it approaches a bait (N = 4). Later, manufactured traps were primarily used, first leghold traps (N = 8) then Conibear traps (N = 9). Traps were set either on the ground (N = 10), under branches (N = 3) or snow (N =1) cover, or on a diagonally-fallen tree (N = 9). Recently, the use of plywood boxes around traps has appeared (N = 5). Traps are visited more or less frequently (varying between every day to once a week). Some trappers check for specific elements before setting a trap: marten tracks (N = 5), hare tracks (N = 5), or the presence of water (river, lake, or creek, N = 8). Conversely, some participants mentioned they just set a trap in the forest without any indices (N = 2) or specific habitat features (N = 3). Traps could be set right beside the road (N = 3), or in a short distance to be invisible from the road (~50 m, N = 2). The most used baits are fish (N = 6) as fresh (N = 3, whitefish, *Coregonus clupeaformis* or small northern pike, *Esox* *lucius*), smoked (N = 1) or canned (N = 1), and hare (skin or foot, N = 3). Trappers mostly hang bones and meat in a tree (N = 7), as a mark of respect for the animal, where carcasses would be eaten by crows or other birds (N = 3), then they either bury remains (N = 1) or burn them (N = 1). Some other participants threw carcasses in a dump site (N = 3) or use them as bait (N = 2). Marten pelts are sold either to the CTA (N = 4) or to other fur buyers (for example Fourrures Grenier in Barraute, N = 3). The majority of participants learned trapping methods from their parents when they were children (N = 8). More recently, younger respondents learned either by staying in the bush with parents after leaving high school (N = 3) or from Cree culture teachers (N = 2). Six participants recognized that people trap less than before, mostly because of lower fur prices (N = 7). Two respondents mentioned that it is now impossible to live on fur trapping, while another suggested that trapping income is now minimal compared to easily acquired ISP and compensation from forest companies.

Elders from the focus group mentioned that, historically, marten fur was used to make hats, coats, knitted and cuffs, but the most dominant use has been for commercial trade. They also reported that marten was eaten in the old days. Martens are mainly trapped from November until the temperature becomes too warm. Elders described in details old ways to trap marten, using a suspended snare and a wooden trap (mostly until 1940's). They explained how to install leghold and Conibear traps, on the ground with branch cover or on a diagonally-fallen tree. They also mentioned the recent appearance of plywood boxes. They compared different methods and agreed that the better ones were leghold traps. They complained that suspended snares leave a mark on marten fur. Despite the fact that Conibear traps were more powerful, they could miss the marten and they are hard to manipulate for an elder. Elders install their traps in the forest near a river or a creek, but do not need to see marten tracks. They use fresh whitefish or goldeye (*Hiodon alosoides)* heads as bait. They visit traps everyday or after two or three days, to be sure that no other animal could damage the fur (fox or mice). After preparing pelts, they hang the carcasses on trees as a sign of respect. They sell furs to the CTA, but they complain that the association does not pay enough and it is now impossible to live by trapping.

4.3.2. Marten ecology

Respondents considered that marten diet is composed of hares (N = 14), squirrels (N = 11), mice (N = 9), grouse (N = 6), other birds (N = 3), bird eggs (N = 1), weasels (N = 3), berries (N = 3) and plants (N = 1), fish (N = 2) and garbage (N = 2). Reported marten predators were fox (N = 6), raptors (N = 5), wolf (N = 4), lynx (N = 2), fisher (N = 2) and bear (N = 1). Maternal and resting dens were reported to be under the snow or the ground (N = 4), under tree stumps (N = 2), in a woodpecker hole (N = 1), on the top of a tree (N = 1) or under a rock (N = 1). Respondents suggested that marten reproduction season is in early spring (March-April, N = 5) and births occur in May-June (N = 2) or during the summer (N = 2). Some mentioned that litter size (number of young) depend on food availability (N = 2). Some participants noticed natural fluctuations in marten populations between years (N = 7), which follow hare fluctuations (N = 5), but some others did not notice any fluctuations (N = 3). In parallel, they identified a period of marten and all animal depletion, called “starvation time” (N = 4) as well as a period of animal abundance (N = 5), sometimes confounded with the period when they were trapping more (N = 3). However, they generally recognized that marten is less abundant nowadays (N = 8) than before.

According to elders from the focus group, marten diet is composed of hares, squirrels, mice and partridge. They mentioned that maternal dens are located under the ground in tunnels or in a woodpecker hole. Elders identified the rut season in February and young birth in July. They also noticed that marten litter size varies with food availability. Elders reported fluctuations in marten populations and they thought that it follows hare fluctuations. They mentioned that marten populations were never really high, and they were very low during the starvation time.

4.3.3. Marten habitat preferences

Participants first reported general characteristics of marten habitat, mentioning that marten is a forest species: marten occurs where trees are (N = 6), in old forests with big trees (N = 2) or in large forest blocks (N = 2). Conversely, they said that martens avoid open areas and clearcuts (N = 9). They also mentioned that martens select habitat where food is available (they mostly referred to hares, N = 7). Trappers stated that rivers and creeks are sought by marten (N = 11). Similarly, marten used blowdown zones where fallen trees are abundant (N = 6) and preferred forests with small trees and vegetation in the understory (N = 4) rather than without (N = 5).

Black spruce forests, the most abundant forest cover type in the region, represented the major habitat for marten (N = 9), and the best habitat according to three respondents. However, young black spruce forest was considered to be of lower quality than old forests (N = 2). Mixedwood forests are also recognized as good habitats (N = 7), and one of the best habitats with black spruce forests for one respondent. From the trappers' viewpoint, aspen forests are good habitats (N = 7), especially when hares are present (N = 2) and when aspen trees are mixed with conifers (N = 2). Opinions on the use of birch forests are variable, some participants mentioned that they are a good habitat (N = 4) and some not so good (N = 4). Similarly, jack pine forests are reported as good by some (N = 3), especially when trees are small because they represent food for hares (N = 2), or not (N = 2). Martens generally avoid recent clearcuts (N = 7), but trappers exceptionally saw marten tracks crossing one (N = 2). Some mentioned that even old clearcuts were avoided (N = 2), others reported that marten used these regenerating areas mostly for foraging activities (N = 5), but did not stay there (N = 7).

Elders from the focus group defined marten general habitat as an old forest. According to them, martens like to frequent creeks and use blowdown zones to install their resting dens. In terms of forest types, elders mentioned that black spruce and jack pine forests are marten habitat whereas aspen forests were not.

4.3.4. Impacts of forestry on marten

It is possible to identify three concerns about impacts of forestry on marten: marten abundance, marten health and commercial value of pelts. First, participants noticed that marten were less abundant (N = 11) since forestry began in the region and that they need to invest more effort to trap martens (N = 4). Some mentioned that marten migrate up north (reference to northern Cree communities, N = 6) in response to forestry (N = 3), where the forest is still untouched (N = 2). Second, they mentioned different impacts that could be related to the general health of individuals. They said that martens are hungry (N = 2), they travel more (N = 4) and they have fewer young because of food scarcity (N = 1). One trapper mentioned that now martens are caught more often in hare snare, since they are unable to break the snare because they are less strong than before. Similarly, some respondents recognized that martens are smaller (N = 2), have less fat (N = 1) and live less long than before (N = 1). Third, trappers complained about lower prices they obtain for marten pelts (N = 4) which are related to forestry: the fur is less dense and less shiny (N = 5) and has changed color (N = 8), being paler than before (which corresponds to lower price than dark-brown fur). They related fur color change to more exposure to the sun (forest opening by clearcuts), which "burns" the fur. Despite the fact that this complaint was widely spread within the community, which reports these impacts frequently during informal discussions, three respondents did not notice any difference in fur quality.

Others impacts of forestry were mentioned, including loss of the knowledge of the land (for marten and Cree people, N = 1), which is modified by forest harvesting, change in the taste of game animals (N = 1), difficulties related to wood (N = 1) and unpolluted water (N = 2) supply, pollution (N = 2) and changes in forest composition after clearcut (N = 1). Two participants expressed concerns about animal and Cree survival.

When asked about the restoration delay after a clearcut, participants provided highly variable responses. They predicted a return of marten when the forest cover regenerates (N = 5) or when hare come back (N = 7). In quantitative terms, responses were: after 10 yrs (N = 2), 20 yrs (N = 1), 30 yrs (N = 1), and even 100-200 yrs (N = 1). One respondent answered when the vegetation will reach 5 feet (1.5 m).

We specifically discussed participant perceptions related to the three landscape types found in the region: natural undisturbed landscape, clustered-cut landscape (conventional cutting) and dispersed-cut landscape (mosaic cutting). Regarding the natural landscape, participants mentioned that this landscape type represents the typical habitat for marten (N = 4), and particularly dark-brown martens (N = 1). Unanimously, dispersed clearcuts were judged as being better than clustered clearcuts (N = 6). Participants mentioned that marten will stay more in dispersed-cut landscapes (N = 1) and travel more in clustered-cut landscapes (N = 2). Except for general perceptions, we collected only few trappers' comments regarding dispersed clearcuts, mostly due to their limited experience with this kind of cutover dispersion (N = 3). Conversely, the large majority of respondents (N = 12) had a high experience with clustered clearcut patterns. Some trappers noticed that marten will use narrow forest strips (N = 7) in clustered-cut landscapes, but others judged strips too narrow for marten (N = 3). Six respondents acknowledged setting traps inside these forest strips, which are judged better when there is a creek in the middle (N = 3). In contrast, three trappers preferred setting traps in larger blocks.

We did not question elders about the impacts of forest harvesting on marten, but they spontaneously mentioned that marten color fades as a consequence of forest opening as the sun “burns” the fur.

4.4. Discussion

4.4.1. Cultural importance of marten

Compared to some other animal species, marten is not associated with a strong cultural use by Crees. Moose and beaver are really the key species in Cree culture because they are the major traditional food (Feit 1992) and are associated with historical periods of starvation and abundance (Marshall 1987). Similarly, black bear presents a particular role (almost sacred) in Cree culture, as it is considered to be a member of the family (Brightman 1993). Marten is instead a trade item and considered as an income source (Table 4.2). Indeed, marten has played an important historical role, as it was the second focus species (after beaver, Obbard et al. 1987) that drove the commercial relationship between Crees and the Hudson Bay Company. Whereas beaver represented historically 50% of fur income (1920-1930, Marshall 1987), it currently represents only 30% (Table 4.2, Quebec Ministry of Natural Resources and Wildlife). Despite a general decrease in fur trapping, we observed a recent change in the relative importance of marten and beaver in fur income, marten accounting for 53% during the past 9 years (Table 4.2).

Even though marten is mostly viewed as a “cash species”, the community showed an emotional reaction associated to disturbances affecting any used species, notwithstanding their relative cultural importance. Even a research project, like ours, created emotionality when issues such as marten capture and the use of radio-collars for research were discussed by the community. Despite the fact that trapping intensity is now relatively low, people complained that our activities caused disappearance of marten. Through discussions, we realized that marten represents a symbol of the Cree way of life. Contradictorily, Cree people have a holistic vision of the environment; all species are important, being part of the whole (Berkes 1999). For them, there is no prioritization among species.

In the Cree vision of the world, animals give themselves to hunters so that humans can survive in a principle of reciprocity (Berkes 1999, Feit 1999, 2000). This sacrifice obliges hunters to respect animals through different actions, which allow the rebirth of animal souls (Feit 1999) and success in future hunts (Feit 2000). Disposal of remains, like hanging bones on trees, as mentioned by participants, is part of the acts of respect and is common for different harvested species (Berkes 1999). Crees also interpret signs, such as the frequency of observations of animals and tracks, which are considered communications between animals, spirits and trappers (Feit 1987). These signs inform them about animal population trends and influence their future hunting and trapping efforts. Some species are recognized to respond to management decisions (i.e., level of harvest, for example moose and beaver), but some do not (for example hare and grouse, Feit 1987). Cree management systems for moose (Feit 1987), beaver (Feit 1986, Berkes 1998) and fish (Berkes 1999) are well documented. We did not question participants about their traditional marten management system, and no indications were given either. The Cree marten management system is possibly quite similar to the one for beaver, which is frequently based on a rotational use of different zones within hunting territories (Feit 1986). One zone is used during one year or one season, while the others are in fallow in order to let the population grow (Berkes 1998). Berkes (1999) and Berkes et al. (2000) show that Cree management systems, based on a detailed knowledge of the ecosystem, are remarkably convergent with Adaptive Management (Stankey et al. 2005).

Waswanipi Crees experienced numerous socio-economic changes over the last 35 years, including the implementation of the ISP in 1975 (Government of Quebec 1975), a substantial increase in the availability of cash income and the consequent use of modern transportation and weapons that increase hunting efficiency, an increase in the accessibility of the territory (for Crees and outsiders), an increase in the rate of forest harvesting on hunting territories, and a rapid increase in the Cree population (Feit 1992). All these changes could have influenced hunting and trapping pressure on the land base, as well as their traditional management systems. For example, current effects of forestry represent a confounding element in trappers' interpretation of trends in animal populations and harvest levels, which disrupt their traditional management of wildlife (Feit 1992), in addition to modifying their detailed knowledge of the land (Feit 1986).

4.4.2. Marten ecology

In the context of the ecology portion of this project, we investigated marten diet from faeces (N = 61) collected during the capture sessions of September through November 2006 (Cheveau et al. chapter 1). Marten diet during the fall was composed of 36% berries (American mountain-ash, *Sorbus americana*), 33% different species of small mammals (mice, voles and shrews), 13% squirrels (red squirrel, northern flying squirrel and chipmunk), 9% snowshoe hare, 8% birds and 1% weasel (Mandin et al. *unpublished report*). However, diet is known to vary between seasons and regions (Martin 1994). In eastern North America, the most frequently reported items composing the diet of marten are similar to our findings and Cree responses: snowshoe hares, small mammals, squirrels, birds and berries (Bateman 1986, Thompson and Colgan 1990, Simon et al. 1999, Cumberland et al. 2001, Robitaille and Laurence 2007).

As part of the project, we installed radio-collars on marten during the winters of 2005-2006 and 2006-2007 (Cheveau et al. chapter 2). In case of deaths, carcasses were found and cause of death was evaluated by necropsy when possible. We were able to determine that seven martens were probably killed by a predator: five by a raptor, one by a fox and one by an undetermined terrestrial predator (Cheveau, *unpublished data*). In the literature, we found references to raptors, lynx, fox and fisher as potential marten predators, for species that are present in the region (Thompson 1994, Hodgman et al. 1997, Bull and Heater 2001). Scientific information and Cree responses were thus quite concordant.

We also found three marten dens (potentially maternal dens) during the field work. All these dens were located underground: two under tree roots and one between rocks (Cheveau, *unpublished data*). Seven collared martens were also found dead in dens under the snow (N = 3) or under the ground (mostly under tree roots, N = 4). The most often reported structures in the literature for resting and denning sites are tree cavity in snags and standing live trees as well as under stumps and logs, but all from western North America (Raphael and Jones 1997, Ruggiero et al. 1998). Only few data come from eastern North America, but we suspect that in our study area, where mature trees remain quite small, tree cavity of sufficient size for marten are rare, which could explain why marten used underground dens. Our observations match Cree trappers' responses whereas the majority also reported underground resting sites.

