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RÉSUMÉ

Le taux actuel d'extinction des espèces alarme de nombreux chercheurs à travers le monde. La fragmentation et la détérioration des habitats constituent la menace la plus importante à la biodiversité. Par ailleurs, le groupe des amphibiens subit, depuis quelques décennies, un impressionnant déclin : une espèce sur trois serait menacée dans le monde ou en voie de le devenir. L'échange d'individus entre les populations locales est particulièrement important pour la viabilité de ce groupe d'animaux. La fragmentation et la détérioration des territoires, suite à l'implantation de routes, semblent constituer un problème sérieux. Toutefois, le manque d'informations quantitatives quant à l'impact réel des routes sur la dynamique des populations de ces espèces est criant. Nous avons étudié l'impact des routes sur la dynamique de colonisation et d'extinction d'amphibiens en milieux humides dans deux régions du Québec. Pour chaque aire d'étude, nous avons établi 54 sites dans un rayon de 50 km autour du Parc national de Plaisance, en Outaouais, ainsi que du Parc national d'Aiguebelle en Abitibi-Témiscamingue. Ces sites ont été sélectionnés à partir d'un gradient de distances à la route (< 50 m, 50-100 m, > 100 m) ainsi que du type de route (non asphaltée, secondaire asphaltée, principale asphaltée), afin de remplir équitablement les 9 combinaisons possibles. La détection de trois espèces d'anoures (Grenouille verte (*Lithobates clamitans*), Grenouille du Nord (*Lithobates septentrionalis*), et Rainette crucifère (*Pseudacris crucifer*)) a été effectuée lors de stations d'écoute, à raison de 3 ou 4 visites par saisons d'échantillonnage, et ce, de 2012 à 2014. Nous avons émis l'hypothèse que la quantité de couvert forestier dans un rayon de 500 m permet de mitiger les effets négatifs des routes. Pour ce faire, nous avons considéré des interactions entre le couvert forestier et la distance du milieu humide à la route (couvert x distance) ou le type de routes (couvert x type de route). Des facteurs pouvant influencer l'occurrence initiale des espèces tels que le pH, la conductivité et la taille du site ont également été pris en compte. De plus, certains facteurs liés à la probabilité de détection imparfaite tels que les facteurs météorologiques, le passage d'automobiles et le nombre de jours écoulés depuis la fonte des neiges ont été considérés. Des variables concernant le paysage – dans un rayon de 500m autour des sites – telles que la quantité de routes et la quantité de milieux humides furent ajoutés aux modèles. Nous avons analysé les données à l'aide de modèles d'occupation dynamique. Les résultats indiquent que, pour la Grenouille verte, la probabilité d'extinction diminue avec le pH du milieu humide et augmente avec la proportion de résineux dans le couvert forestier. La probabilité de détection augmente avec le nombre de jours écoulés depuis la fonte des neiges. Pour la Grenouille du Nord, la probabilité de colonisation augmente avec la taille du milieu humide. Pour la Rainette crucifère, l'effet de la distance à la route sur la probabilité d'occurrence dépend du couvert forestier (interaction couvert x distance), mais diffère selon le type d'essence. Pour la distance de 50 m à une route, la probabilité d'occurrence diminue avec le couvert forestier de type mixte alors que pour des distances de 100 et 150 m à une route, la probabilité d'occurrence augmente avec le couvert forestier de type mixte. Pour la distance de 50 m à une route, la probabilité d'occurrence augmente avec le couvert forestier de type feuillu alors que pour des distances de 100 et 150 m à une route, la probabilité d'occurrence diminue avec le couvert forestier de type feuillu. De plus, la probabilité de détection de la Rainette crucifère diminue avec le nombre de jours écoulés depuis la fonte des neiges. Les différences de résultats observées entres les espèces à l'étude peuvent s'expliquer par la variabilité intrinsèque, notamment en ce concerne la préférence d'habitat et le moment de reproduction. Les résultats obtenus n'ont pas révélés d'effet des routes sur la dynamique de colonisation et d'extinction des amphibiens en milieux humides. Néanmoins, nous croyons que la poursuite de ce projet à long terme permettrait d'affiner les résultats obtenus, considérant par ailleurs, l'expansion toujours grandissante des réseaux routiers dans les aires à l'étude.

Mots clés: amphibiens; milieux humides; routes; extinction; colonisation; couvert forestier; occupation.

CHAPITRE I

INTRODUCTION GÉNÉRALE

1.1 Mise en contexte

En 2015, la planète Terre comptait déjà plus de 7,2 milliards d’êtres humains, alors qu’au début du siècle dernier, la population mondiale se chiffrait aux environs de 1,6 milliards (United Nations 2015; FAO 2015). Cette explosion démographique a fait accroître l’utilisation de ressources afin de combler les besoins de cette nouvelle population (United Nations 2015). De plus, l’évolution technologique a eu pour effet d’augmenter la consommation en biens divers engendrant une demande en matières premières et en énergies fossiles toujours plus grande (FAO 2015). En effet, une pression grandissante s’est exercée dans les domaines de l’agriculture afin de combler les besoins alimentaires, et dans l’industrie des forêts afin de fournir bois d’œuvre, pâtes et papiers et autres produits dérivés (FAO 2015). Ainsi, le taux de déforestation est à la hausse au fil des décennies (FAO 2015). Depuis l’année 2008, on compte désormais plus de personnes vivant dans les villes que dans les campagnes à travers le monde (United Nations 2015; FAO 2015). Suite à l’important développement urbain, les réseaux routiers ont également connu une recrudescence impressionnante afin de permettre la distribution de ces ressources (Central Intelligence Agency 2015; FAO 2015; United Nations 2015). En somme, les écosystèmes furent modifiés, et ce, à une vitesse qui ne correspond pas toujours à la capacité d’adaptation du milieu et des espèces (Shaffer 1981).

Plusieurs études à travers le monde s’accordent pour dire que le taux actuel de disparition des espèces serait hautement supérieur au taux normal estimé à partir d’archives fossiles, et qu’il serait même porté à augmenter au cours du prochain siècle (Sala *et al*. 2000; Ricciardi et Rasmussen 2001; Green 2003; Baillie 2004).

Selon la liste rouge des espèces menacées de l'Union Internationale pour la Conservation de la Nature (UICN), les modifications des écosystèmes – imputables à la croissance et au développement de la population humaine – expliqueraient le déclin de plus de 66% des espèces animales menacées à travers le monde (IUCN 2015 [En ligne]). Parmi ces modifications, l'UICN cite : la perte d'habitat résultant du développement commercial et résidentiel, l'amplification des réseaux de transports et les activités de l'industrie forestière (IUCN 2015 [En ligne]). Les amphibiens constituent le groupe le plus touché par ces changements en représentant 36.5% de ces espèces (soit, 4243/11 627 espèces menacées), comparativement aux mammifères (22,3% d'espèces menacées), aux oiseaux (21,7% d'espèces menacées) et aux reptiles (19,4% d'espèces menacées) (IUCN 2015 [En ligne]).