We did not investigate marten reproductive behaviour during the project. However, Buskirk and Ruggiero (1994) reviewed marten reproductive biology. Mating seems to occur mostly in July, but marten, like other mustelids, undergoes a delayed pregnancy. Foetus development stops for 7-8 months and active gestation lasts only 1 month. The only reference that we found in the literature for Quebec was reported in Strickland et al. (1982) from a fur farm, where parturition dates of martens were between March 28 and April 17. Strickland and Douglas (1987) estimated that marten produce an average of three young per female per year, but fecundity is related to prey availability, as reported by trappers. Some participants mentioned that maternal dens and marten young are really difficult to find, which limits the amount of information concerning reproduction. Apart from a divergence between Cree and conventional science observations about dates of the mating season, all other information is consistent.

Cycles in game animal abundance are part of the Cree worldview, which is related to the willingness of animals to be hunted, including marten (Berkes 1999). Animals such as marten are said to go under the water or underground during periods of rarity, but the belief in their return is very strong (Berkes 1999). Marten fluctuations could be detected from time series, for example from fur sales. Fur sales could however be influenced by factors other than marten abundance, including environmental conditions (for example weather and prey availability) and trapping effort, which is itself influenced by fur prices (as suggested by Table 4.2) and social context (Strickland 1994). Nevertheless, fur sale data could provide a broad index of population fluctuations over decades when other population monitoring data do not exist. Historical archives reported two periods of animal scarcity (and starvation for Crees) in Waswanipi history (Marshall 1987), the first was related to large fires occurring just before the turn of the XX^{e} century and the second was consecutive to the Depression and the 2nd World War (1930-1950). We suggest that the time of all animal depletion mentioned in interviews ("starvation time"), which was reported mostly by elders (based on their own experience and information from their parents), refers to the 1930-50 period, while the period of marten abundance was more recent (mentioned by younger trappers based on their own experience), possibly in 1980's.

4.4.3. Marten habitat preference

Marten is known to be a forest specialist (Buskirk and Powell 1994) that avoids open areas (Drew 1995, Potvin et al. 2000) throughout its geographic range. Thompson and Harestad (1994) showed that one of the most important elements that guide marten habitat choice is prey availability, which was also reported as a key factor by Cree respondents. Typical marten habitat is characterized by forests with a complex vertical and horizontal structure (Payer and Harrison 2003). Cree participants also reported that fallen trees, small trees and understory (elements of a complex forest structure) are part of a good habitat for marten. For Crees, black spruce and mixedwood stands were the best habitats for marten. We investigated marten winter habitat use in the study area. Our field results showed that the most used habitat by marten were mixedwood forests followed closely by older black spruce forests (Cheveau et al. chapter 2). Conversely, the least used habitats were recent clearcuts and open areas like bogs, while old clearcuts were used moderately. For their part, trappers mentioned that recent clearcuts were avoided, while old clearcuts were only used punctually, for foraging activities.

4.4.4. Impacts of forestry on marten

Because this project (ecology portion) investigated the effects of forest harvesting on marten in the Waswanipi territory, we could compare Cree trappers' observations with our scientific findings. The first impact that Cree trappers mentioned was a reduction in marten abundance following forest harvesting. We also found that marten abundance was two-fold higher in unharvested than in harvested landscapes of the study area (Cheveau et al. chapter 1). In parallel, participants reported that martens from harvested zones presented different characteristics that suggest lower health. For Crees, the fat content of killed animals represents an index of their health, which is often verified after a successful trapping and is part of the information they use in their future decisions concerning land and wildlife management (Berkes 1999). We also found that marten body condition (i.e., index of general health) was lower where the surrounding proportion of clearcuts was higher (within 5-10 km^2, Cheveau et al. chapter 1). Finally, trappers complained that marten pelts were of lower quality since forest companies began their activities in the region. As mentioned by respondents, fur quality could be influenced by individual marten health (shininess, hair density) and fur color, which was noticed to be paler in harvested zones. We also investigated changes in fur color during our field work. Pale martens were captured in sites with a higher proportion of clearcuts than were reddish-brown and dark-brown martens (Cheveau et al. chapter 1). Pale marten furs were sold for lower price than were dark-brown furs (Obbard 1987).

Waswanipi trappers did not equally experience forest harvesting, some being impacted for the past 30 years, while others experienced clearcuts only recently (< 10 years). This influences their personal knowledge on the delay of animal return after clearcut. We did not investigate restoration delay for martens in the study area. However, according to participants, marten return is highly correlated to snowshoe hare use of regenerating clearcuts. In the same study area, Jacqmain et al. (2007) found that the height of the regenerated forest cover after clearcut varies depending on forest composition. Moreover, they showed that regenerating stands after clearcuts are suitable for snowshoe hare when they reach 4 m in height (from Waswanipi hunters' interviews and scientific data). It was evaluated that this height was reached at about 25 to 30 years in black spruce regenerating stands and 10 to 15 years in balsam fir regenerating stands. This difference could partly explain the variability in trappers' responses regarding the restoration delay (in terms of years). Hence, in addition to their experiences with forest harvesting, the location of the participants' family hunting territory will also influence his/her experience in terms of restoration delay, northern hunting territories being dominated by black spruce stands while southern ones are composed of a larger part of balsam fir and mixedwood stands.

Cree trappers all judged that dispersed clearcuts (i.e., mosaic cuts) were better than clustered clearcuts (i.e., conventional cuts), despite the fact that they did not necessarily experience them personally in their territory. From our field data, we found only few advantages for marten in dispersed-cut landscapes compared to clustered-cut landscapes. Marten body condition (i.e., general health) was higher in dispersed-cut than in clustered-cut landscapes (Cheveau et al. chapter 1). Moreover, dark-brown martens showed a tendency to be more frequent in dispersed-cut than in clustered-cut landscapes, and conversely, pale martens were more frequent in clustered-cut than in dispersed-cut landscapes (Cheveau et al. chapter 1). However, marten abundance (Cheveau et al. chapter 1), home range size and habitat selection (Cheveau et al. chapter 2) were similar in the two landscape types. Nevertheless, movements were more constrained by higher density of forest/clearcut edges in clustered-cut landscapes than in dispersed-cut landscapes (Cheveau et al. chapter 3). For now, landscapes harvested using dispersed clearcuts maintained more forest habitat than clustered clearcuts, because cut-blocks are usually smaller and because approximately half of the landscape was uncut. However, according to the current Cree-Quebec Agreement (Government of Quebec and Grand Council of the Crees (*Eeyou Istchee*) 2002), forest companies will harvest the remnant proportion of stands when regeneration in clearcuts reaches 3 or 7 m in height, depending on the location. It was not clear in our investigation if Cree participants considered this second harvest or not.

Trappers mentioned that marten used narrow forest strips retained in clustered-cut landscapes, and recognized that they installed traps in them. As part of the project, we found that martens used narrow forest strips in the study area as movement corridors (Cheveau et al chapter 3). Marten track density was even three times higher in these corridors than in large patches. We suggest that this higher intensity in the use of narrow forest strips by marten could be explained by a concentration of activities considering their avoidance of clearcuts (Cheveau et al chapter 3).

4.4.5. Integration of the two knowledge system sources

Information from Cree knowledge and scientific findings on marten in the Waswanipi region are highly concordant. For most of the scientific knowledge, we referred directly to field data collected in the Waswanipi Cree traditional territory and not to the literature. This allowed having a better grasp on marten responses to current forest conditions induced by forest management activities in the Waswanipi land base. We suggest that the fact that both knowledge sources came from the same study area contributes to this strong match between Cree knowledge and scienctific data. In a context where Crees and forest managers often experienced a lack of mutual understanding, we showed that Cree and scientific knowledge could be easily bridged because they are not so different. This exercise clearly attests that efforts in developing an integrated Cree and scientific knowledge could increase mutual understanding and then favour acceptability of management guidelines based on a common vision.

The main divergence we noticed between Cree and scientific knowledge concerned the marten mating season, which could easily be explained by the difficulty in observing marten in this specific context. Cree trappers observed more marten movements in February-March, which they interpreted as the mating season. We do not question this observation, despite the fact that reproduction does not occur during this season. This behaviour should instead be investigated in order to understand why marten are apparently highly mobile at this specific moment. We suggest that such a mismatch could serve to develop future hypotheses and future research for scientists, as proposed by Jacqmain et al. (2008).

The integration of the two knowledge sources (Cree and scientific) could lead to an improvement in the practice of local wildlife management and contribute to empower community knowledge holders. Despite the benefits of co-management, Nadasdy (2005) identifies potential unintended consequences. From his experience, co-management is often restricted to technical questions (how to gather and incorporate traditional knowledge into management processes), and trapped in existing structures which excludes political and ethical considerations. Moreover, true participation is rare (Cheveau et al 2008), power relations staying unchanged and community participation being used to legitimize decisions (Nadasdy 2005).

4.5. Conclusion

This project was only the first step in a potential process for the incorporation of indigenous knowledge in forest management and community involvement in forest planning. We have not validated wether the common vision established here for marten could successfully improve mutual understanding between Cree land users and forest managers in the Waswanipi territory. The next step would be to transmit our research results to co-management committees (Joint Working Groups) and collect feedback regarding their contribution to co-management process. However, because Cree people have a holistic vision of the forest, specific forest management guidelines for marten must be integrated into a larger habitat management strategy for the hunting territories.

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Table 4.1. Participant attributes. Sex designed men (M) and women (F). Zone designed the northern sector (N) were forest is mostly composed by black spruce and the southern sector (S) were black spruce forests interspersed with white birch-balsam fir mixed forests.

Participant	Sex	Age	On ISP[1]	Tallyman	Trapper's status	Zone	Forestry experience	Clearcut pattern[2]
001	M	30-60 yrs	no	no	-	S	>20 yrs	clustered
002	M	30-60 yrs	no	no	part-time	S	>20 yrs	clustered
003	F	>60 yrs	yes	no	part-time	S	>20 yrs	clust./disp.
004	M	30-60 yrs	yes	yes	part-time	N	<10 yrs	-
005	M	30-60 yrs	yes	yes	-	N	10-20 yrs	clust./disp.
006	M	>60 yrs	no	no	-	N	<10 yrs	-
007	M	<30 yrs	no	no	-	N	<10 yrs	clustered
008	M	30-60 yrs	yes	no	part-time	S	>20 yrs	clustered
009	M	>60 yrs	yes	no	part-time	N	>20 yrs	clustered
010	M	<30 yrs	no	no	-	N	>20 yrs	clustered
011	M	30-60 yrs	?	no	part-time	S	>20 yrs	clustered
012	M	30-60 yrs	yes	no	full-time	N	10-20 yrs	clustered
013	M	>60 yrs	no	no	-	-	-	-
014	M	30-60 yrs	no	yes	full-time	N	10-20 yrs	clust./disp.
015	M	>60 yrs	yes	yes	part-time	N	>20 yrs	clustered
016	M	30-60 yrs	no	no	-	-	-	-

[1] Income Security Program

[2] clustered or dispersed clearcuts

Table 4.2. Trapping statistics for the Waswanipi territory, compiled by the Quebec Ministry of Natural Resources and Wildlife (zone 88) for the 2000-2009 period.

	Number of sold furs			Mean price		Generated income		
Year	Beaver	Marten	Total[1]	Beaver	Marten	Beaver	Marten	Total[1]
2000-2001	958	1204	2509	27,16 $	36,22 $	26 019 $	43 609 $	80 276 $
2001-2002	728	685	1596	24,53 $	45,03 $	17 858 $	30 846 $	59 700 $
2002-2003	581	370	1197	22,96 $	41,88 $	13 340 $	15 495 $	41 660 $
2003-2004	865	613	1815	26,00 $	43,62 $	22 490 $	26 739 $	67 206 $
2004-2005	874	998	2171	24,18 $	50,57 $	21 133 $	50 469 $	85 206 $
2005-2006	739	802	1741	35,06 $	90,70 $	25 909 $	72 741 $	108 793 $
2006-2007	668	384	1300	26,02 $	59,30 $	17 381 $	22 771 $	48 250 $
2007-2008	386	324	811	23,52 $	55,77 $	9 079 $	18 069 $	30 458 $
2008-2009	222	188	457	17,11 $	43,23 $	3 798 $	8 127 $	13 679 $
Mean[2]	46%	40%	-	-	-	30%	53%	-

[1] from all species

[2] for the 2000-2009 period

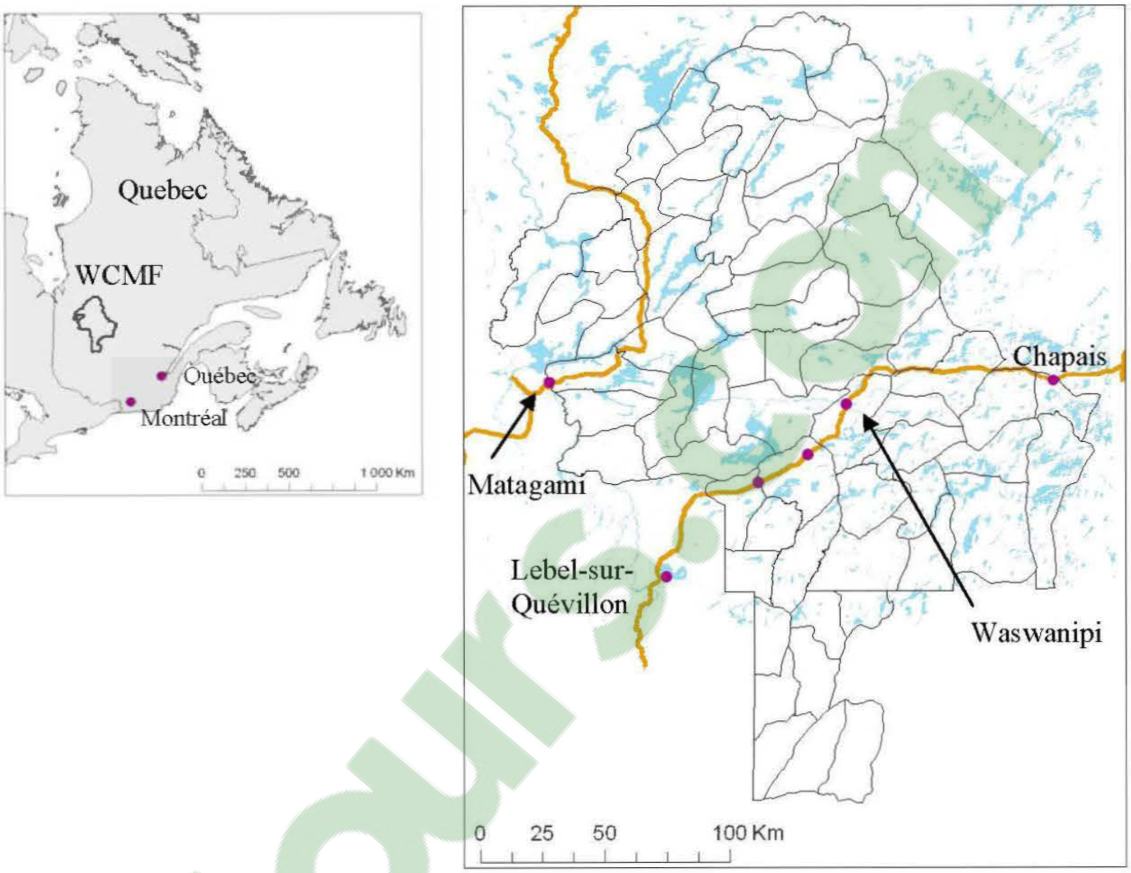

Figure 4.1. Waswanipi Cree Model Forest study area composed of the 62 traditional hunting territories of Waswanipi Crees (300-1300 km²).