C'est au cours des années 1980, que pour la première fois, le déclin des amphibiens vint sonner l'alarme auprès des scientifiques de par le monde, et particulièrement lors du premier Congrès Mondial d'Herpétologie, en 1989 (Barinaga 1990; Bradford 1991; Wake 1991; Crump *et al.* 1992; Blaustein *et al.* 1994; Gardner 2001; Green 2003; Stuart *et al.* 2004). Depuis, plusieurs scientifiques ont tenté d'expliquer les causes potentielles de ce déclin. Par exemple, l'augmentation du rayonnement ultraviolet (UV), les pluies acides, l'intensification de l'agriculture, les changements climatiques, l'augmentation du taux de pollution, la contamination chimique de l'environnement, le prélèvement d'individus dans la nature, sont autant de causes expliquant l'importante diminution des amphibiens dans le monde (Blaustein *et al.* 1994; Daszak *et al.* 1999; Bridges et Semlitsch 2001; Blaustein *et al.* 2003; Sparling *et al.* 2009). D'ailleurs, une maladie infectieuse, la chytridiomycose, fut associée à une part significative du déclin des amphibiens en engendrant des mortalités de masse dans plusieurs régions isolées, depuis la fin des années 1990 (Berger *et al.* 1998; Longcore *et al.* 1999). Un champignon du groupe des chytridiomycètes, le *Batrachochytrium dendrobatidis*, transmis dans l'eau ou par contact entre amphibiens, principalement dans les régions montagneuses où la température est relativement fraîche, serait à l'origine de cette pandémie qui aurait déjà affecté plus de 17% des espèces (Berger *et al.* 1998; Longcore *et al.* 1999; Baillie 2004). Pour toutes ces raisons, le déclin des amphibiens est actuellement si important, qu'une espèce sur trois serait menacée dans le monde ou en voie de le devenir (Wake 1991; Baillie 2004). Néanmoins, bien que certaines menaces frappent de manière localisée, la fragmentation et la destruction des habitats apparaissent comme étant la menace la plus pesante et son importance est exacerbée par son étendue (Knutson *et al.* 2001; Baillie 2004).

Dans cet ordre d'idées, en 2015, le réseau mondial comptabilisait plus de 64 285 009 km de route (Central Intelligence Agency 2015). Avec 1 042 300 km de routes, le Canada se trouve à la 7^{e} position en termes d'importance de son réseau routier (Central Intelligence Agency 2015). À l'image de l'accroissement routier à l'échelle planétaire, l'expansion du réseau routier au Québec a été très rapide, malgré la nature relativement jeune de l'histoire de cette région du globe. En effet, il n'y avait que 267 km de routes au Québec en 1734, lorsque les villes de Québec et Montréal furent reliées pour la première fois (Gilchrist 2012 [En ligne]). Moins de 300 ans plus tard, la province du Québec compte environ 185 000 km de routes (Transports Québec [En ligne]). Par ailleurs, au cours de la période allant de 1985 à 2000, au Québec, le nombre de véhicules de promenade a augmenté de 68% et le nombre de camions lourds a augmenté de 34.5% (Transports Québec [En ligne]). En plus de détruire et modifier l'environnement, puis de fragmenter les territoires de manière irréversible (Sanzo et Hecnar 2006; Fahrig et Rytwinski 2009), les routes génèrent une pollution considérable et surpassent les apports naturels en sels (principalement le NaCl, utilisé comme déglaçant hivernal sur les routes) (Karraker *et al.* 2008). En effet, certains auteurs ont démontré que l'augmentation des sels aux abords des routes pouvait augmenter les risques de malformations, engendrer un déséquilibre dans l'osmorégularité et diminuer de façon générale la survie des individus (Karraker *et al.* 2008; Reeves *et al.* 2008; Karraker et Gibbs 2011).

Plusieurs chercheurs se sont penchés sur l'impact des routes sur les populations animales au cours des dernières décennies (van Gelder 1973; Fahrig *et al.* 1995; Forman et Alexander 1998; Hels et Buchwald 2001; Spellerberg 2002; Fahrig et Rytwinski 2009; Benítez-López *et al.* 2010; van der Ree *et al.* 2011). Dans certains cas, des modifications au niveau morphologique et comportemental chez les amphibiens vivant près des routes, furent rapportés (Spellerberg 2002; Mazerolle *et al.* 2005; Sanzo et Hecnar 2006; Bouchard *et al.* 2009; Parris *et al.* 2009; Hoskin et Goosem 2010). En effet, étant donné leur plus faible capacité de dispersion et certains caractères spécifiques à leur cycle vital, les amphibiens semblent particulièrement affectés par la création de routes à proximité de leur domaine vital, comparativement aux mammifères et aux oiseaux (Wilbur 1980; Cushman 2006; Becker *et al.* 2009). Toutefois, malgré l'apparente hausse du taux de mortalité des populations d'amphibiens aux abords des routes (Fahrig *et al.* 1995; Forman et Alexander 1998; Spellerberg 2002; Mazerolle 2004; Fahrig et Rytwinski 2009), peu de chercheurs se sont réellement attardés à étudier l'impact de celles-ci sur la dynamique des populations (Lesbarrères et Fahrig 2012).

Une méthode fréquemment utilisée pour déterminer l'impact de la circulation sur les populations consiste à compter le nombre de « carcasses » sur les bordures de routes, mais comme le mentionne Slater (2002), il s'agit d'une méthode qui sous-estime la perte réelle d'individus. Comme alternative, il est possible de déterminer l'occurrence des espèces. Néanmoins, les données de type « présence-absence » soulèvent un souci important en écologie animale (MacKenzie *et al.* 2002; MacKenzie *et al.* 2003; Gu et Swihart 2004), et d'autant plus en herpétologie, étant donné la faible probabilité de détection de ces organismes sur le terrain (Mazerolle *et al.* 2007). À ce titre, certains auteurs suggèrent de prendre en compte la probabilité de détection lors de travaux avec l'herpétofaune, afin d'accroître la qualité des résultats (Schmidt 2003; Mazerolle *et al.* 2005).

1.2 Généralités

1.2.1 Situation des amphibiens au Québec et au Canada

Au Canada, en 2010, sur les 49 espèces d'amphibiens recensées par le Réseau Canadien de Conservation des Amphibiens et des Reptiles (RÉCCAR), 19 figuraient sur la liste des espèces en péril du Canada (COSEPAC [En ligne]). En 2015, le COSEPAC ([En ligne]) recensait, pour sa part, 41 espèces d'amphibiens sur le territoire dont 25 avaient comme statut « En voie de disparition », « Menacée » ou « Préoccupante ». Deux espèces de salamandres, soit la population carolinienne de la salamandre pourpre (*Gyrinophilus porphyriticus*) ainsi que la population carolinienne de la salamandre tigrée de l'Est (*Ambystoma tigrinum),* avaient obtenu le statut de « Disparue du pays », respectivement en mai 2011 et en novembre 2012.

Aussi, au Québec, sur les 21 espèces recensées par l'Atlas des Amphibiens et des Reptiles du Québec (AARQ [En ligne]), trois figurent sur la liste des espèces désignées menacées ou vulnérables (LEMV) du Québec. Il s'agit de la Rainette faux-grillon de l'Ouest (*Pseudacris triseriata*), de la population carolinienne de la Salamandre pourpre (*Gyrinophilus porphyriticus*) – tous deux de statut «Vulnérable », respectivement attribués en septembre 2001 et en octobre 2009 – et la Salamandre sombre des montagnes (*Desmognathus ochrophaeus*) – au statut «Menacé » attribué en octobre 2009 – (Ressources naturelles du Québec [En ligne]). La perte d'habitat, et particulièrement la perte ou la détérioration des milieux humides, est la menace la plus importante à laquelle ces espèces doivent faire face (Lesbarrères *et al.* 2014).

Toutefois, comme en témoigne la fréquence de mise à jour des données, il est difficile de suivre la réelle progression des populations d'amphibiens à travers le Canada. Ceci peut être principalement expliqué par l'effort de suivi moins important pour ce groupe peu charismatique pour le grand public.

1.2.2 Importance des amphibiens dans les écosystèmes

Les amphibiens servent parfois d'indicateur biologique, compte tenu des particularités de leur biologie et de leur cycle vital. Il importe donc de bien comprendre les mécanismes impliqués dans leur déclin étant donné l'entrelacement entre la santé des amphibiens et la « santé » de l'environnement (Wake 1991). Tout comme les reptiles et les poissons, les amphibiens sont ectothermes, c'est-à-dire que leur température corporelle dépend de la température ambiante du milieu et que leur métabolisme dépend du climat (Vitt et Caldwell 2009). Aussi, leur peau est humide et dépourvue de protection, ils respirent par celle-ci et cette perméabilité, jointe à leur caractère d'ectothermie, leur confère une plus grande sensibilité aux changements des conditions de l'environnement (Alford et Richards 1999; Vitt et Caldwell 2009).

Le rôle des amphibiens est déterminant dans de nombreuses chaînes trophiques, principalement celles des milieux humides (Lehtinen *et al.* 1999). Puisqu'au cours de leur cycle vital ils occupent différentes niches écologiques, ils peuvent « jouer » les rôles successifs de sources de nourriture et de prédateurs (Burton et Likens 1975). Par exemple, Burton et Likens (1975) ont étudié les salamandres en milieu forestier afin de mieux comprendre leur rôle dans l'interaction et l'échange d'énergie dans le cycle des nutriments d'un écosystème du New Hampshire. Ces salamandres se sont révélées être une partie importante de la biomasse des vertébrés, représentant environ le double de celle des oiseaux, et équivalant environ à la même que celle des petits mammifères en saison de reproduction. De plus, les anoures adultes ont un rôle de régulation important au niveau des insectes puisqu'ils s'en nourrissent en grande quantité, mais ils constituent également une source alimentaire importante pour de nombreux oiseaux et mammifères. Un impressionnant effet « top-down » a également été démontré par quelques auteurs (Woolbright 1991; Beard *et al.* 2002) en lien avec une espèce d'anoure terrestre de la forêt humide subtropicale, *Eleutherodactylus coqui*. Cette espèce aurait un rôle déterminant dans la dynamique des nutriments de leur écosystème en transformant les insectes dont ils se nourrissent en nutriments disponibles pour les plantes et les microbes. La biomasse des amphibiens est donc un élément clé dans le maintien des milieux qu'ils colonisent (Gibbons *et al.* 2006).

1.3 Biologie des anoures

1.3.1 Qu'est-ce qu'un amphibien ?

Le terme amphibien vient du grec *amphibios* qui signifie « deux vies » et il caractérise généralement des tétrapodes qui partagent leur cycle vital entre deux milieux : l'aquatique et le terrestre (Vitt et Caldwell 2009). Les amphibiens ont un cycle vital particulier, défini par une combinaison d'états (morphologique, physiologique, comportemental) et d'environnements nécessaires afin de le compléter (Wilbur 1980; Werner 1986). Les caractéristiques telles que le développement et le type de reproduction varient énormément entre les différents groupes d'amphibiens à travers le monde. Par exemple, certains amphibiens ont un stade de développement totalement terrestre comme c'est le cas de la Salamandre rayée (*Plethodon cinereus*), que l'on retrouve dans l'est du Canada et des États-Unis (Sayler 1966; RÉCCAR [En ligne]).

Comme mentionné précédemment, la température corporelle des amphibiens dépend de l'environnement. Ainsi, la limite sud du Canada correspond à la limite nordique de plusieurs espèces, et certaines espèces ont des adaptations particulières afin de contrer le gel hivernal, comme c'est le cas de la Grenouille des bois (*Lithobates sylvaticus*) (Costanzo *et al.* 1993; Storey et Storey 1996). Dans la présente étude, nous ciblons exclusivement les anoures, incluant les grenouilles, les crapauds et les rainettes.

1.3.2 Cycle vital des anoures

Les anoures sont caractérisés par une métamorphose s'échelonnant entre les stades de l'œuf et de l'adulte (Figure 1.1, Déry [En ligne]).

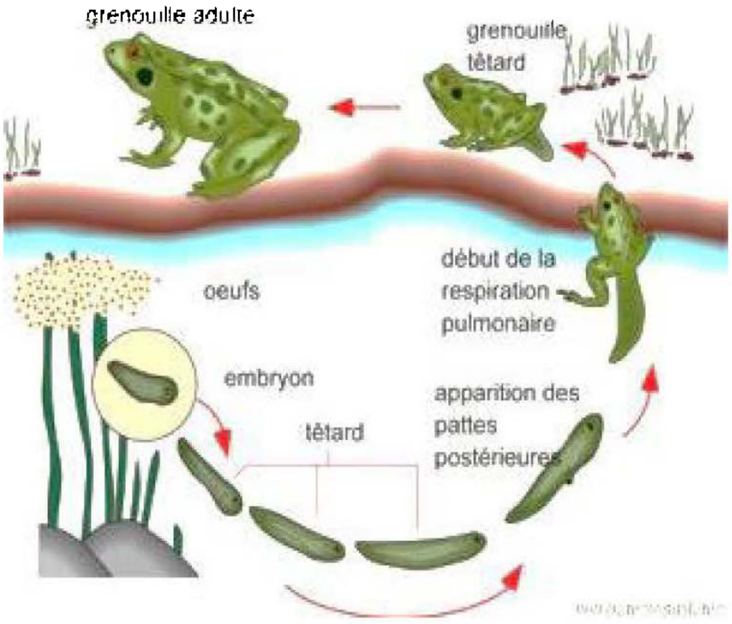

Figure 1.1 Cycle vital d'un anoure.

Comme indiqué en Annexe 1, les populations d'anoures du Québec connaissent également quelques variations au niveau de leur cycle vital. Par exemple, la période de stade larvaire est bien plus longue chez le Ouaouaron (*Lithobates catesbeianus*) que chez le Crapaud d'Amérique (*Anaxyrus americanus*), respectivement de 24 à 36 mois et 2 à 3 mois (ANNEXE 1). Néanmoins, chaque espèce connaît le même ordre général de phases, à savoir :

Œuf => Embryon => Têtard => Métamorphe => Adulte

De plus, selon l'espèce, les stades sont plus ou moins nombreux et définis lors du développement de l'embryon à l'adulte (Gosner 1960).