Appendix 4.1. Topic and picture list for individual semi-directive interviews.

Marten in Waswanipi history and culture

- Marten in Waswanipi history and culture
- Revenues from marten
- What can be done with marten pelt and other parts?
- Did ancestors always trap marten?
- Traditional uses of marten
- Appropriate way to treat marten (pelt, bones, meat, etc.)
- What does the marten eat?
- Who eats the marten?
- Stories from old days
- Marten life:
    - What does the marten need to survive (food, cover, water, tranquillity, dens)?
    - When does the marten make babies?
    - Where does the marten have their babies?
    - How long do babies stay with mother?
    - Do they stay in different places in winter and summer?

Trapping marten

- Full time on ISP or occasional trapping
- If no longer trapping marten, why?
- Marten trapping:
    - Month
    - How you chose the spot to set up trap (look for tracks before)?
    - Method (on tree, on ground, cabin with fir), bait, lure?, what kind of trap (leghold traps or Conibear rotating-jaw traps)?
    - In line or one by one
    - Checking traps (how often?)
    - When setting traps: distance from the road, from the shore, from camps etc…

- Where do you sell pelts?

Trends in marten populations (causes)

- Pelt quality, size, colour
- Trends in marten population (short and long term)
- Trends in marten prey (rabbit, squirrels, ptarmigan, small mammals)
- Trends in marten predators (which are marten predators?)

Where to set up traps for marten

- What kind of forest do you need to set up your trap (need of river, creek, type of trees, dead wood, etc…)? ***pictures of different forest habitats***
- Does the marten go in the buffers?
- If you had forestry on your trapline, did you change the way you set up your traps?
- Looking at pictures, which cutting is worse for the marten? ***pictures of different landscapes***
    - Do you install traps in clear-cuts residual blocks?
    - If you know mosaic cutting, do you think the blocks are big enough?
    - How long before the marten comes back after cutting (or how high, and what kind of trees/brush)?
    - Do you set up traps in buffers?
    - If yes, is it better when there is a creek?

Picture list of forest habitats:

- Paper birch forest
- Trembling aspen forest
- Mixedwood forest (paper birch, trembling aspen and black spruce)
- Jack pine forest
- Black spruce forest:
    - young (~50 yrs)
    - old (~120 yrs)
    - with vegetation and small trees in the understory

        - without vegetation in the understory
        - with fallen trees
- Alder zone near a creek
- Recent clearcut (~ 1 yr)
- Old clearcut with regeneration (~ 15-20 yrs)

Picture list of forest landscapes:

- Natural landscape (unharvested)
- Harvested landscape with conventional clearcuts (clustered)
- Harvested landscape with mosaic clearcuts (dispersed)

CONCLUSION GÉNÉRALE

5.1. Synthèse des principaux résultats

La martre d'Amérique est considérée comme une espèce sensible à la perte et à la fragmentation de son habitat issues des perturbations anthropiques, telles que l'exploitation forestière (Thompson 1991, Thompson 1994, Chapin et al. 1998, Payer et Harrison 1999, Potvin et al. 2000). Cependant, nous avons montré dans cette thèse que la martre semblait plus tolérante en forêt boréale résineuse que ce que les études plus méridionales ont pu montrer jusqu'à maintenant.

5.1.1. Effets de la perte d'habitat sur la martre

Selon Fahrig (1997, 2003), l'impact de la perte d'habitat est de loin plus dommageable pour les populations animales que l'effet de la fragmentation. Ainsi, si l'on se restreint à la seule perte d'habitat, la martre s'y est montrée sensible à l'échelle du paysage (100-300 km^2, chapitre 1). En effet, son abondance a diminué de moitié dans les paysages aménagés qui présentaient une plus grande proportion de « non-habitats », par rapport à un paysage naturel (non récolté). Par contre, la perte d'habitat, (ici les coupes forestières) n'a eu qu'un effet faible à plus petite échelle (5-10 km^2), et ce, tant sur l'occupation des sites (effet marginal), l'abondance (diminution minime) que sur la condition physique des martres (principalement sur les mâles adultes, chapitre 1). De la même manière, nous n'avons pas détecté de relation entre la taille des domaines vitaux et la proportion de coupes à l'intérieur de ceux-ci (chapitre 2). Au contraire, les martres de notre région d'étude semblaient même tolérer davantage de coupes à l'intérieur de leur domaine vital que dans d'autres régions puisque 35% des femelles dépassaient le seuil de tolérance à la coupe communément accepté dans la littérature (30-40%, Chapin et al. 1998, Fuller et Harrison 2000, Potvin et al. 2000). Par contre, nous avons noté un effet de la proportion de coupes sur la couleur des fourrures des martres; les martres pâles étaient plus présentes dans les sites ayant une plus forte proportion de coupes (chapitre 1). Cet effet n'a cependant qu'une répercussion sur la valeur des fourrures pour les trappeurs (chapitre 4), puisque la couleur de la fourrure n'est pas liée à la condition physique de l'animal.

Nous n'avons par contre pas de données sur la productivité de la population (nombre de jeunes produits par femelle). Bien que la proportion de jeunes dans la population soit plus importante lors de la 2^e année d'inventaire (c'est-à-dire en automne 2006), et ce, malgré le fait que la nourriture disponible ait été faible durant la période de reproduction précédente, il est impossible de savoir si les jeunes capturés ont été produits dans les paysages échantillonnés ou dans des secteurs adjacents servant de source de population. Ainsi, il est impossible de conclure si la persistance de la population de martres dans les paysages étudiés est basée sur une production locale de jeunes ou alimentée par une dynamique source-puits (cf Van Horne 1983, Pulliam 1988).

5.1.2. Effets de la fragmentation de l'habitat

Les effets de la fragmentation de l'habitat, c'est-à-dire de la configuration spatiale des habitats, n'entrent en ligne de compte qu'au-delà d'un certain seuil de perte d'habitat (Andrén 1994). Ce seuil se situerait entre 60 et 80% de perte d'habitat (Andrén 1994), mais est spécifique à chaque espèce (Andrén et al. 1997). Bien que les paysages étudiés n'aient pas atteint un tel niveau, celui-ci se situant plutôt entre 40 et 60% de perte d'habitat, nous avons pu détecter certains effets de la configuration des coupes forestières (dispersées vs. agglomérées) sur la martre. En effet, à l'échelle des domaines vitaux (5-10 km^2, chapitre 1), la condition physique des martres était meilleure dans les paysages de coupes dispersées par rapport aux paysages de coupes agglomérées. Nous avons aussi observé une tendance (certes non significative) de la dispersion des coupes sur la couleur des fourrures : les martres pâles étaient proportionnellement plus nombreuses dans les paysages de coupes agglomérées, alors que les martres foncées étaient, elles, proportionnellement plus nombreuses dans les paysages de coupes dispersées (chapitre 1). De plus, le type de lisières forêt/coupe créées par la dispersion des coupes (simples vs. multiples) avait aussi un effet différent sur les déplacements des martres. En effet, l'angle de déplacement était plus petit (mouvement plus parallèle) lorsque la martre se rapprochait d'une lisière simple (effet barrière), alors que l'inverse se produisait dans le cas de lisières doubles (effet de canalisation, chapitre 3). Par contre, à l'échelle du paysage (100-300 km^2, chapitre 1), l'abondance des martres ainsi que la taille des domaines vitaux (chapitre 2) n'étaient pas influencées par la dispersion des coupes. La sélection d'habitat de la martre était, elle aussi, similaire dans les 2 types de paysages, tant à l'échelle des domaines vitaux qu'à l'échelle du paysage (chapitre 2). Par ailleurs, aucun seuil n'a été détecté tant à l'échelle des paysages qu'à celle des domaines vitaux (chapitre 1).

5.1.3. Évitement des coupes et effet des lisières

L'évitement des différents types de milieux ouverts, tels que les coupes, est bien connu chez la martre et peut être associé à un comportement de diminution du risque de prédation (Drew 1995, Smith et Schaefer 2002), mais aussi à la faible abondance de leurs proies préférées dans de tels milieux (Thompson et Colgan 1994, chapitre 3). Nous avons confirmé cet évitement à différentes échelles, tant à travers la sélection d'habitat des martres que lors du suivi des déplacements. En effet, les coupes récentes et les dénudés humides, ainsi que dans une moindre mesure les coupes en régénération, se sont révélés les habitats les moins utilisés par les martres à l'intérieur de leurs domaines vitaux (chapitre 2). De la même manière, les martres concentraient leurs activités dans des zones où la quantité de coupes (surtout récentes) était minime (kernels, chapitre 2). Plus finement, les martres s'aventuraient très peu dans les coupes au cours de leurs déplacements hivernaux et s'éloignaient rarement de la lisière (chapitre 3). D'ailleurs, les lisières forêt/coupe représentaient, en quelque sorte, des « frontières » pour les martres. En fait, les pistes de martres présentaient une légère tendance à être plus nombreuses près des lisières (effet non significatif, chapitre 3). L'effet barrière des lisières (simples) se faisait sentir sur une distance d'environ 75 à 100 m de la lisière, zone où les martres se déplaçaient de façon plus linéaire et plus parallèle à la lisière que lorsqu'elles étaient plus profondément dans la forêt (chapitre 3). Les martres concentraient aussi leurs activités dans des zones où la densité des routes et des lisières forêt/coupe était plus faible (kernels, chapitre 2). De même, nous avons montré que l'abondance des martres à l'échelle des domaines vitaux (5-10 km^2) était négativement influencée par la densité des lisières (chapitre 1).

5.1.4. Utilisation des forêts résiduelles

Étant donné que la martre évite les coupes, elle concentre donc ses activités dans les forêts résiduelles. En effet, les martres utilisaient toutes les forêts résiduelles, peu importe la composition de l'habitat environnant dans une superficie de 5 à 10 km² (analyse d'occupation de sites, chapitre 1). Seules les martres adultes utilisaient en plus grande proportion les sites avec une plus grande quantité de forêts mixtes (effet marginal en 2006, chapitre 1). Cet attrait pour les forêts mixtes s'est confirmé dans la sélection d'habitat par les martres à la fois à l'échelle des domaines vitaux et à l'échelle du paysage (chapitre 2), et ce, dans les deux types de paysages (coupes agglomérées et dispersées) même si la quantité de forêts mixtes variait dans ces paysages (8% vs. 25%). Les forêts mixtes étaient les habitats les plus recherchés par les martres, devant les forêts résineuses de plus de 120 ans et les forêts résineuses de 70 et 90 ans, qui pourtant représentaient à elles deux plus de 75% du paysage (chapitre 2). D'ailleurs, la taille des domaines vitaux des femelles diminuait lorsque la proportion de forêts mixtes augmentait (chapitre 2), caractéristique désignant habituellement un habitat de qualité optimale (McLoughlin et Ferguson 2000). Nous émettons l'hypothèse que c'est la structure de ses peuplements qui en fait de bons habitats pour la martre, lui procurant un couvert contre les prédateurs, une bonne disponibilité en proies (notamment des lièvres, *Lepus americanus*) et des sites de repos. Par ailleurs, les martres concentraient leurs activités dans des zones où la proportion de forêt résiduelle, de même que de forêt d'intérieur, était plus importante (kernels, chapitre 2). Dans les paysages de coupes agglomérées, les martres utilisaient intensivement les corridors forestiers (60-100 m de large) où l'on observait une plus grande densité de pistes que dans les blocs de forêts (effet de concentration des déplacements, chapitre 3). La condition physique des femelles était, quant à elle, associée positivement à la quantité de forêts résiduelles dans leur domaine vital (marginalement significatif, chapitre 2).

5.1.5. Impact de la disponibilité de la nourriture

Bien qu'il ne s'agissait pas du sujet principal de cette thèse, la disponibilité des proies représentait un effet confondant potentiel entre les différentes années du suivi. En effet, les lièvres présentent des cycles d'abondance d'environ 10 ans dans la région (Krebs et al. 2001), alors que les petits mammifères, particulièrement le campagnol à dos roux de Gapper (*Myodes gapperi*), présentent, eux, des cycles de 4 ans (Cheveau et al. 2004). Durant la durée de cette thèse, les lièvres sont restés à des abondances relativement faibles (Assels et al. 2007). Par contre, les petits mammifères ont connu un creux d'abondance en 2005 suivi d'une augmentation substantielle en 2006 (chapitre 1). Bien que l'abondance et la condition physique des martres n'aient que peu changé entre les deux années d'investigation (2005 et 2006, chapitre 1), la structure d'âge de la population était, quant à elle, très différente au cours des deux automnes de captures (39% de juvéniles en 2005 et 64% en 2006). L'occupation des sites a aussi été affectée par la disponibilité des petits mammifères. En 2005, aucune variable ne semblait expliquer l'occupation des sites par les martres, indiquant que les martres avaient possiblement modifié leur sélection d'habitat (chapitre 1). De plus, les distances de dispersion, surtout chez les juvéniles, étaient aussi plus grandes en 2005, ce qui explique la probabilité de capture (détection) plus élevée lors de cette année où la disponibilité des proies était faible (chapitre 1). Par ailleurs, les domaines vitaux étaient près de trois fois plus grands en 2005 (résultats non présentés, N = 4, moyenne MCP100 = 14,4 km^2). Malgré l'impact des fluctuations d'abondance des petits mammifères sur les populations de martres, le lièvre semble lui aussi jouer un rôle important pour la martre. En effet, la distribution des pistes de martres à l'approche des lisières semblait suivre la distribution des pistes de lièvres (chapitre 3). D'ailleurs, Martin (1994) suggère que les lièvres représentent une part importante de l'alimentation hivernale des martres dans l'est de l'Amérique du Nord, alors que Thompson et Colgan (1990) suggèrent que les petits mammifères complètent leur alimentation entre les captures de lièvres.

5.2. Implications en aménagement forestier

Si on tente de mettre en perspective nos résultats dans le cadre de la réglementation actuelle de l'aménagement forestier, tant au niveau de la loi sur les forêts que dans le contexte de l'Entente Cris-Québec, il est possible d'anticiper les défis à venir auxquels feront face les aménagistes forestiers et les Cris. Notons par ailleurs qu'au moment d'écrire ces lignes, le gouvernement du Québec vient d'adopter un nouveau projet de loi (PL-57, Loi sur l'aménagement durable du territoire forestier) qui réforme le régime forestier du Québec.

Rappelons que l'Entente Cris-Québec impose le maintien en tout temps de 30% de forêts de plus de 7 m à l'intérieur des territoires de chasse familiaux (300-1300 km² à Waswanipi). Dans le cadre de cette thèse, nous avons analysé la réponse des populations de martres dans des paysages déjà largement coupés. Cependant, ils présentaient tous encore 40 à 60% de forêts résiduelles de plus de 7 m. Bien que nos résultats montrent que la martre se maintient dans les paysages étudiés (à des abondances certes plus faibles), nous n'avons pas investigué la réponse de la martre au niveau de 30% prévu dans l'entente Cris-Québec. Il nous est donc impossible de statuer si cette mesure sera suffisante ou non à maintenir l'espèce à des abondances acceptables pour les Cris qui tirent des revenus du piégeage de la martre (chapitre 4). Par ailleurs, nous avons noté que la condition physique des individus était affectée négativement par la quantité et la dispersion des coupes. Ainsi, cet effet pourrait augmenter avec la proportion de coupes dans le paysage et entraîner des répercussions sur la survie des individus et, en corollaire, des populations.