1.3.3 Alimentation

Au cours des différents stades de la métamorphose, les préférences alimentaires évoluent également. Alors que le stade « têtard » est principalement caractérisé par un état aquatique et un mode alimentaire herbivore, l'adulte est plutôt identifié en tant que carnivore terrestre (Wilbur 1980; Vitt et Caldwell 2009). Néanmoins, cette stratégie permet de diminuer la compétition entre les stades. Chez les espèces d'anoures, le stade larvaire représente principalement une période pour la croissance, alors que le stade terrestre est caractérisé par la dispersion et la reproduction (Wassersug 1975; Werner 1986; Berven 1990).

1.3.4 Dispersion et survie inter-stade

La dispersion est un élément primordial et limitant pour la survie des populations locales (Werner 1986). Cet élément sera abordé dans la section suivante avec le concept de métapopulations. La maturité sexuelle varie selon les espèces (Annexe 1) et est atteinte entre 1 et 6 année(s) chez les espèces retrouvées au Québec. Les individus doivent donc survivre jusqu'au stade adulte afin de perpétuer l'espèce et différentes stratégies sont employées. Une espèce qui a une espérance de vie de 4 ans mais qui n'atteint la maturité sexuelle que vers l'âge de 2 à 3 ans – comme c'est le cas pour le Crapaud d'Amérique (Annexe 1) – devra produire un nombre important d'œufs, puisque le taux de survie des premiers stades de vie est très faible et qu'à chaque stade une quantité importante de recrues est perdue (Wilbur 1980; Werner 1986; Berven 1990).

1.4 Rôle des métapopulations

La métapopulation est un concept important en écologie des populations. Il sous-entend que la persistance d'une population locale est déterminée par la capacité des populations à échanger des individus entres elles, ou à coloniser de nouveaux milieux (Hanski et Gilpin 1991; Hanski 1999). L'échange de gènes entres les différentes populations favorise également la persistance à long terme des espèces (Willi *et al.* 2006). Lorsque l'habitat d'une parcelle devient moins adéquat, ou lorsqu'un évènement stochastique intervient dans la qualité de ce dernier, une extinction de la population de la parcelle peut survenir (Hanski 1999). Dans ce concept de métapopulations, une population n'est pas statique dans le temps ou dans l'espace. Ainsi, il est possible d'observer des processus de recolonisation après une extinction locale (Hanski et Gilpin 1991; Lehtinen *et al.* 1999).

Ce concept s'applique à certaines espèces, pour lesquelles les ressources nécessaires à la réalisation du cycle vital débordent de la parcelle (Pope *et al.* 2000; Marsh et Trenham 2001). Il importe donc de comprendre la dynamique de colonisation et d'extinction des espèces principalement dans l'optique de conservation et protection de celles-ci.

1.5 Espèces à l'étude

1.5.1 Grenouille verte
(*Lithobates clamitans*) (Crother *et al.* 2000)

Cette espèce est commune et répandue au Québec (Fig. 1.2) (Desroches et Rodrigue 2004; AARQ [En ligne]). Elle est plutôt aquatique, fréquentant principalement les milieux humides de type permanents où elle hiberne au fond de l'eau l'hiver et s'y reproduit l'été, bien qu'il puisse être possible de la retrouver aux abords des milieux humides temporaires et sur la terre ferme (Desroches et Rodrigue 2004; Elliott *et al.* 2009; AARQ [En ligne]; RÉCCAR [En ligne]). Il s'agit d'un des plus grands anoures de la province, pouvant atteindre plus de 11 cm (Desroches et Rodrigue 2004; Elliott *et al.* 2009). La reproduction a lieu principalement aux mois de juin et juillet, mais peut parfois se prolonger jusqu'en août (Desroches et Rodrigue 2004; Elliott *et al.* 2009; RÉCCAR [En ligne]). La longévité exacte dans la nature est inconnue mais on sait qu'elle atteint sa maturité sexuelle entre 2 et 3 ans et que la durée du stade larvaire est de 12 à 15 mois (Desroches et Rodrigue 2004). Les femelles peuvent avoir de 3 à 4 pontes totalisant jusqu'à 5 300 œufs en une saison (Desroches et Rodrigue 2004; RÉCCAR [En ligne]).

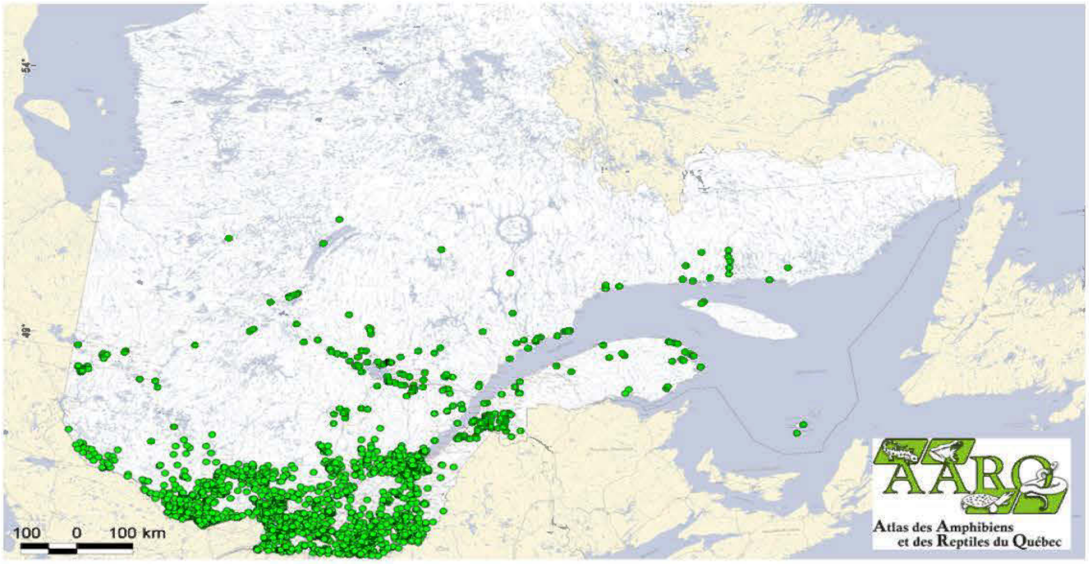

Figure 1.2 Répartition géographique connue de la Grenouille verte dans la province du Québec, Canada (AARQ [En ligne])[1].

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[1] Conception de la carte : Centre de Données sur le Patrimoine Naturel du Québec, 2009. Cette carte reflète de manière générale l'accès limité des biologistes au territoire, notamment via le réseau routier.

1.5.2 Grenouille du Nord
(*Lithobates septentrionalis*) (Crother *et al.* 2000)

Cette espèce est commune et répandue au Québec (Fig. 1.3) (Desroches et Rodrigue 2004; AARQ [En ligne]). Elle est sédentaire et vivement aquatique; fréquentant divers plans d'eau permanents toute l'année (Desroches et Rodrigue 2004; Elliott *et al.* 2009; AARQ [En ligne]; RÉCCAR [En ligne]). Desroches et Rodrigue (2004) avancent qu'il serait possible que les juvéniles de cette espèce aient moins tendance à se disperser par voie terrestre, comparativement aux autres espèces d'anoures, préférant utiliser les voies d'eau comme les ruisseaux. Adulte, elle peut atteindre jusqu'à 7 cm (Desroches et Rodrigue 2004; Elliott *et al.* 2009; AARQ [En ligne]; RÉCCAR [En ligne]). La période de reproduction a lieu au milieu de l'été, s'étalant de juin à août (Desroches et Rodrigue 2004; Elliott *et al.* 2009; RÉCCAR [En ligne]). La longévité exacte dans la nature n'est pas connue mais on sait qu'elle atteint sa maturité sexuelle, à l'instar de la grenouille verte, entre 2 et 3 ans et que la durée du stade larvaire est également de 12 à 15 mois (Desroches et Rodrigue 2004). La femelle pond ses œufs en masses lâches qui coulent au fond de l'eau et pouvant totaliser jusqu'à 1 700 œufs lors d'une saison (Desroches et Rodrigue 2004; RÉCCAR [En ligne]).

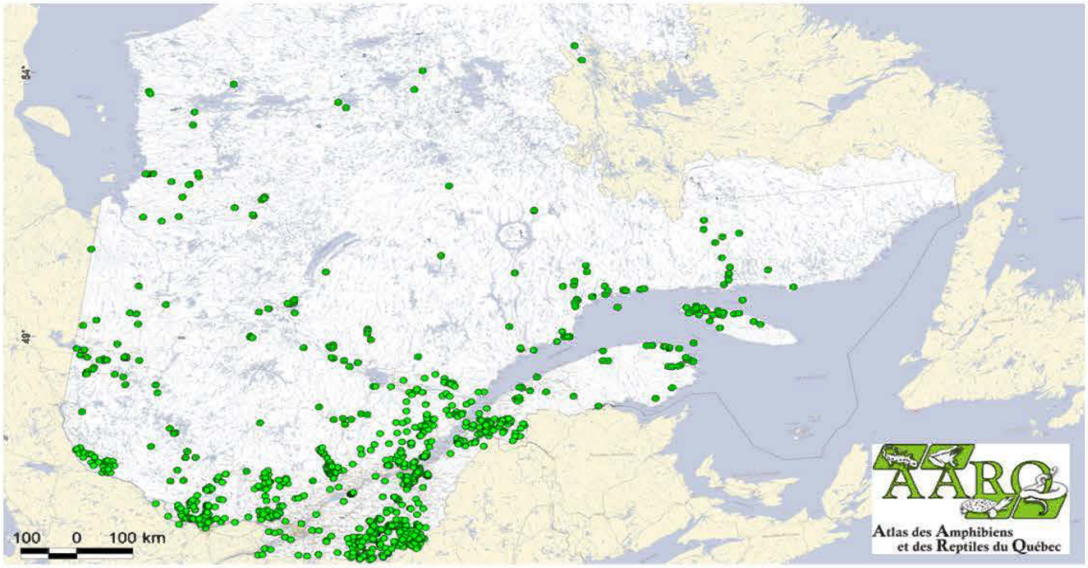

Figure 1.3 Répartition géographique connue de la Grenouille du Nord dans la province du Québec, Canada (AARQ [En ligne])[2].

1.5.3 Rainette crucifère
(*Pseudacris crucifer*) (Crother *et al.* 2000)

Cette espèce est très commune et répandue au Québec (Fig. 1.4) (Desroches et Rodrigue 2004; AARQ [En ligne]). Elle fréquente principalement les milieux terrestres; les forêts et boisés en régénérations, à proximité des milieux humides – qui peuvent être permanents ou temporaires – où elle ne va en réalité que pour se reproduire (Desroches et Rodrigue 2004; Elliott *et al.* 2009; AARQ [En ligne]; RÉCCAR [En ligne]). Les mâles chantent habituellement perchés sur des branches ou dans les herbes adjacentes aux milieux humides (Desroches et Rodrigue 2004; Elliott *et al.* 2009). Lors de la période d'hibernation, elle trouve habituellement refuge sous des bûches ou dans la litière forestière comme c'est une espèce tolérante au gel (Desroches et Rodrigue 2004; RÉCCAR [En ligne]). Adulte, elle n'atteint que rarement plus de 3 cm (Elliott *et al.* 2009; Desroches et Rodrigue 2004; AARQ [En

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[2] Conception de la carte : Centre de Données sur le Patrimoine Naturel du Québec, 2009. Cette carte reflète de manière générale l'accès limité des biologistes au territoire, notamment via le réseau routier.

ligne]; RÉCCAR [En ligne]). La période de reproduction débute très tôt à la fin du printemps et peut s'étaler jusqu'au mois d'août selon l'aire de distribution (Desroches et Rodrigue 2004; Elliott *et al.* 2009; RÉCCAR [En ligne]). La longévité est de 2 à 4 ans alors que la maturité sexuelle est atteinte vers 2 ans et que la transformation larvaire prend de 1,5 à 3,5 mois (Desroches et Rodrigue 2004; RÉCCAR [En ligne]). La femelle pond jusqu'à 1 600 œufs en une saison, soit individuellement ou en petits amas, qui s'accrochent à la végétation ou coulent tout simplement au fond de l'eau (Desroches et Rodrigue 2004; RÉCCAR [En ligne]).

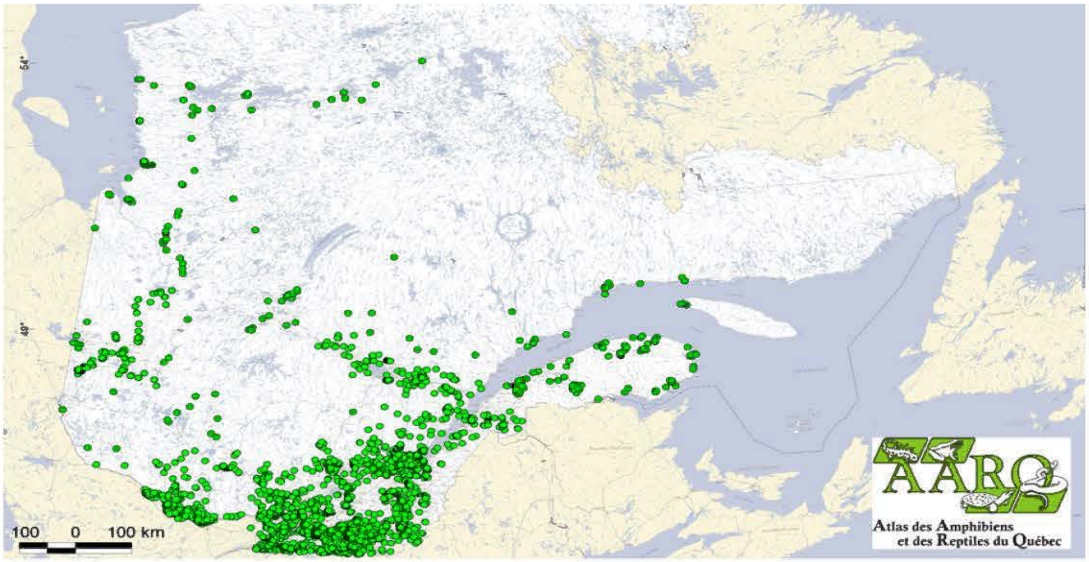

Figure 1.4 Répartition géographique connue de la Rainette crucifère dans la province du Québec, Canada (AARQ [En ligne])[3].