L'utilisation des coupes en mosaïque est prônée à la fois par la loi sur les forêts et par l'Entente Cris-Québec. Les Cris y sont largement favorables (chapitre 4); en grande partie parce qu'elle permet le maintien de plus de forêts dans le paysage. D'un point de vue écologique, l'intérêt des coupes en mosaïque sur les coupes conventionnelles agglomérées s'est révélé assez ténu pour la martre. Les seuls gains que nous avons recensés sont liés à la condition physique des martres (ce qui n'est tout de même pas négligeable) et à la couleur de la fourrure, ce qui représente toutefois des éléments importants aux yeux des trappeurs. Ainsi, alors que la pertinence des coupes en mosaïque semble remise en question actuellement, leur intérêt réside peut-être dans leur plus grande acceptabilité sociale pour les Cris. Il est aussi possible qu'à long terme ce type de coupe mitige l'effet de l'accumulation de coupes sur la condition physique des martres. Toutefois, il est utile de rappeler que les coupes en mosaïque que les Cris apprécient sont les « coupes à trois passes » qu'ils ont choisi d'instaurer sur leurs terres (catégorie I) plutôt que les « coupes à deux passes » mises de l'avant par la loi sur les forêts. Par ailleurs, il n'est pas clair si les trappeurs rencontrés avaient conscience de la réalisation d'une 2^e série de coupes ni de comment celle-ci serait faite (2 ou 3 passes). Alors que nous n'en sommes encore qu'à la 1re passe, il est temps de se questionner sur les règles qui régiront cette 2^{e} passe. Quelle quantité de forêt résiduelle sera maintenue? Sous quelle forme (configuration spatiale)? Suivant le principe de filtre brut (Hunter 1999), l'aménagement écosystémique s'inspire des patrons de paysages post-feu pour guider la rétention après coupe (Perron et al. 2009). Dans ce contexte, les corridors de déplacements ne seraient pas maintenus. Cependant, nos résultats ont montré leur importance dans la connectivité des paysages pour la martre (chapitre 3), mais cela a également été démontré pour d'autres espèces animales (Leboeuf 2004, Gagné 2006). Ainsi, la rétention de telles structures linéaires pourrait être ajoutée aux stratégies déjà envisagées (Drapeau et Imbeau 2006).

Lors de la signature de l'Entente Cris-Québec (2002), les Cris ont demandé la protection des peuplements feuillus et un aménagement adapté spécifique pour les peuplements mixtes (art. 3.11). Aucune mesure n'a été prise depuis. Bien que les peuplements mixtes et feuillus ne soient pas ciblés par la récolte forestière, ils ne sont pas non plus considérés lors de la planification de la récolte. Or, ces peuplements pourraient représenter des « hot spots » de biodiversité dans les paysages boréaux résineux, tant en termes de diversité végétale, animale que structurale. Nous savons qu'outre la martre qui recherche les peuplements mixtes (chapitre 2), l'orignal (*Alces alces*), espèce emblématique pour les Cris, restreint sa sélection d'habitat à ces seuls peuplements durant l'hiver (Jacqmain et al. 2008), de même que les lièvres qui y trouvent la majeure partie de leur nourriture (Jacqmain 2003). Il est alors évident que le maintien d'une connexion entre les îlots de forêts mixtes et la matrice forestière résiduelle est indispensable à la circulation des espèces animales. Ainsi, il serait souhaitable d'inclure, lors de la planification de la récolte, une stratégie de rétention des peuplements mixtes et feuillus bien connectés.

5.3. Perspectives futures de recherche

L'accumulation de résultats montrant l'importance des rares forêts mixtes en forêt boréale résineuse suggère que ces peuplements présentent un caractère distinct, et ce, possiblement en raison de leur structure forestière complexe acquise plus tôt dans leur développement. Dans notre secteur d'étude, les vieux peuplements résineux possèdent également une structure complexe, accumulant du bois mort (chicots et débris ligneux) et s'ouvrant pour permettre aux espèces de sous-bois de croître. Cependant, il faut attendre de nombreuses années avant de voir cette structure se développer (~ 120 ans, Harper et al. 2005). Les peuplements résineux plus jeunes (50-90 ans) ont souvent une structure plus simple, avec une canopée fermée, un sol recouvert de mousses et de sphaignes et un sous-bois très peu diversifié (Harper et al. 2005). Les forêts mixtes acquièrent une structure complexe beaucoup plus tôt dans leur développement (Bergeron 2000). Il serait pertinent d'étudier la spécificité des forêts mixtes en forêt boréale résineuse, tant au niveau de leur structure que de la composition de la communauté végétale et animale (notamment en termes de proies) qui s'y rattache.

Pour la martre, une étude plus approfondie de l'utilisation de l'habitat permettrait de mieux comprendre l'importance des corridors forestiers quant à leur rôle écologique pour l'ensemble des activités de celle-ci. Ainsi, bien que nous ayons montré que la martre les utilise intensivement, nous n'avons pas d'indication de la raison de sa présence dans de telles structures linéaires. Les corridors sont-ils utilisés aux seules fins de déplacements entre deux blocs de forêts ou la martre y chasse-t-elle? Lors de nos suivis de pistes (22 km), nous n'avons pu identifier qu'un site de capture d'une proie par une martre. Il faudrait donc augmenter significativement l'effort (surtout la longueur des suivis de pistes) afin d'obtenir des indications sur le comportement de chasse de la martre. Parallèlement, l'avènement de nouvelles technologies, telles que les colliers GPS miniatures, ouvre des perspectives immenses vers une meilleure compréhension de la sélection fine de l'habitat, notamment à l'aide de localisations plus fréquentes (plusieurs par jour et par nuit) et plus nombreuses.

D'un point de vue sociologique, il serait intéressant de vérifier, par un suivi de la question, si la création d'une vision commune (Cris-scientifique) sur la martre et les impacts de l'aménagement forestier sur cette espèce peut améliorer la compréhension mutuelle des différents intervenants impliqués dans la gestion collaborative de la forêt sur le territoire de Waswanipi. De même, nous pourrions évaluer si cette approche contribue à améliorer la satisfaction des intervenants en débouchant sur des mesures d'atténuation concrètes.

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ANNEXE I

CURRENT STATUS AND FUTURE DIRECTIONS OF TRADITIONAL ECOLOGICAL KNOWLEDGE IN FOREST MANAGEMENT: A REVIEW

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Résumé

Au cours des 25 dernières années, la quantité d'études traitant des savoirs écologiques traditionnels (SET) a constamment augmenté, atteignant maintenant plus de 200 publications par an. L'objectif de cette revue de littérature était de déterminer comment ces connaissances sont utilisées aujourd'hui pour l'aménagement forestier à travers le monde, et comment les populations locales sont impliquées dans la planification de l'aménagement forestier. Tous les articles traitant des SET, publiés entre 1983 et 2005, ont été extraits à partir de la base de données *ISI Web of Science*. Malgré l'abondance de littérature publiée sur les SET, seulement 30 études abordaient spécifiquement l'aménagement forestier comme tel. Dans ces études, l'intégration du SET pouvait prendre différentes avenues : s'inspirer des règles ancestrales de gestion (5 études), adapter les pratiques dans le temps et l'espace en fonction des valeurs à prioriser (3), ou en divisant le territoire en zones à vocations différentes basées sur l'utilisation des terres (6). L'engagement des communautés pouvait être « actif », à travers des comités de cogestion composés des différents utilisateurs de la forêt (incluant des membres de la communauté), chacun ayant une voix; ou « passif » lorsque des aménagistes extérieurs utilisaient des critères et indicateurs développés à partir des valeurs et objectifs de la communauté. Cependant, d'importants changements de mentalité et des décisions politiques fermes seront encore nécessaires avant de voir une participation entière des communautés et une incorporation effective des SET et de la science occidentale dans la planification forestière. Il semble, par ailleurs, que plusieurs initiatives canadiennes présentent des processus prometteurs pour s'assurer de l'incorporation des savoirs traditionnels et pour instaurer une gestion participative.

Mots-clés

Savoirs écologiques traditionnels, SET, aménagement forestier durable, gestion participative, autochtones, planification de l'aménagement forestier

Abstract

In the last 25 years, the number of published studies that refer to traditional ecological knowledge (TEK) has constantly increased, with now more than 200 papers published each year. The objective of this review was to determine how this knowledge is used in current forest management around the world, and how local communities are involved in forest management planning. Published papers from 1983 to 2005 relating to TEK were found using the *ISI Web of Science* database. Despite the growing amount of literature published on TEK, we found only 21 studies that specifically address forest management *per se*. In these studies, TEK integration took different paths: using traditional management rules as a framework (5 studies), using value maps to adapt practices in time and space (3), or by a zoning process that divides the land into areas in which different land uses are emphasised (6). Some community involvements are "active" with co-management committees composed of stakeholders (including community members), each having a voice; some are "passive" with external managers using criteria and indicators previously developed from community values and objectives. Although important changes in mentality and firm political decisions are still required before more efficient partnership between TEK and western science is reached in forest management planning, our review showed that Canadian initiatives proposed promising processes that could ensure better TEK incorporation and improved community participation.

Key words

Traditional ecological knowledge, TEK, sustainable forest management, community participation, first nations, forest management planning

A1.1. Introduction

The use of traditional ecological knowledge (TEK) in resource and environmental management is a fairly recent development. Although this knowledge was long recognised by ethnologists, its utility and relevance to applied management appeared only in the early 1980s (IUCN Symposium on TEK 1982, TEK Working Group 1984). Moreover, it is only since the publication of *Our common future* by the World Commission on Environment and Development (UNWCED) (1987), which encouraged the use of TEK to solve problems in modern resource management, that TEK finally received its currently widely accepted international recognition (Johnson 1992).

The most commonly accepted definition of traditional ecological knowledge is as follows:

> "Traditional ecological knowledge or TEK is a cumulative body of knowledge and beliefs, handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment. Further, TEK is an attribute of societies with historical continuity in resource use practices; by and large, these are non-industrial or less technologically advanced societies, many of them indigenous or tribal" (Berkes 1993).

Ericksen and Woodley (2005) added that traditional ecological knowledge may or may not be indigenous (in this case named indigenous ecological knowledge, IEK), but in both cases has roots firmly in the past, contrary to local knowledge (LK) which is also place-based experiential knowledge, but not necessarily old (i.e. farmers or fishermen) (Joshi et al. 2004). Otherwise, the term TEK remains controversial because it could be interpreted as old and static (Lewis 1993). On the contrary, traditional knowledge is highly dynamic and cumulative, and although based on the experience of previous generations, it is verified at each new generation, but added to and adapted to meet present socio-economic and technological changes by adaptive processes and trial-and-error learning (Baines and Hviding 1993, Johnson 1992). Tradition further implies historical continuity of culture and knowledge over many generations.

Traditional People, who could be defined as people living a subsistence lifestyle, close to nature and natural resources, have developed a nested worldview, synthesized by Berkes (1999) in the *knowledge-practice-belief complex*, where knowledge of the land, animals and plants is embedded in a management system with land use practices, which are sustained by social institutions, all being defined by particular ethics and beliefs. Contrary to some scientists' assumptions (Howard and Widdowson 1996), western science views on land use and resource management are also rooted in a culture and system of thought (based on reductionism, experimentation, reason and rationality); consequently, they are not completely value-free (Stevenson 1997).

These fundamentally different worldviews have been one of the major causes of conflict regarding land use and management between managers, scientists and government on one hand and traditional communities on the other hand. However, all systems of human knowledge are created by similar processes and are more alike than a focus on their apparent differences may suggest (Feit 1988). Moreover, western models of knowledge, have also changed between centuries, from a dichotomy of organic and inorganic world to a vision where animals and habitat are incorporated in a complex system. When ecological theories were based on climax equilibrium and maximum sustainable yield, TEK was firstly regarded as superstition by most scientists, and consequently of no practical use for current management issues (Howard and Widdowson 1996).

Recently, new paradigms that introduce concepts like complexity, system dynamics and resilience have led scientists to widely accept new approaches such as adaptive and sustainable management (Holling 1986, Holling et al. 1998). Western scientists now encourage the preservation of ecosystem integrity, even after resource exploitation (Regier 1993), which is considered as the fundamental principle of ecosystem-based management (Hunter 1999). These approaches also include a broader view of the connections between ecosystems and societies, which encourage more public involvement, especially where traditional Peoples are connected to the ecosystems in which they live. These new concepts appear to be reducing the distance between science and traditional ecological knowledge (Davidson-Hunt and Berkes 2001). However, if the concept of knowledge is understood with regards to how we know (instead of what we know) through people relationships with their environment, and not as an objective truth, then there is a common ground to enable multiple perspectives to contribute to ecosystem management (Woodley 2005).

Traditional Peoples' ecological knowledge systems are based on survival and success in hunting, fishing, gathering and trapping, which ensure a sustainable use of resources. Local land use practices allow communities to remain economically and socio-culturally viable, while at the same time ensuring that there is a stable foundation for the well-being of future generations. This understanding of sustainability is notably similar to definitions expressed by the Brundtland Commission (UNWCED 1987) and Agenda 21 (UNCED 1992). The need for traditional knowledge incorporation and community involvement in resource management is recognised by the Rio Declaration, the Convention on Biodiversity and different national strategies around the world. The New Forest Code (1994, cited in Poissonnet and Lescuyer 2005) in Cameroon, the Strategy for the Involvement of communities and voluntary agencies in the Regeneration of Degraded Forests in India (1990, cited in Martin and Lemon 2001), as well as the Canadian National Forest Strategy Coalition (2003) are recent examples of this worldwide trend. In forest management, the Forest Stewardship Council (FSC) and other certification agencies especially encourage participation of indigenous communities in co-management committees to obtain sustainable forest management labels.

Sustainable forest management can be defined as maintaining and enhancing the long-term health of forest ecosystems for the benefit of all living things, both nationally and globally, while providing environmental, economic, social and cultural opportunities for the benefit of present and future generations (CCFM 2000). This definition implies respect and protection of the ecosystem *per se* (species and natural processes), but also its multiple uses (timber and wildlife harvesting, non-timber forest products, recreational activities, indigenous traditional activities). Because traditional Peoples living in forest ecosystems are highly dependent on non-timber forest products (NTFP), research on these products has often been run in parallel with studies on TEK and forest management. Collectively, these studies provide a privileged arena of investigation in which ecology, TEK, and timber and NTFP harvesting can be brought together in a sustainable forest management framework (Davidson-Hunt and Berkes 2001).

Considering the recent recognition of TEK in forest resource and environmental management, how are such considerations currently integrated into forest management planning? In this paper, we propose 1) a review of case studies around the world that try to integrate TEK and which involve communities in forest planning and management, and 2) a synthesis of different processes that have been developed to promote efficient participation of communities and inclusion of their knowledge in sustainable forest management.