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[3] Conception de la carte : Centre de Données sur le Patrimoine Naturel du Québec, 2009. Cette carte reflète de manière générale l'accès limité des biologistes au territoire, notamment via le réseau routier.

1.6 Hypothèses de recherche

1.6.1 Hypothèse 1 : La distance à une route

La composition du paysage environnant les milieux humides est souvent associée à la probabilité d'occurrence des amphibiens en fonction de la biologie propre des espèces, et pour plusieurs d'entre-elles, le couvert forestier autour d'un milieu humide joue un rôle aussi important que le milieu humide – en fournissant une protection contre les prédateurs et un lieu pour l'hibernation (Guerry et Hunter 2002). En effet, si tous les éléments essentiels au cycle biologique de l'animal se trouvent à proximité du milieu humide, les individus auront moins tendance à migrer. Comme le mentionnent Guerry et Hunter (2002), le milieu humide est principalement employé pour la reproduction, alors que la terre ferme sert à la recherche de nourriture et à l'hibernation. Nous croyons que plus un milieu humide sera situé près d'une route, plus la probabilité de colonisation sera faible et plus la probabilité d'extinction sera grande. Ces relations seront toutefois mitigées par la quantité de couvert forestier autour des milieux humides. Cette hypothèse se fonde également sur l'important taux de mortalité qui survient au moment de la période de reproduction, lors de la migration vers un site de ponte quand les individus doivent traverser une route – puisque certains d'entre eux peuvent franchir une distance de plus de 500 m afin de trouver un milieu adéquat– et pouvant atteindre plus de 30% de mortalité chez certaines espèces de crapaud (van Gelder 1973; Oldham 1966). De plus, certains auteurs mentionnent que l'effet d'une route peut s'étendre à plus de 100 m en bordure de celle-ci, en fonction du type de route et des différents effets combinés aux facteurs environnementaux (Forman 2000; Forman et Deblinger 2000).

1.6.2 Hypothèse 2 : Le type de route

Comme le suggèrent Hels et Buchwald (2001) dans leurs travaux, plus l'intensité de la circulation est importante, plus le risque pour un amphibien –traversant une route–

de se faire frapper sera grand. Par ailleurs, les routes plus importantes créent une ouverture plus grande dans le paysage (*i.e.* un milieu ouvert plus grand à traverser), ce qui pourrait influencer la probabilité de traverser des amphibiens, surtout si la route sépare les éléments déterminants du cycle vital des anoures. Dans cet ordre d'idées, nous croyons qu'une route importante en taille telle qu'une route principale asphaltée, aura une plus importante circulation automobile qu'une route non asphaltée, où la circulation automobile est considérée nettement plus faible. Ainsi, nous émettons l'hypothèse que, plus la circulation automobile sera importante sur une route donnée (*i.e.* plus le type de route sera important, à savoir : route principale asphaltée > route secondaire asphaltée > route non asphaltée), plus la probabilité de colonisation du milieu sera faible, et plus la probabilité d'extinction du milieu sera grande. Comme expliqué précédemment, il est également possible que cette relation soit nuancée par la quantité de couvert forestier autour du milieu humide. Par exemple, bien qu'il soit prédit que la probabilité d'extinction sera plus grande en milieux humides à proximité d'une route principale asphaltée, cet effet pourrait être amoindri par le couvert forestier (*i.e.* plus le couvert forestier serait important aux abords d'une route où la circulation est élevée, moins la probabilité d'extinction sera importante, puisque les individus auraient moins tendance à la traverser). Autrement dit, la présence d'un important couvert forestier pourrait tamponner l'effet d'une route principale asphaltée à 250 m d'un milieu humide sur les populations d'amphibiens, conformément aux zones tampons minimales proposées par Semlitsch et Bodie (2003).

De plus, bien qu'il soit prédit que la probabilité d'extinction sera plus grande en présence d'une route principale asphaltée comparativement à une route non asphaltée, la distance à la route devrait nuancer l'effet. Par exemple, l'effet d'une route principale asphaltée pourrait se prolonger plus loin dans la forêt que celle d'une route non asphaltée. Ainsi, la probabilité d'extinction devrait diminuer avec la distance à la route, mais cette relation serait plus prononcée pour les routes non asphaltées étant donné l'impact négatif moins important sur la qualité du milieu pour ce type de route.

1.6.3 Hypothèse 3 : Probabilité de détection

Tel que suggéré par Oseen et Wassersug (2002), certaines variables environnementales peuvent faire varier la fréquence et l'intensité du chant des anoures. Lors d'échantillonnage par stations d'écoute, les ondes sonores doivent être perçues de manière optimale afin de détecter la présence des espèces concernées. La probabilité de détection des espèces est susceptible de varier en fonction de plusieurs paramètres. Ainsi, les facteurs tels que la température, le vent, la pluie et le passage d'automobiles sont jugés les plus à même de réduire les possibilités de détection des espèces. De plus, le nombre de jours depuis la fonte des neiges (SnowDay) devrait également influencer le chant des amphibiens au cours de la saison étant donné que chaque espèce a une période de reproduction spécifique et que les conditions environnementales varient d'une année à l'autre (Annexes 1 et 2). Ces variables seront considérées dans les différents modèles.

1.7 Objectifs de recherche

À la lumière de ces informations, et spécialement dû à l'impressionnante progression du réseau routier au Québec, il apparaît important, voire urgent, d'enrichir les connaissances que nous avons de l'impact réel des routes sur la faune, et particulièrement sur les amphibiens. L'objectif principal de ce projet est de déterminer l'impact des routes sur la dynamique de colonisation et d'extinction des populations d'amphibiens en milieux humides. Cette étude vient donc s'insérer dans le cadre actuel de la perte de biodiversité mondiale parmi laquelle les amphibiens sont surreprésentés.

CHAPITRE II

IMPACT DES ROUTES SUR LA DYNAMIQUE DE COLONIZATION ET D'EXTINCTION D'AMPHIBIENS EN MILIEUX HUMIDES

The impacts of roads on colonization and extinction dynamics of wetland amphibians.