A1.2. Methods

In this study, we do not intend nor pretend to do an exhaustive review of all TEK-related work which would include reports and unpublished theses. Our intent was specifically to review published case studies, easily available worldwide, which justify our choice to use a recognized database of peer-review papers published in scientific journals. We therefore used the *Web of Science ISI* online database provided by Thomson Scientific, and we generated a list of published papers from "*Social Science Citation Index", "Arts and Humanities Citation Index"* and "*Science Citation Index (Expanded)"* between 1983 and 2005 (done on January 31th, 2007), which included the terms "*traditional knowledge"*, "*indigenous knowledge"*, "*local knowledge"*, "*environmental knowledge"* or "*ecological knowledge"* as *Title* or *Topic* (1805 papers). These databases cover over 8,700 leading journals of science, technology, social sciences, arts, and humanities. From this bibliography, we extracted articles with *"forest"* or *"forestry"* in the *Title* or *Topic* (222 papers). If we define forest management both as the planning of use and the use of the forest for any purpose (wood or NFTP harvesting, hunting, spiritual activities, etc), only 21 articles were specifically related to forest management *per se*. We analysed documents from Asia, Africa, South and North America, with special attention given to how TEK was collected (gathering methods) and how it was really integrated into management decisions. We also evaluated community involvement level (when possible) using Berkes' (1994) levels of co-management: (1) informing, (2) consultation, (3) co-operation, (4) communication, (5) advisory committees, (6) management board and (7) partnership of equals/community control.

There is no "universal" recognized keyword to drive our research in the database, thus some interesting papers could indeed have been missed (for example those using "community-based forest management" or "participatory forest management"). We also acknowledge that literature published in languages other than English/French were not considered in our review.

For the synthesis of different processes enhancing efficient participation of local communities in forest planning and management, we specifically analysed three Canadian initiatives from papers published in scientific journals (Karjala and Dewhurst 2003, Karjala et al. 2004) and also pertinent chapters (Stevenson and Webb 2003, 2004) or reports (Stevenson 2005, Pelletier 2002, Waswanipi Cree Model Forest 2007) (gray literature) not listed in the *Web of Science ISI* database.

A1.3. Results

Since 1983, 1805 papers on TEK were published according to our initial search within the *Web of Science ISI* database. Publishing rates have constantly increased in the last 15 years with more than 25 papers per year since 1991, 50 per year since 1995, 100 per year since 1998, and more than 200 per year since 2003 (Fig. A1.1). Among these contributions, 12% (222 papers) contained forest-related issues, for example, knowledge on natural disturbances (Kovacs et al. 2004) or wildlife (Dahl 2005), medicinal plant and non-timber forest product (NTFP) uses (Pakia and Cooke 2003a, b), among others.

A1.3.1. TEK gathering

Nineteen of the retained 21 documents presented a case study, in which a majority (15) gathered TEK. In these studies, only two did not describe the methodology that was used to collect community knowledge or values, even if they mentioned that they created land use and occupation maps (Polansky and Heermans 2004, Sekhar 2004). Eight studies used open-ended or informal interviews to collect traditional knowledge or to gather perceptions of stakeholders involved in management committees. Six were based on group discussions, five were based on participant observations, three were based on non-participant observations, one used questionnaires, three produced value or objective maps, and finally, six created traditional land use and occupation maps (Table A1.1). Some described in detail the subjects that were tackled during discussions and interviews (Dei 1993, Herrmann 2005, Karjala and Dewhurst 2003, McCall and Minang 2005, Silvano et al. 2005, Wickramasinghe 1997), but others not at all (e.g. Sekhar 2004). Five studies also compared traditional knowledge with scientific data collected in the field (Herrmann 2005, Polansky and Heermans 2004, Robiglio and Mala 2005, Silvano et al. 2005, Wickramasinghe 1997).

In order to improve Iranian forest management plans, Ghazanfari et al. (2004) used only participant observation to document traditional management practices to increase local community acceptance. Herrmann (2005) used both participant observation and open-ended interviews to document indigenous traditional management of *Araucaria araucana* forests in Chile, the cultural role of this species and perceived risks linked to its exploitation (seed gathering). Moreover, he specifically discussed with participants how TEK could contribute to forest management. In Sekhar's (2004) study, the goal was to compare tree species chosen for planting by local populations in India considering religious, social and economic factors with choices made by scientific experts. She conducted non-participant observations and open-ended interviews, but did not give details on treated subjects. In Wickramasinghe's (1997) case study, group discussions and participant observation were used to document traditional use of forest resources in Sri Lanka and reasons for opposition to an integral conservation project developed after an important degradation of the local forest. In Thailand, Roth (2004) used only focus group interviews with key informants in a similar case of local opposition to a conservation park, particularly in relation to fixed geographical borders which are not a traditional way of viewing the land. Dei (1993) studied human causes of local deforestation in Ghana using participant and non-participant observations, as well as interviews on traditional harvesting, myths and taboos related to forests, and land use for timber harvesting and agriculture. Natcher et al. (2005) used the same approach (participant and non-participant observations as well as interviews) to evaluate satisfaction of stakeholders involved in a committee for co-management of fish, wildlife and other renewable resources in Yukon (Canada). In a study documented by Silvano et al. (2005), the Brazilian government wanted to reforest grazing land, in order to protect an important river (for water supply) and create a corridor between two forest areas. The authors used a questionnaire to evaluate local perceptions on land degradation (ecological integrity of the river, water quality and forest ecosystem services). Farmers acknowledged advantages of the surrounding forest ecosystem for livestock, but they were not interested in more protection, because they did not recognize their responsibility in land degradation. Comparisons between local knowledge and scientific data have shown that, for example, farmers have underestimated the issue of water quality degradation in this river basin. This suggests that we need to be cautious and adequately verify information before using it in management strategies (Davis and Wagner 2003).

A second group of articles focused on land use or value maps as tools that should incorporate TEK. In fact, these maps are not outputs but spatial representations of TEK, traditional practices and values. Polansky and Heermans (2004) evaluated the potential use of different high-technology tools (satellite maps, aerial photos, GIS) associated with TEK to improve forest management in Zambia. From panchromatic aerial photos, simplified maps were created, in which land use information (non-timber forest products, agriculture, and zone limits) was added with the collaboration of the local population (no details concerning their methodology). Robiglio and Mala (2005) conducted a similar study in Cameroon, using GIS and TEK (collected from focus group discussions and field trips) to create traditional land use and occupation maps. Karjala and collaborators (Karjala and Dewhurst 2003, Karjala et al. 2004) carried out open-ended interviews and meetings with focus groups in order to generate land use and occupation maps including values, needs and perspectives from an indigenous community using criteria and indicators, in a context of conflict between commercial forestry and traditional rights of First Nations in British Columbia (Canada). McCall and Minang (2005) used the same approach (open-ended interviews, focus group discussions) to realize land use maps. Furthermore, they used criteria and indicators developed with the community to evaluate governance of a Cameroon communal forest. Natcher and Hickey (2002) used all available methods to realize land use maps using criteria and indicators developed with an indigenous community in Alberta (Canada). Robinson and Ross (1997) proposed also to create land use maps using interviews to gather indigenous values and objectives in Alberta (Canada), with numerous validations during the process. Using open-ended interviews and priority value maps, McGregor (2002) evaluated satisfaction level of different forest users (indigenous and non-indigenous) who participated in consultations regarding forest planning in Ontario (Canada).

A1.3.2. TEK incorporation

Different types of information (traditional management practices, land use, and values) that are contained in TEK may be incorporated into forest management. Traditional management practices could inspire modern management (Berkes et al. 2000) to reduce conflicts between stakeholders (Herrmann 2005), or simply because they have often been sustainable for thousands of years (Stevenson 1997). Five studies proposed to incorporate some traditional management rules into modern forest management plans (Ghazanfari et al. 2004, Herrmann 2005, Roth 2004, Sekhar 2004, Wickramasinghe 1997). Traditional knowledge of land use can also be helpful in creating a zoning of the land base to share land between different users or adapt forestry practices for other uses at specific moments. Six studies proposed such an approach (Dei 1993, McCall and Minang 2005, Natcher and Hickey 2002, Polansky and Heermans 2004, Robiglio and Mala 2005, Robinson and Ross 1997). The integration of local values and objectives can also be considered as a means to involve communities in forest management, using criteria and indicators, value maps or both. This can lead to generating land use maps that incorporate spatial and temporal traditional uses. Three studies proposed this approach (Karjala and Dewhurst 2003, Karjala et al. 2004, McGregor 2002).

More than half of the studies incorporated TEK in a forest management process (Table A1.2). A project was aimed at establishing a new forest planning process that would incorporate TEK ultimately (Ghazanfari et al. 2004), whereas for others, the integration of TEK is already underway, although at different levels. Some studies have collected information on traditional management practices or traditional land uses, but this information is not yet or not efficiently incorporated within forest management plans, but could be so in the near future (Dei 1993, Polansky and Heermans 2004, Wickramasinghe 1997).

A1.3.3. Community involvement

Six studies presented no community involvement in forest management (Dei 1993, Gautam and Watanabe 2002, Herrmann 2005, Polansky and Heermans 2004, Silvano et al. 2005, Wickramasinghe 1997) (Table A1.2). For example, Silvano et al. (2005) collected local knowledge to propose reforestation strategies, but the local community did not take part in the development of these forest management strategies. In Herrmann's (2005) study, it was not clear how TEK was incorporated into forest management, although the Chilean government encouraged the local community to reforest clear-cuts with cultivated *Araucaria araucana* (*ex situ* conservation). The local community certainly was involved in conservation of this endangered species, but not really in its *in situ* management. In Wickramasinghe's (1997) study, the government of Sri Lanka decided to preserve a forest without accounting for local population dependence on forest resources. Thompson (1999) presented a synthesis of different pilot projects in social forestry in Indonesia, but local community involvement is not described in detail. Nevertheless, it included gathering TEK (on forest and on traditional management) and it was a participatory process.

It is difficult to draw generalities from case studies as they are each at different stages of development; these include: (a) a call to involve local communities (Dei 1993, Gautam and Watanabe 2002, Herrmann 2005, Wickramasinghe 1997), (b) the development of a future participation process (Ghazanfari et al. 2004, Natcher and Hickey 2002), (c) presentation of an active participation process (Natcher et al. 2005), (d) evaluation of an active participation process (Karjala et al. 2004, Martin and Lemon 2001, McCall and Minang 2005, Polansky and Heermans 2004, Robiglio and Mala 2005, Roth 2004), (e) evaluation of stakeholder satisfaction (McGregor 2002), and (f) a comparison of different approaches that incorporated TEK (Klooster 2002). Furthermore, the autonomy of involved communities and the level of their assimilation into modern ways of life are different. In addition, the ecological, social and political context of each study is specific, including projects of local development using forestry, conservation or restoration projects that are in conflict with traditional subsistence gathering, agriculture or agroforestry, together with projects of commercial timber harvesting, which limits indigenous rights.

Levels of co-management (sensu Berkes 1994) by the local community varied considerably among the studies. For example, the case study presented by Polansky and Heermans (2004) and Robiglio and Mala (2005) were at level 3 (co-operation). Robiglio and Mala (2005) presented Cameroon Communal Forests. This kind of land tenure is quite new and the participation process is probably a work in progress. McGregor's (2002) study concluded that to reach stakeholders' satisfaction, a minimal co-management level requires a mutual exchange of information (level 4: communication), but only if the control of methods of knowledge sharing is relinquished to the indigenous community. Herrmann (2006) suggested also, in a second study, the implementation of a process that represents a minimum of level 4. This level seems to have been attained in another Cameroon Communal Forest (McCall and Minang 2005). The two projects in India reached level 5 (advisory committees), but they were possibly difficult to implement because of differences between the Forest Department and the villagers who depended on trees as a resource (fuelwood, fruits, fodder) (Sekhar 2004). Moreover, Martin and Lemon (2001) explained that these new institutional arrangements often reproduce the social relationships that marginalize certain groups of people, especially women. Processes developed by the different Canadian projects seem to be at level 6 (management board), following Berkes' (1994) scale (Karjala and Dewhurst 2003, Karjala et al. 2004, Natcher and Hickey 2002). Natcher and Hickey (2002) presented a management board composed by five community representatives out of 14 members, along with two others from economic development corporations, but final decisions with regards to forest management remained contingent upon the approval of the Minister of Environmental Protection (level 6). Karjala and Dewhurst (2003) and Karjala et al. (2004) presented the *Aboriginal Forest Planning Process*, which aims at integrating TEK and western science within a participatory process (using evaluation of management scenarios with different protection levels) in a co-managed research forest (local indigenous community-University of Northern British Columbia). Robinson and Ross (1997) described a parallel consultation process that complements the regular process, providing the community detailed information on annual operating plans, but with no direct participation in their elaboration. Natcher et al. (2005) also presented co-management committees including community representatives, which correspond to level 7 (partnership of equals) on Berkes' scale. Roth (2004) suggested that general information provided by foresters and detailed information of a specific area provided by local traditional knowledge could be merged in order to manage a territory (level 7: partnership of equals). Moreover, he noted the importance of having common ground and developing trust among stakeholders. After this, it is possible to find similarities in the different views to start a process of collaborative management, using negotiations. Ghazanfari et al. (2004) presented a work in progress, that they hope, will reach ultimately level 7. Finally, Klooster (2002) presented a case of strict community control (level 7), with report production and frequent control by community audits in Mexico.

A1.3.4. Economic involvement

Only one paper addressed issues of economical involvement of local communities in the forest sector (Parsons and Prest 2003). Parsons and Prest (2003) proposed possible economic partnerships between local communities and industries: opportunities for employment (in sawmills or in different logging contracts), training programs and joint ventures to build - or save - a mill or a value-added production. This represents a form of involvement but this is not participation *per se*, thus it could not be translated on a scale of participation like Berkes' (1994) one.

A1.4. Discussion

A1.4.1. Enhancing participation: How to facilitate TEK incorporation?

Robinson and Ross (1997) judged that the impact of indigenous knowledge on forest management was, with a few exceptions, negligible. This viewpoint was also expressed more recently by Stevenson and Webb (2003, 2004), indicating that the situation has not really changed over the past decade. According to these authors, the real contributions of indigenous knowledge to sustainable forest management (SFM) have yet to come. Effective policies, institutions and practices need to be developed in order to fully and equitably incorporate local communities into SFM. TEK integration still entails numerous challenges, given that: 1) TEK is disappearing at a fast pace, given that it is passed through oral tradition that may be lost over time - like language, TEK won't survive unless practiced - and the lack of human resources to document it before it is lost, 2) there are practical problems in trying to reconcile two very different worldviews and in translating ideas and concepts from one culture to another, 3) there is a problem of attitude that prevents both western scientists and traditional Peoples from respecting the value of each other's knowledge systems, and among western scientists, between natural and social scientists regarding research methods, and 4) there is a political problem that slows down the development of institutional arrangements which recognise the validity of TEK (Johnson 1992).

To gather TEK, there could be two different approaches: one could involve outsiders (familiarized to community worldview) who could collect traditional knowledge, or, the second could build on local people (familiarized to western science) who possess the knowledge, to transmit it to resource managers. In the first case, numerous methods exist to collect traditional knowledge or community perspectives and values, but not all have the same usefulness for incorporation in resource management. The more global is the understanding of local knowledge the more it could be well included in management plans (Ericksen and Woodley 2005). TEK gathering is often fragmentary, focusing on some specific aspects only. To bypass this problem, researchers usually use more than one approach. The review of methodologies is not exhaustive, for example rapid appraisal (RRA) and participatory appraisal (PRA) are also largely used. These methods, which allow to learn about local conditions in an iterative and expeditious manner, are used in the identification of community problems, and for monitoring and evaluation of ongoing activities, in different fields (particularly in social and medical sciences, Campbell 2001). In the second case, it is urgent to keep this knowledge alive within traditional communities. There is an unavoidable assimilation of these communities (dramatic cultural changes) and then a failure to transmit traditional knowledge to younger generations (trans-generation conflicts, state-defined education, Johnson 1992).

The lack of empowerment of local communities in using their knowledge in resource use decision-making is mainly due to the fact that this knowledge does not easily fit into the western scientific paradigm (Stevenson and Webb 2004). Because of its removal from context and subsequent distortion in translation, traditional knowledge loses part of its essence and becomes coarse information (sanitized knowledge). Moreover, when decisions using this information are taken, the absence of TEK holders and users contribute to increase this phenomenon (Stevenson and Webb 2004). Woodley (2005) developed a conceptual framework for representing traditional ecological knowledge which allows understanding how people relate to their environment, instead of documenting knowledge, that could assist in bridging differences in worldviews and epistemologies between researchers and traditional Peoples. In her framework, TEK emerge from context (history, demographic factors and biophysical features of the place where they live), practice (action, physical interaction and experiential learning) and belief (influence of spirituality and values on how they act within their environment). If context changes in time or space, a process of "reflexivity" allows more factual knowledge to be adapted in a new context. This process, which is considered part of the resilience and adaptive capacity of the community, may be a means to use TEK in a context of resource management (Woodley 2005).

Despite the fact that the *Millennium Ecosystem Assessment* initiative developed a general framework for collecting traditional knowledge and for incorporating it in current management purposes, each local committee had to modify and adapt this framework to fit the local context and then obtain the recognition of the community. However, some recommendations were proposed by Ericksen and Woodley (2005) to incorporate multiple knowledge systems. First, TEK gathering should be done by an interdisciplinary team composed by ecological scientists as well as social scientists (like anthropologists, philosophers of science or community-oriented researchers) to provide full understanding of local knowledge. All members of this team should endorse the interdisciplinary perspective and be ready to spend time working in close collaboration in a continuous way over years. In-depth understanding of local knowledge by outsiders requires skill, training, respectful behaviour, an open and non-judgmental attitude and experience of the place. Secondly, this team should be constantly assisted by a coordination office which acts as a bridge between scientists and users. On the other hand, all sources of information (scientific and traditional) should be critically assessed and validated through a proposed process including triangulation (cross-validation from different sources) and review by community as well as stakeholders (Ericksen and Woodley 2005). In order to give the same weight to each type of knowledge, Ericksen and Woodley (2005) proposed a cross-validation process, where local experts validate scientific knowledge and scientists validate local knowledge.

The lack of clarity in the definition of objectives that are pursued by government agencies and industries is undoubtedly at the source of some difficulties limiting the incorporation of TEK into forest management (Robinson and Ross 1997). Officially, governments and industries support the principle of integrating TEK into forest management plans and implement consultation processes with native communities affected by timber harvesting, but precise mechanisms are usually lacking. At best, some projects take inspiration from traditional practices or develop new approaches to reduce conflicts between stakeholders (Herrmann 2005). Even though some initiatives have appeared (Jacqmain et al. 2005), it is clear that the integration of these two knowledge systems (i.e., TEK and western science) at a large scale remains a distant goal, because the following conditions must be met: support for the comprehensive documentation of TEK, a recognition of alternative knowledge systems, support for cross-cultural education, and mostly, political recognition of local population claims to land and resources (Johnson 1992).

Between different approaches, the development of land use and occupation maps could also be a good first step to incorporate local population interests and practices, but Natcher (2001) discussed methodological limitations in land use mapping. Among these limitations are: financial constraints related to this kind of research, unequal financial support between communities, lack of technical training at the community level to implement locally controlled research, problems of representation of community land use reflecting only one perspective (under-represented factions of the community, like women), problems of "completeness" (a long process, only permitting a small part of the territory to be recorded), and problems of respecting intellectual property rights regarding the information, together with its decontextualisation by industry land managers. Moreover, mapping traditional land uses often has failed to recognise the cultural importance of landscapes for Indigenous People, restricting this knowledge to a simple spatial distribution of physical features on the landscape (Natcher 2001). It is the activity *per se* that is valued by local people more than the particular place where the activity occurs. On the other hand, this mapping is important for establishing legally recognized land tenure and boundaries, and also in providing a common ground to negotiate and develop an acceptable strategy for the different stakeholders (Roth 2004). According to Roth (2004), documenting the spatial organisation of environmental practices will contribute to the understanding of the challenges and possibilities to integrate TEK. Effective integration will pass through an analysis of compatibility and convergence between TEK and science inside each of the four spheres of Berkes' (1999) *knowledge-practice-belief complex*. Roth (2004) further suggested using spatial tools to facilitate the integration of TEK and science through their similarities. Land use maps also present some limits for both industry and government viewpoints: methodological and technical issues (accuracy of the data), accessibility of the data, unresolved issues regarding land claims, the need for government leadership (in developing appropriate policies), differing expectations and agendas, and the need for a global coordinated system for the conduct of land use studies (MacKinnon et al. 2001).

Some difficulties with incorporating TEK into forest management are easy to solve, while others are more complex. The first step is certainly the official recognition of the usefulness of TEK in forest management. Next, an in-depth understanding of local knowledge through their relationship with their environment could be collected by an interdisciplinary team and the establishment of clear procedures to incorporate this information efficiently could be developed, with respect to knowledge holders. Because TEK are place-based, processes to incorporate this knowledge also need to be locally developed with the community, even if a general framework could provide a benchmark. Land use maps (to have a spatial common ground of discussion), combined with value maps (to take in account more abstracted concepts and objectives), seem to be a positive step forward in the incorporation of TEK, as long as that they are really considered in forest management planning.

A1.4.2. How to enhance community involvement in forest planning and management?

Within the boreal forest of Canada, timber harvesting planning is contingent on conflicts between commercial forestry and indigenous ancestral rights to access wildlife resources. Several authors have used processes derived from Stevenson's models (Stevenson

2005, Stevenson and Webb 2003, 2004) to analyse the integration of TEK into forest planning. Stevenson and Webb (2003) first described the current status-quo (Fig. A1.2a), which revealed recurrent problems (as mentioned in the previous section), unacceptable for the majority. However, local communities are not really involved in the process; at best some of their knowledge is considered. First, only a small part of TEK is included in final forest management decision-making, because a large part is not accessible to managers, through either a lack of will or understanding (Stevenson 1996). Second, the data gathering process of TEK is problematic, because this knowledge is largely oral, taped, "filtered" by the translation process, transcribed and divorced from its original context, which causes important loss of information throughout the process of data acquisition (Stevenson and Webb 2003). Finally, collected TEK is often sanitized and modelled to match a western vision to make it more useable. Moreover, few considerations are given to knowledge holders (non-respect of intellectual property). When it is collected, knowledge becomes information, which is transcribed, and subsequently, this represents the authority reference, excluding knowledge holders. This process represents the actual scientific way of knowledge propagation. Results, when published, become universal knowledge and are no longer possessed by an individual or a group, contrary to registered patent.

Stevenson and Webb (2003) proposed a second model that more efficiently incorporates TEK with a real involvement of knowledge holders in decision-making (Fig. A1.2b). In this model, TEK is considered in its entirety and is equal to western science. The focus is not on how TEK could inform western science, but on the management approaches and ecological relationships that TEK and western science are intended to produce (Stevenson and Webb 2003). Western science emphasises ecological components, and then managers can provide information and knowledge regarding the assessment and management of valued ecosystem components. On the other hand, First Nations (and many traditional Peoples) valorize ecological relationships (network between the components), then local peoples can provide wisdom and knowledge relevant to manage valued ecosystem relationships, particularly between human-forest resource relationships. The two visions are complementary (Stevenson and Webb 2004). Currently, management considerations almost always focus on information about the resource and exclude knowledge of the relationships.

Stevenson and Webb (2003) also presented a process to establish an efficient participatory management in 4 steps. First, the communities need to document, assess and prioritise their needs, uses and values with respect to the forest and forest resources. Second, communities develop their land use and forestry objectives, plans, policies and regulations. Third, local communities negotiate with government agencies and industries to agree on effective policies, institutions and strategies that will meet their objectives, rights and interests, with respect to the other stakeholders. Finally, an adaptive management approach is required to modify existing policy and practice when required in the future. This means that traditional Peoples become stakeholders and have the same weight in all steps of forest management decision-making. During such a deliberative process, each stakeholder has to be open and critical about its role, in order to contribute to the establishment of a successful management plan.

The Waswanipi Cree Model Forest used this approach to build its own action plan. They proposed a negotiation mechanism of forest management plans with forest companies based on the principle of equality between community representatives and forest companies (Pelletier 2002, Waswanipi Cree Model Forest 2007). The first step documented traditional land use and occupation of the territory. These maps were combined with management objectives and values developed for each season (depending on traditional activities) to realize zoning maps with conservation values to prioritise (Waswanipi Cree Model Forest 2007). In parallel, a diagnosis of the state of the forest was conducted, not only from a forester perspective but also from the community perspective. For each zone, different management strategies were proposed in a guide to forest companies, who had to produce forest management plans. These plans had to be negotiated until consensus was reached (Pelletier 2002). Some issues were identified during the process, like unequal power between industry and community, communication problems, schedule delays, and management of special issues. The goal was not to move from forestry supremacy with no place for traditional land use to traditional land use supremacy with no place for forestry. This model did not aim at prioritising community land use while adjusting *ad hoc* forest harvesting, but intended to give equal weight to the stakeholders in a multiuse management of the forest.

A similar model was developed by Karjala and collaborators (*Aboriginal Forest Planning Process*) based on criteria and indicators that rank community priorities in the first place (Fig. A1.2c). First, they proposed to realize land use maps. Second, they compiled values, local needs and perspectives depending on people's age-groups. Third, they realized four thematic maps (fishing, hunting-trapping, plant gathering and cultural sites). Finally, they developed a zoning system and realized maps presenting three categories of criteria and indicators (spatial, quantitative and qualitative) (Karjala and Dewhurst 2003, Karjala et al. 2004). The authors have provided the list of problems they encountered throughout the project: lack of human, financial, technical and information resources; mistrust, misuse of information and misunderstandings; difficulty in collecting values and goals without legal and policy provisions; difficulty in identifying important and testable indicators; and lack of power of the community in decision-making.

All these case studies are in the same country and in the same context. Approaches involving local communities were also quite similar and seem to be appreciated by these communities. In all cases, the researchers acted as initiators or at least as mediators (in some countries, it could also be NGOs) among the community, government and industries. This mediation was not obligatory, however, but depended on the context and the capacity of each community. For example, in Thailand, a local community initiated a project of communal forestry (bottom-up project). Despite the fact that community members were highly motivated, legal support was absent, and governmental forest institutions could not transfer appropriate technology to the community. Hence, the scope for developing strategies combining TEK and science is limited, formal institutional arrangements are lacking, and community access to high-level technology is limited (Abdus Salam et al. 2006).

A1.4.3. Economic involvement

Hickey and Nelson (2005) defined four categories of economic partnership to help communities in choosing which is the best adapted to each situation, depending on pursued goals and objectives. These authors presented a Canada-wide survey of economic partnerships between First Nations and forest companies, including local opportunities for employment, training programs and joint ventures. Parsons and Prest (2003) noted an increase in aboriginal people involvement with professional and technical expertise in forestry. However, access to large amounts of capital necessary to build a mill, for example, are not easily available to small communities. Consequently, an association with an existing forest company is often necessary to initiate such projects (Hickey and Nelson 2005). In doing so, local communities can gain control over where and when logging is conducted, but never over how much area is to be harvested, because the annual allowable cut is determined by provincial governments throughout Canada. In some cases, this cause conflicts inside the community, because people feel betrayed even by indigenous forest companies, as well as by non-indigenous ones. Wyatt (2004) drew up the profile of an indigenous forest company in Quebec (Canada), and showed that indigenous workers are often restricted to road construction, planting and thinning jobs while non-indigenous people are mostly employed in harvesting operations. This lack of local professional skills is one of the most important issues for First Nations local communities as well as financial mismanagement. In another context, Mengue-Medou and Waaub (2005) evaluated socio-economic issues and positive/negative impacts of forest exploitation in Gabon, where public participation is quite low. The establishment of a forest company in the region under study created employment, but not as well as expected, again because of the lack of professional skills in the local population. Finally, there were mostly negative social effects for the local community: population increase with unemployment increase, conflict between villagers who need non-timber forest products and industries who want timber, loss of spiritual and cultural practices, loss of ancestral rights, loss of the social structure of power, increase in the prices for essential foods (Mengue-Medou and Waaub 2005).

There is often confusion between two types of involvement of a local community in forest management: economic involvement could in no case replace involvement in forest planning and management. Economic involvement is important in creating jobs and economic development, but community involvement during the planning process has a greater impact in terms of community empowerment, political power, ancestral territory rights recognition and traditional knowledge recognition. These two kinds of involvement are not exclusive, but there typically is a diversity of opinions within communities, i.e., the young could prefer jobs over recognition of territory rights while elders prefer the opposite (Natcher and Hickey 2002). Nevertheless, creating jobs (for example, trained managers and auditors) inside a management institution represents a form of involvement (Klooster 2002) but it is not participation.

A1.5. Conclusion

The few attempts for TEK incorporation into forest management could be classified into two categories of results: conflicts between different uses (for example, in a context of conflict between commercial forestry and indigenous rights for access to wildlife and non-timber forest products), and conflicts between conservation and traditional or commercial use (for example, in developing countries where deforestation is practised to facilitate agriculture). In all cases, conflicts resulted in the dispossession of ancestral lands by local populations. In the first category, Canadian initiatives seemed to be one of the most advanced in terms of community involvement, with several pilot projects already established (seven of the peer-reviewed papers are from Canada, four from Africa, seven from Asia and three from South America). Similar processes have been proposed, where the aim is a greater participation of local communities in forest planning and management. All reviewed processes need a long time to be implemented: to adapt a general framework to the local context, that becomes recognized by the community and to establish mutual confidence between stakeholders. Incorporating TEK in forest management plans could be done with land use maps where commercial forestry, traditional activities, and integral protection are zoned. With respect to traditional activity seasons or wildlife-associated “seasons” (e.g. reproduction, wintering), these zones need not be necessarily exclusive in time and in space. Concrete trials need to be tested; results, successes and failures have to be spread in order to improve processes and inform other researchers. Considerable efforts will be required before reaching a high level of participation of local communities and an efficient incorporation of TEK. This may only be accomplished through recognition of alternative knowledge systems, a greater open-mindedness, and support for inter-cultural education (in both directions). Important changes in mentality and firm political decisions (through law which imposes a participation process for example) are still required before more efficient partnerships between TEK and western science are reached in forest management planning. Concurrently, it will be important to train local people (community members) who will adopt the process and run it once it will be functional and well established.