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2.1 Abstract

Habitat degradation and fragmentation are the leading factors affecting the loss of biodiversity. Functional connectivity between local populations is essential for the long term viability of metapopulations. The fragmentation and deterioration of amphibian habitat following the establishment of roads may pose a serious problem for population persistence. The purpose of our study was to quantify the impact of roads on colonization and extinction dynamics of wetland amphibians. We selected 54 wetlands in each of two study areas, one located in the vicinity of Plaisance National Park (Northern temperate zone) and another in Aiguebelle National Park (Boreal zone), in Québec, Canada. Using a stratified sampling design, we randomly selected wetlands along a gradient of road distances from wetlands (< 50 m, 50-100 m, > 100 m) and road types (unpaved, secondary asphalted, primary asphalted). Each site was visited 3 to 4 times per season for three breeding seasons (2012 - 2014). We used call surveys to detect anurans at each site. We also determined road density (km/km^2), wetlands proportion around ponds and the proportion of different types of forest cover in these 500 m buffers. We used dynamic occupancy models to estimate the occupancy, extinction, and colonization parameters, after accounting for imperfect detectability. The results for the Green Frog (*Lithobates clamitans*) indicate that the extinction probability decreased with the wetland pH and increased with the proportion of conifer cover within 500 m buffer. The probability of detection increased with the number of days since snow melt (SnowDay). For the Mink Frog (*Lithobates septentrionalis*), the probability of colonization increased with wetland size. For the Spring Peeper (*Pseudacris crucifer*), the effect of road distance on the occupancy probability depended on forest cover (forest cover x road distance) within 500 m buffer. For a 50 m distance to a road, the occupancy probability decreased with mixed forest type cover, whereas occupancy probability increased for distances of 100 m and 150 m from a road. At 50 m from a road, occupancy probability increased with deciduous forest type covered, whereas for distances of 100 m and 150 m to a road, the occupancy probability decreased. Moreover, the probability of detection decreased with SnowDay. The differences in observed results among the species under study can be explained by the intrinsic variabilities, particularly regarding habitat preference and the timing of the breeding season. Although we found no impact of road types and road distance on colonization and extinction dynamics of wetland amphibians, we suggest that more long-term monitoring is required considering that road networks are still expanding in our study areas.

Keys words: anurans, wetlands, roads, extinction, colonization, forest cover; occupancy.

2.2 Introduction

The world population has grown from 1.6 billion to over 7.2 billion people in 2015 since the beginning of the last century (United Nations 2015; FAO 2015). To ensure that the needs of this growing population are met, an increasing amount of natural resources are being used (FAO 2015). For example, the agricultural sector as well as the forestry industry have undergone considerable pressures (FAO 2015). The former to meet food demands and the latter to provide construction wood, pulp, and paper. Moreover, technological changes have resulted in an increase in the consumption of various goods, resulting in a demand for raw materials including fossil fuels (FAO 2015). Following the significant urban development, road networks have also experienced an impressive increase in order to allow the distribution of these resources (Central Intelligence Agency 2009; United Nations 2015).

Several studies worldwide suggest that the current extinction rate of species is greater than the normal rate estimated from the fossil record and that it will continue to increase over the next century (Sala *et al.* 2000; Ricciardi and Rasmussen 2001; Green 2003; Baillie 2004). According to the Red List of Threatened Species of the International Union for the Conservation of Nature (IUCN), habitat loss resulting from commercial and residential development, amplification of the transport networks, forestry activities, and other ecosystem changes due to growth and development of the human population accounts for over 66% of animal species threatened worldwide. Amphibians represent 36.5% of the world’s endangered species (i.e. 4 243/11 627 endangered species) and are the group most affected by these changes, followed by mammals (22.3%), birds (21.7%), and reptiles (19.4%) (IUCN [Online]).

Given their biology and their life cycle, amphibians are sometimes used as biological indicators (Wake 1991). It is important to understand the mechanisms involved in the decline of amphibian populations as their health is closely related to the « health of the environment ». Like reptiles and fish, amphibians are ectotherms: their body temperature depends on the ambient temperature and their metabolism depends on the climate (Vitt and Caldwell 2009). Their skin is highly permeable, devoid of protection from dehydration, and is used for breathing and drinking. These characteristics make them sensitive to changes in environmental conditions (Alford and Richards 1999; Vitt and Caldwell 2009). The role of amphibians is critical in many food chains, mainly those in wetlands (Lehtinen *et al.* 1999). As they occupy different ecological niches during their life cycle, they can fulfill their roles from food source to predator (Burton and Likens 1975). Moreover, adult anurans (frogs, toads, and treefrogs) have an important role in controlling insects, but they are also an important source of food for many birds and mammals. As such, the amphibian biomass is a key element in maintaining the environments they colonize (Gibbons *et al.* 2006).

Metapopulation theory is an important concept in population ecology. It implies that the persistence of a local population is determined by the ability of populations to exchange individuals or to colonize new environments (Hanski and Gilpin 1991; Hanski 1999). The gene exchange between the different populations also promotes long-term persistence of species (Willi *et al.* 2006). When habitat patches decrease in quality or following stochastic events, an extinction of the population in the patch may occur (Hanski 1999). In this concept of metapopulation, a population is not static in time or in space. Thus, it is possible to observe recolonization after extinction processes (Hanski and Gilpin 1991; Lehtinen *et al.* 1999). This concept is also important for species that are characterised by special habitat needs to complete their life cycle, such as is the case for amphibians (Pope *et al.* 2000; Marsh and Trenham 2001). It is therefore important to understand the dynamics of colonization and extinction of these species, mainly in the perspective of protection and conservation.

In the 1980's, amphibian declines became a concern for scientists around the world, especially following the first World Congress of Herpetology, in 1989 (Barinaga 1990; Bradford 1991; Wake 1991; Crump *et al.* 1992; Blaustein *et al.* 1994 ; Gardner 2001; Green 2003; Stuart *et al.* 2004). Since then, many scientists have tried to explain the potential causes of these declines and species extinctions. Different factors have been suggested as potential causes of these declines. For example, increased ultraviolet (UV) radiation, acid rain, the intensification of agriculture, climate change, increased pollution levels, chemical contamination of the environment, the removal of individuals in nature, are all plausible causes for the decline of amphibians in the world (Blaustein *et al.* 1994; Daszak *et al.* 1999; Bridges and Semlitsch 2001; Blaustein *et al.* 2003; and Sparling *et al.* 2009). Nevertheless, although some threats are local, fragmentation and habitat destruction appear to have the greatest impacts and their importance is exacerbated by their extent (Knutson *et al.* 2001; Baillie 2004). In addition to destroying and modifying the environment (Sanzo and Hecnar 2006; Fahrig and Rytwinski 2009), roads generate considerable pollution that exceed the natural intake of salts (mainly NaCl, using as winter deicer on roads) (Karraker *et al.* 2008). Indeed, some authors report that the increase in roadside salt could increase the risk of malformations, causing an imbalance in osmoregulation and reducing the survival of individuals (Karraker *et al.* 2008; Reeves *et al.* 2008; Karraker and Gibbs 2011).