As already noticed by Davis and Wagner (2003), who did a review on TEK gathering methodologies, too many researchers are not reporting critical details of their research designs and methodologies. Moreover, several interesting processes and initiatives were not published in accessible papers for others practitioners. We strongly encourage people (researchers, practitioners and communities) working in TEK-related topics and in traditional community involvement in natural resource management, more specifically forest management, to widely diffuse their research and results in peer-reviewed journals worldwide as the end result of any scientific process. A larger diffusion of studies would allow others working in the same domain to benefit from successes and errors made elsewhere. Secondly, this would contribute to increase recognition of TEK value for scientists and practitioners, and hence break down misunderstandings and ignorance related to TEK. As mentioned by Davis and Wagner (2003), it is time to move beyond current preoccupations with regards to theoretical issues and general endorsements of the value of traditional ecological knowledge, and begin the search of processes to document traditional ecological knowledge that efficiently involves local communities, and finally, to diffuse these processes in published and accessible papers.

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Table A1.1. Methodological approaches used to collect traditional ecological knowledge or community perceptions in reviewed studies related to forest management (N = 15). (Six studies were omitted because they presented no case study or no TEK gathering).

Articles	Open-ended interview	Group discussion	Questionnaire	Participant observation	Non-participant observation	Value and objective map	Land use and occupation map
Dei 1993	x			x	x		
Robinson and Ross 1997							x
Wickramasinghe 1997		x		x			
McGregor 2002	x					x	
Natcher and Hickey 2002	x	x		x			x
Karjala and Dewhurst 2003	x					x	
Ghazanfari et al. 2004				x			
Karjala et al. 2004	x	x				x	
Polansky and Heermans 2004							x
Roth 2004		x					
Sekhar 2004	x				x		
Herrmann 2005	x			x			
McCall and Minang 2005	x	x					x
Robiglio and Mala 2005		x					x
Silvano et al. 2005			x				x

Table A1.2. TEK gathering, incorporation, and community involvement in reviewed studies related to forest management (N = 21). Levels of Community involvement have been evaluated (when possible) using Berkes' (1994) levels of co-management : (1) informing, (2) consultation, (3) co-operation, (4) communication, (5) advisory committees, (6) management board and (7) partnership of equals/community control.

Articles	Country	Case study	TEK gathering	Gathering description	TEK incorporation	Community involvement	Involvement description	Target community
Dei 1993	Ghana	yes	yes	yes	no	no	-	Local population
Robinson and Ross 1997	Canada, AB	yes	yes	yes	yes	6-7	yes	Indigenous
Wickramasinghe 1997	Sri Lanka	yes	yes	yes	+/-	no	-	Local population
Thompson 1999	Indonesia	yes	no	-	no	yes*	yes	Local population
Martin and Lemon 2001	India	yes	no	-	no	5	yes	Local population
Gautam and Watanabe 2002	Nepal	no	-	-	-	no	-	Local population
Klooster 2002	Mexico	yes	no	-	no	7	yes	Local population, metis
McGregor 2002	Canada, ON	yes	yes	yes	yes	4	yes	Indigenous
Natcher and Hickey 2002	Canada, AB	yes	yes	yes	yes	6	yes	Indigenous
Karjala and Dewhurst 2003	Canada, BC	yes	yes	yes	yes	6	yes	Indigenous
Parsons and Prest 2003	Canada	no	-	-	-	economic	yes	Indigenous
Ghazanfari et al. 2004	Iran	yes	yes	yes	yes	ultimately 7	yes	Local population
Karjala et al. 2004	Canada, BC	yes	yes	yes	yes	6	yes	Indigenous
Polansky and Heermans 2004	Zambia	yes	yes	no	+/-	3	no	Local population
Roth 2004	Thailand	yes	yes	yes	yes	7	yes	Local population
Sekhar 2004	India	yes	yes	no	yes	5	yes	Local population
Herrmann 2005	Chile	yes	yes	yes	no	no	-	Indigenous
McCall and Minang 2005	Cameroon	yes	yes	yes	yes	4	yes	Local population
Natcher et al. 2005	Canada, YK	yes	no	-	no	7	yes	Indigenous
Robiglio and Mala 2005	Cameroon	yes	yes	yes	yes	3	yes	Local population
Silvano et al. 2005	Brazil	yes	yes	yes	no	no	-	Farmers

* This article presents a synthesis of different experiments of social forestry, which is impossible to rank using Berkes' (1994) scale.

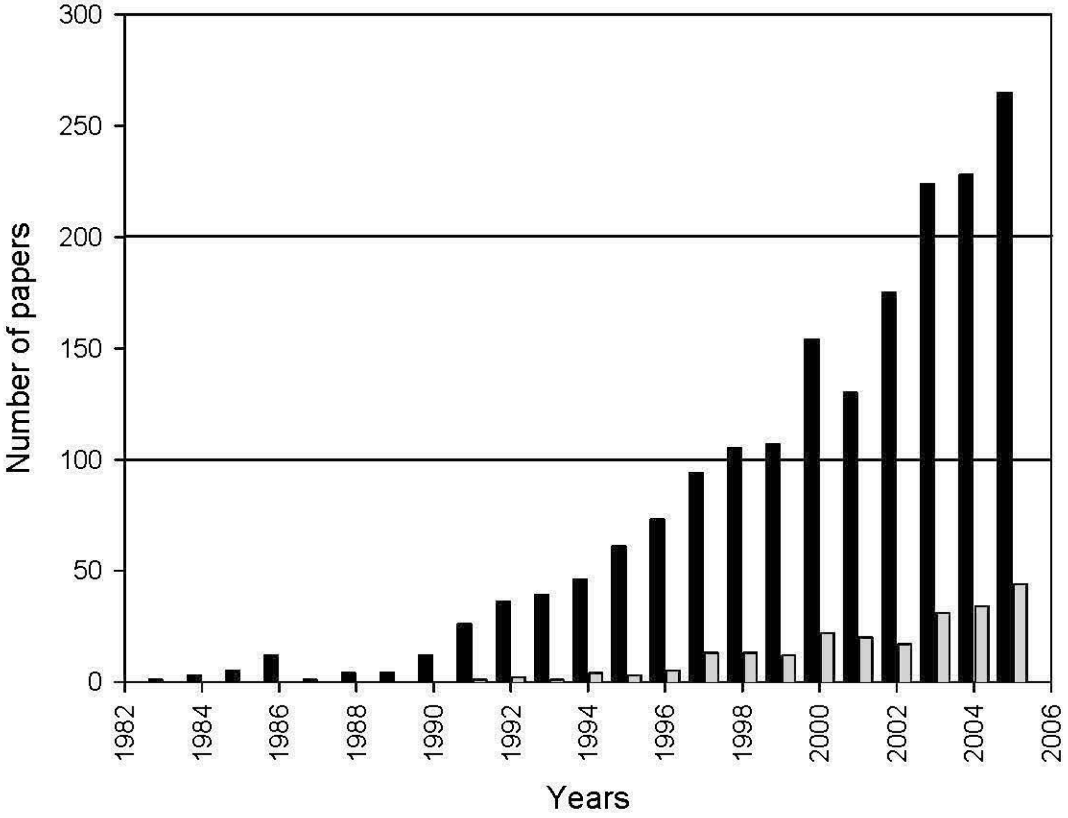

Figure A1.1. Progression of the number of published papers that address the issue of traditional ecological knowledge (TEK) in peer-reviewed journals (source: *Web of Science ISI* database). In dark, all papers on TEK; in grey, papers that refer to TEK in forest-related issues.

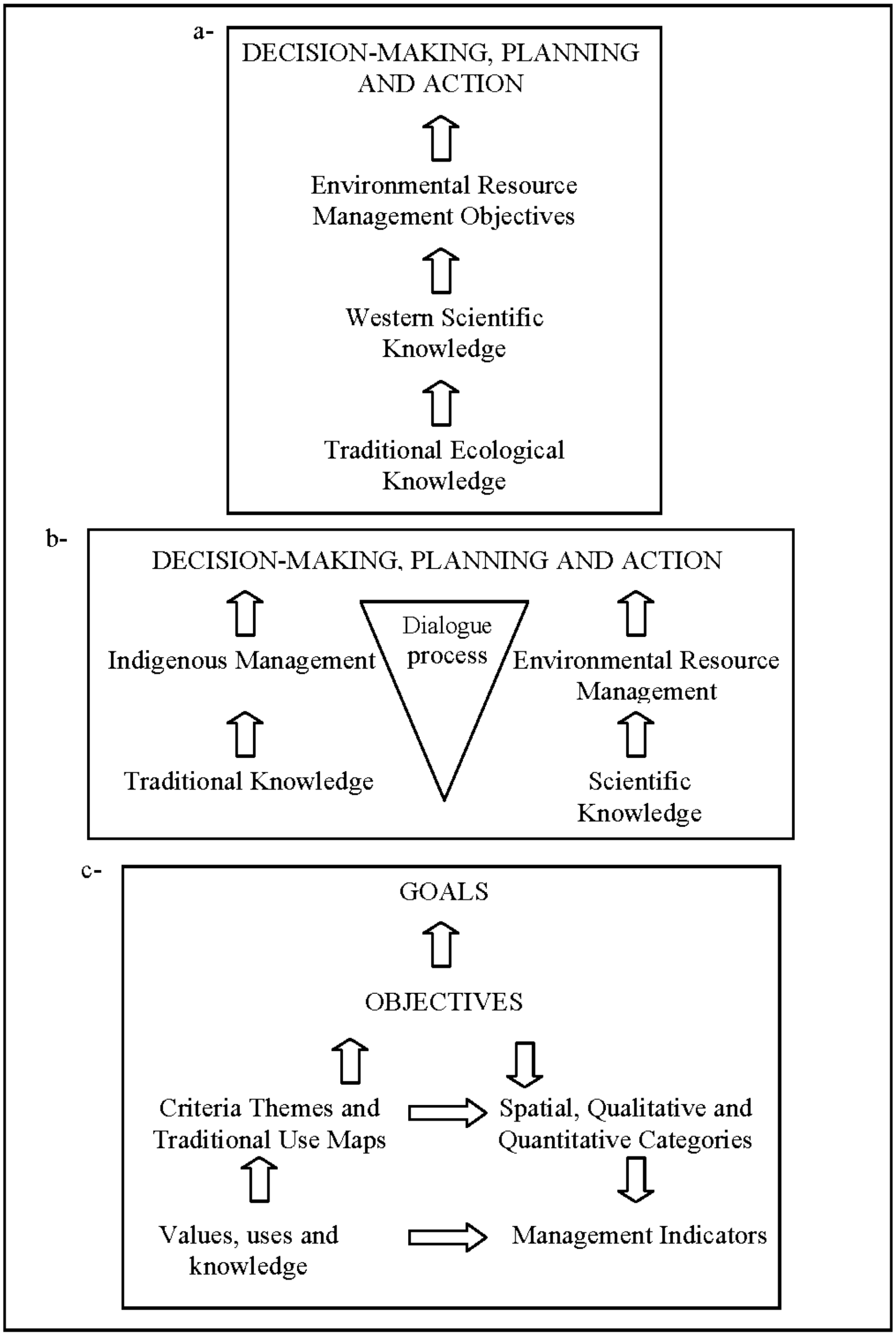

Figure A1.2. Models for incorporating TEK in resource management : a) status-quo (from Stevenson and Webb 2003), b) Stevenson's model (modified from Stevenson 2005) and c) Karjala's model (*Aboriginal Forest Planning Process* framework) (from Karjala et al. 2004).

ANNEXE II

FIELD USE OF ISOFLURANE AS AN INHALANT ANESTHETIC IN THE AMERICAN MARTEN (*MARTES AMERICANA*)

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Résumé

Nous avons évalué l'efficacité et l'aspect pratique de l'usage de l'isoflurane comme anesthésiant par inhalation couplé à l'oxygène chez la martre d'Amérique (*Martes americana*) sur le terrain. Soixante-huit martres ont été capturées dans la Forêt Modèle Crie de Waswanipi (Québec, Canada) entre octobre et novembre 2005 et anesthésiées avec de l'isoflurane dans 100% d'oxygène (1l/min) à l'aide d'un masque facial. L'induction se faisait à 3% d'isoflurane pour tous les animaux. La longueur moyenne (± écart-type) de l'induction était de 1,8 ± 1,2 min. Le maintien de l'anesthésie se faisait entre 1 à 4% d'isoflurane. Les manipulations duraient en moyenne 16,4 ± 7,1 min mais fluctuaient beaucoup. La longueur de la récupération, définie comme l'intervalle de temps entre la fin de la procédure et le réveil de l'animal, était courte (6,3 ± 2,8 min), et bien inférieure aux longueurs de récupération rapportées avec les anesthésiants injectables (≥ 70 min). En comparaison d'une administration « ouverte » d'isoflurane décrite dans des études précédentes, l'utilisation d'une machine à anesthésie limite le risque d'une potentielle surdose d'anesthésiant qui serait fatale. Nous concluons que parmi les techniques d'anesthésie actuellement disponibles, l'isoflurane couplé à l'oxygène est sécuritaire et utile pour l'anesthésie de martres sur le terrain, lorsque les contraintes liées au transport de l'équipement peuvent être surmontées.

Mots-clés

Martre d'Amérique, anesthésie, étude de terrain, inhalant, isoflurane, *Martes americana*, cône de contention

Abstract

We evaluated the effectiveness and practicality of using isoflurane as an inhalation anesthetic with oxygen as a gas carrier for American martens (*Martes americana*) in a field setting. Sixty-eight martens were trapped in the Waswanipi Cree Model Forest (Québec, Canada) from October to November 2005 and anesthetized with isoflurane in 100% oxygen (1l/min) using a face mask. Induction setting of isoflurane was 3% for all animals. Mean (± SD) length of induction was 1.8 ± 1.2 min. Maintenance isoflurane settings ranged from 1 to 4%. Procedures lasted an average of 16.4 ± 7.1 min and were uneventful. Length of recovery, defined as the interval between the end of the procedure and animal release, was short (6.3 ± 2.8 min), and well below reported lengths of recovery using injectable anesthetics ($\geq$ 70 min). As compared to open drop administration of isoflurane described in previous studies, the use of an anesthesia machine prevents the risk of potential fatal anesthetic overdose. We conclude that among anesthesia techniques currently available, isoflurane with oxygen as a gas carrier is a safe and useful field anesthetic in martens, when issues with equipment portability can be overcome.

Key words

American marten, anesthesia, field study, inhalant, isoflurane, *Martes americana*, restraint cone

A2.1. Introduction

The choice of an anesthetic agent in wild mammals depends on numerous factors, such as field logistic constraints, equipment cost and safety for both animals and staff. Chemical immobilization with injectable agents requires little equipment and is relatively simple to administer. In addition, newer drugs are associated with fewer adverse effects than older agents. As a result, injectable drugs are frequently used for the anesthesia of wild mammals in the field. However, recoveries may be prolonged, control of anesthesia levels is limited, and most injectable drugs are associated with cardiovascular and respiratory side effects. Inhalant anesthesia reduces the hazards of general anesthesia and ensures a rapid postoperative recovery of vital functions (Calvey and Williams 2001).

American marten (*Martes americana*) has been recognized throughout its range as being sensitive to anthropogenic forest disturbances such as commercial forestry (e. g. Potvin et al. 2000). In this context, marten population maintenance is often recommended as an indicator for forest habitat integrity (Thompson 1991). In addition to this, the marten is an important species for fur trapping by local and native peoples.

Different anesthetic protocols have been reported for the field anesthesia of this mustelid, either using injectable drugs (Belant 1992, Kreeger 1999, Belant 2005) or inhaled agents (Herman et al. 1982, Potvin et al. 2004). However, to the best of our knowledge, field use of isoflurane using a portable anesthesia machine with a precision vaporizer previously has not been described for martens. The objective of our study was to assess the efficiency, practicality, and usefulness of isoflurane inhaled anesthetic with oxygen as a gas carrier for American martens in a field setting and compare that protocol with other published methods for this species.

A2.2. Materials and methods

The study was conducted in October and November 2005 at the Waswanipi Cree Model Forest, municipality of James Bay, Québec, Canada (49°70’N, 76°50’W). This project was carried out according to animal utilization protocols approved by the animal care committees of the institutions involved in this project, both of which operate under the auspices of the Canadian Council on Animal Care.

Martens were captured in live traps (model 202, Tomahawk Live Trap Company, Tomahawk, Wisconsin, USA) covered with branches of spruce or fir, and baited mostly with beaver or hare flesh and trapping lures. Each trap was checked daily. Trapped martens were transferred into an open-ended restraining cone made of fabric with a 25-mm-diameter opening that allowed safe restraint of the animal with direct open access to its snout (Fig. A2.1). Anesthesia was induced with a facial mask (Guinea pig mask, 11/16" diameter, J.A. Webster Inc., Sterling, Massachusetts, USA) placed on the snout with isoflurane (AErrane®, Baxter, Toronto, Ontario, Canada) set at 3% and delivered in 1 l/min oxygen using a custom-made portable anesthesia machine. This machine was built using a semiopen, nonrebreathing Mapleson D modified Bain circuit connected to an isoflurane vaporizer (Tec-3 Model, Dispomed, Joliette, Québec, Canada) and an oxygen flow meter (Dispomed). The system was firmly fixed inside a solid plastic case (57 x 46 x 33 cm, Stormcase model IM2750, Hardigg, Saint-Jean-sur-le-Richelieu, Québec, Canada) to provide adequate resistance to rough conditions during transport. The total weight of this assembly was 10.2 kg. A portable type E medical oxygen cylinder (67 cm long x 10 cm in diameter and weighing 7.7 kg) was used as a source of oxygen.

The procedures were performed on the back seat of a crewcab at ambient temperatures varying from 1.5 to 25.8°C (10.4 ± 3.9°C, mean ± SD) in the truck. When sufficient levels of anesthesia were obtained, the isoflurane setting was decreased to an appropriate level, defined as maintaining inhibition of animal movement. Martens were weighed and morphometric measurements were made. Heart rate and hemoglobin oxygen saturation (SpO_2) were monitored with a pulse oximeter (OxiMax NPB-75® Handheld Capnograph/Pulse Oximeter, Nellcor Puritan Bennett, Plesanton, California, USA) with the probe placed on a forelimb digit. Respiratory rates were determined by counting complete thoracic cycles for 30 sec every 5 min. End-tidal carbon dioxide partial pressures ($ETCO_2$) were measured in some individuals with a microstream capnograph (OxiMax NPB-75®) via a pediatric nasal cannula (diameter 2.5 mm, Smart Capnoline® Oral/Nasal circuit, Nellcor Puritan Bennett, Plesanton, California, USA) which was inserted about 5–7 mm inside the nares. Rectal temperatures were monitored throughout the procedure and recorded once on average 12 min after the beginning of anesthesia. Ambient temperatures were measured at the beginning and the end of the procedure in the truck, near the vaporizer. When ambient temperatures were low, hypothermia was prevented by turning the engine on with the door closed but window partially opened.

After a complete physical examination, blood was sampled via the jugular vein, feces were collected from the rectum or in the trap, and a first premolar tooth was extracted using a small root elevator (Super Slim Feline Elevator, J.A. Webster Inc., Sterling, Massachusetts, USA) and extraction forceps. An ear swab was taken for parasitologic examination and an ear tag was installed. Radio transmitters were also attached in selected females (18 of 27 females) using a collar. Isoflurane administration was discontinued when the procedure was completed. Oxygen was delivered via face mask until the marten started to move.

The animals were left in the trap to recover and released as soon as the level of wakefulness was believed to be sufficient. Length of induction was defined as the interval between the beginning of mask induction and a lack of response to tactile stimuli. Length of recovery was defined as the interval between the end of anesthesia (isoflurane = 0%) and the moment when the marten was ready for release (normal gait and normal behavior such as vocalization and signs of aggression).

Statistical analyses were performed using SAS 9.1 (Cary, North Carolina, USA). T-tests for unequal variances were used to examine the effects of period of the day (AM versus PM, which could represent the time spent by the marten inside the trap) on the induction, recovery, and procedure times. Linear regression models were used to test the relationship between weight, body index (weight divided by the total length), ambient temperature, minimum and maximum settings of isoflurane, and on the lengths of induction, recovery and procedure. Alpha was set at 0.05.

A2.3. Results

Forty-one male (weighing 953 ± 131 g) and 27 female (weighing 680 ± 107 g) martens were anesthetized during the project. Each captured marten was anesthetized once. Although some animals appeared subjectively thin, no significant anomalies were observed during the physical examination. All procedures were uneventful and no anesthesia-related mortality occurred. Minimum and maximum maintenance isoflurane settings ranged from 1.0% to 2.5% (1.4 ± 0.3%) and from 2.0 to 4.0% (3.0 ± 0.2%), respectively. Inductions and recoveries were usually rapid and smooth, with animals exhibiting minimal agitation. Mean length of induction was 1.85 ± 1.20 min (range: 0.5 – 6 min) and the whole procedure lasted 16.4 ± 7.1 min (range: 9 – 35 min). Recovery took place in 6.3 ± 2.8 min (range: 2 – 17 min). Respiratory parameters and heart rates were recorded in 18 and 8 martens respectively, on average 10 min after the beginning of anesthesia. Mean respiratory rate was 31 ± 11.5 movements per minute (mpm) (range: 12 – 64). Mean heart rate was 216 ± 17 beats per minute (bpm) (range: 200 – 240). Mean $ETCO_2$ was 40.4 ± 5.8 mm Hg (range: 32 – 53 mm Hg) and all values of SpO_2 remained above 95 % during the procedure. Mean rectal temperature, recorded on average 12 minutes after induction, was 37.4 ± 1.5°C (range: 34.0 – 40.1°C). No relationship was observed between the other variables tested and the lengths of induction, recovery, and procedure.

We compared the lengths of induction and lengths of recovery obtained in our study with results from previously published anesthetic protocols in martens (Table A2.1). Twenty-seven of the 68 martens (40%) were recaptured at least once between one and 33 days after anesthesia. All recaptured animals were very active and appeared to behave normally. The cost of isoflurane and oxygen was estimated to be in average 0.66 USD and 0.55 USD per procedure respectively. Less than 75 ml of isoflurane and three type E medical oxygen cylinders were necessary to anesthetize the 68 martens used in this study.

A2.4. Discussion

Anesthesia of free-ranging animals in a field setting is often challenging. Age and health status are often uncertain, and the body condition of wild animals is regularly suboptimal. In addition, capture-related stress may lead to increased plasma levels of glucocorticoids and catecholamines (Spraker 1982). As a consequence, any adverse effects of anesthetic agents are likely to be greater than those observed in domestic animals. Capture and prolonged containment in live traps, followed by handling and anesthesia, certainly carries some physiological implications, as well as negative effects on the animal's well-being (Mathews et al. 2002). As animal users, scientists are ethically responsible for refining handling techniques used in order to minimize any negative effects of their interventions on individuals and the population as a whole. Field biologists benefit considerably from refining interventions to reduce sampling variances.

The anesthesia protocol used in our study enabled us to perform all procedures rapidly and safely for both staff and animals; no mortality or anesthesia problems occurred in the 68 martens. Induction and recovery were rapid and smooth, and cardiorespiratory parameters remained within the acceptable range for anesthetized animals at all times.

Kreeger et al. (1998) noted that a great deal of maintenance for the vaporizer was required when using isoflurane with an anesthesia machine. The vaporizer we used was calibrated only once, at the beginning of the project. The possibility of inaccuracy due to decalibration or variation in ambient temperature was overcome by adjusting the vaporizer setting according to animal response. Nevertheless, because we used this vaporizer outside its recommended range of temperatures (15 to 35 °C), the reported settings of isoflurane are not necessary equal to the actual concentration delivered. Even if isoflurane has a slightly pungent and ethereal odor and can irritate the upper airways (Calvey and Williams 2001), martens tolerated induction via face mask well; this is consistent with observations of Siberian polecats (Gaynor et al. 1997), sea lion pups (Heath et al. 1997), beavers (Breck and Gaynor 2003), and ferrets (Lawson et al. 2006). Isoflurane is a volatile agent with a low blood:gas solubility coefficient (1.4, versus 2.4 for halothane) and low tissue solubility, resulting in extremely rapid induction and recovery (Calvey and Williams 2001). The two reported protocols using injectable agents in martens were associated with prolonged and variable length of recovery when compared to inhaled agents (Table A2.1). These lengthy recoveries could be detrimental for the animals by increasing the level of stress and the potential for detrimental hypothermia, especially at relatively cold ambient temperatures.

Enduring residual anesthetic effects could also prevent the animal from resuming normal activities such as hunting and caring for young. In addition, long recoveries increase post-procedural monitoring time by handlers, which diminishes the productivity of the field team and therefore adds to the cost of the project. Because dosages of injectable agents are approximate, premature recoveries may occur, potentially exposing the handler to bites and zoonotic disease transfer. Inhalant anesthesia allows for rapid modifications of anesthesia depth and therefore prevents such undesirable outcomes. The definition of the length of recovery varies between studies limiting useful comparisons of the different protocols. Belant (1992, 2005) defines recovery length as the time between immobilization and the animal's ability to maintain an upright posture and respond to external stimuli. Potvin et al. (2004) described recovery time as the time between recumbency and the moment when the animal stood up once released. In the Herman et al. (1982) study, the length of recovery was not clearly defined.

Among the volatile anesthetics commonly used in veterinary anesthesia, isoflurane is considered to have the fewest effects on the heart but its effects on respiration are more pronounced than with halothane (McKelvey and Hollingshead 2003). Only 0.2% of the isoflurane dose is metabolized (Calvey and Williams 2001), which allows for its utilization in neonatal and geriatric animals, as well as in animals with hepatic or renal damage (McKelvey and Hollingshead 2003). Cardiovascular depression is a common side effect of most injectable agents. Mean heart rates measured during tiletamine-zolazepam-xylazine and ketamine-xylazine anesthesia were, respectively, 163 ± 34 bpm (Belant 2005) and 125 ± 7.5 (SE) bpm (Belant 1992), compared to 216 ± 17.2 bpm in the present study. Mean respiratory rates were superior with injectable agents (67 ± 30 mpm and 65 ± 15.1 (SE) mpm respectively) compared to inhalation agents (31 ± 11.5 mpm, similar to normal respiratory rates of a similarly-sized mustelid, the domestic ferret (33 – 36 mpm, Fox 1998)). Moreover, isoflurane is delivered in 100% oxygen, and oxygen arterial partial pressure is likely to be greater using this technique. In our study, ventilation was assessed in 18 martens by measuring $ETCO_2$. All values obtained were in the expected range for a small mammal and indicated that ventilation and pulmonary gas exchanges were adequate throughout the procedure. An added safety feature of precision vaporizer delivered inhalant anesthesia is that endotracheal intubation and assisted manual ventilation are possible if apnea occurs. General anesthesia can induce hypothermia, because muscular activity and metabolism are depressed. In the present study, mean rectal temperature 12 minutes after induction with isoflurane (37.4 ± 1.5°C) were similar to those reported with the combination of tiletamine-zolazepam-xylazine (37.0 ± 2.2°C) (Belant 2005).

Inhaled agents have been used in free-ranging martens with open drop exposure by injecting the liquid agent into a closed chamber (Herman et al. 1982, Potvin et al. 2004). Halothane and isoflurane are compounds with a high vapor pressure and evaporate so easily that they can reach a concentration of more than 30% at a barometric pressure of 760 mm Hg, depending on ambient temperatures (Mathew et al. 2002). That level could cause a fatal anesthetic overdose (McKelvey and Holligshead 2003). A precision vaporizer limits the evaporation of these agents and allows for their safe use in anesthesia. Use of an anesthesia machine also permits maintenance of the animal under an appropriate level of anesthesia for as long as needed.

Even if toxicity to isoflurane has not been clearly demonstrated, exposure of staff to the anesthetic gas is always an issue. Induction via a face mask reduces this exposure when compared to induction in a closed chamber.

The main disadvantages of the use of isoflurane anesthesia in the field are the weight and bulk of the equipment for remote use, and that mask induction without injectables requires adequate control and restraint of the animal. Our method for induction worked very well for this purpose. Oxygen cylinders can be safety issue for air transport. Kreeger (1999) reported that a great deal of training was necessary to work with an anesthesia machine. For our study, wildlife biologists and technicians were trained by a veterinarian at the beginning of the study until an adequate number of animals were handled (n = 10-15) and they did not encounter any difficulties using the machine on their own. Because no mortalities occurred, training appeared sufficient. Anesthesia is never risk-free and whatever the technique used, adequate training of the staff is recommended to improve anesthesia safety as much as possible.

In our study, isoflurane anesthesia proved safe, easy to use, and provided smooth and rapid inductions and recoveries. Additional costs of this technique were considered to be balanced by the improvement of animal welfare and overall anesthesia security, and by the gain in productivity of the field team. When equipment transport is possible, isoflurane inhalant anesthesia with a precision vaporizer provided excellent results in the American marten under field conditions.

A2.5. Acknowledgments

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Table A2.1. Comparison of induction and lengths of recovery with different anesthesia techniques described in American martens.

Anesthetic agent[a]	Route of administration	*n*	Induction (min)	Recovery (min)	
			Mean ± SD (Range)	Mean ± SD (Range)	Reference
Isoflurane	Semi-open, non-rebreathing	68	1.8 ± 1.2 (0.5 - 6.0)	6.3 ± 2.8 (2.0 - 17.0)	Current report
Isoflurane	Open drop	62	1.6 ± 0.8 (0.5 - 4.8)	3.6 ± 2.4 (0.5 - 9.7)	Potvin et al. 2004
Halothane	Open drop	264	2.6 ± 0.7 (1.5 - 8.3)	3.1 ± 1.5 (1.0 - 10.0)	Herman et al. 1982
TZX	Injectable	19	2.5 ± 1.8 (0.8 - 6.1)	70.8 ± 31.9 (30 – 122)	Belant, 2005
KX	Injectable	5	1.8 ± 0.5 (1.2 - 2.5)	100.4 ± 43.2 (62 – 175)	Belant, 1992

[a] TZX: Tiletamine-Zolazepam + Xylazine; KX: Ketamine + Xylazine.

Figure A2.1. Restraint cone used to immobilize martens for mask induction with isoflurane. A. The fabric cone is connected to the trap to allow transfer of the animal. B. Mask induction of the animal in the restraint cone is greatly facilitated by the safe access to its snout via the opening in the extremity of the cone.