Several researchers have studied the impact of roads on wildlife populations in recent decades (van Gelder 1973; Fahrig *et al.* 1995; Forman and Alexander 1998; Hels and Buchwald 2001; Spellerberg 2002; Fahrig and Rytwinski 2009; Benítez-López *et al.* 2010; Van der Ree *et al.* 2011). In some cases, changes in morphological and behavioral levels in amphibians living near roads were reported (Spellerberg 2002; Mazerolle *et al.* 2005; Sanzo and Hecnar 2006; Bouchard *et al.* 2009; Parris *et al.* 2009; Hoskin and Goosem 2010). Indeed, because of their low dispersal capacity and certain traits, amphibians appear to be particularly affected by the creation of roads near their habitat (Wilbur 1980; Cushman 2006; Becker *et al.* 2009). However, despite the apparent increase in mortality in amphibian populations on roadsides (Fahrig *et al.* 1995; Forman and Alexander 1998; Spellerberg 2002; Mazerolle 2004; Fahrig and Rytwinski 2009), few researchers have actually studied road impacts on population dynamics (Lesbarrères and Fahrig 2012).

In 2015, the global road network spanned more than 64 285 009 km (Central Intelligence Agency 2015 [Online]). In 1734, the province of Québec accounted only 267 km of roads between the cities of Montréal and Québec (Gilchrist 2012 [En ligne]). Less than 300 hundred years later, the network has grown to more than 185 000 km of roads (Transports Québec [En ligne]). Furthermore, during the period 1985 to 2000, the number of passenger vehicles increased by 68% and the number of heavy trucks increased by 34.5% (Transports Québec [En ligne]). With 1 042 300 km of roads, Canada is in the 7th position in terms of importance of its road network (Central Intelligence Agency 2015[Online]).

In light of this information, especially due to the impressive growth of the road network in Québec/Canada, it is important to enrich the knowledge on the impact of roads on wildlife, particularly on amphibians. The main objective of this project is to determine the impact of roads on the dynamics of colonization and extinction of amphibian populations in wetlands.

2.3 Methods

2.3.1 Study areas and site selection

The project was initiated in spring 2012 in two regions of western province of Quebec, Canada. The first region is in the vicinity of Plaisance National Park (45°35'44"N, 75°05'34"W) and is included into the northern temperate zone

(Ministère des Forêts, de la Faune et des Parcs [En ligne])(Figure 2.1). The second region is located in the vicinity of Aiguebelle National Park (48°30'33"N,78°44'57"W) and is included in the boreal zone (Ministère des Forêts, de la Faune et des Parcs [En ligne]) (Figure 2.1).

All wetlands near a road were initially identified using Google^{tm}earth within a radius of 50 km of each National Park. We identified 309 and 317 wetlands within the radii of 50 km of the national parks of Plaisance and Aiguebelle, respectively.

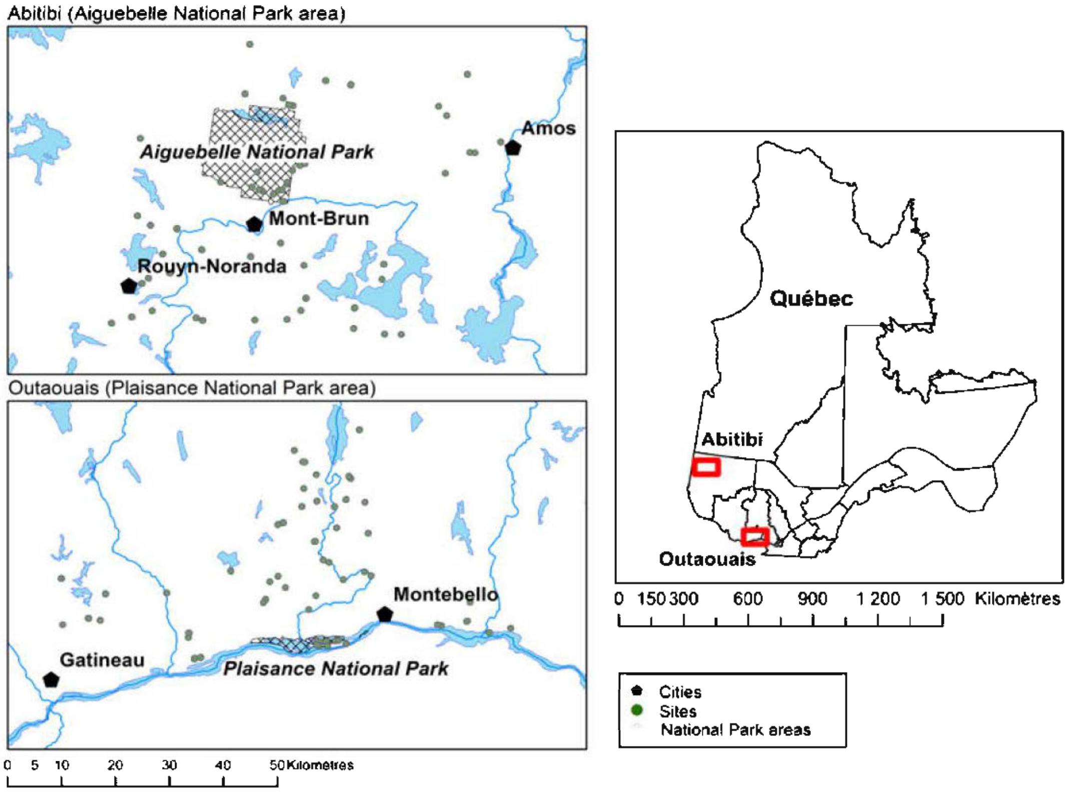

Figure 2.1 Site distribution in the two study areas of the province of Québec, Canada.

We stratified our wetland selection according to road type and distance to road. We considered three types of roads, namely; « unpaved », « secondary asphalted », and « primary asphalted ». The road type depends largely on the volume of car traffic and the speed limit. « Unpaved roads » are characterized by a lower number of cars and a rather slow movement of vehicles. In our study, we had a mean car traffic of 4.6 cars/hour for that category. These roads are mostly found in the countryside, are not often used, and the speed limit is ordinarily < 50 km /h. « Secondary asphalted » roads are characterized by a medium number of cars and speed limit. We had mean car traffic of 25 cars/hour for that category. These roads are mostly found into cities or residential areas and the speed limit is usually between 50 – 90 km/h. « Primary asphalted roads » are characterized by a higher number of cars as they link main cities and present a higher speed limit (≥ 90 km/h). We had mean car traffic of 166.2 cars/hour in our study for that category of road.

Also, here, the term « road », refers to an infrastructure where car traffic occurs, with a minimum road surface width of 8 m and a maximum road surface width of 14 m. Paths or trails were not treated as roads in our study, as we focused on areas impacted by motorized vehicle traffic. We considered three different road distance categories, namely: < 50 m, 50-100 m, and > 100 m. We grouped into clusters of distance sites for analysis but they have been conducted as numerical datas. We randomly selected 6 wetlands for each combination of road type and distance in each study area. Consequently, 54 sites were selected for each of the two regions (Table 2.1).

Table 2.1 Distribution of sites according to the selection criteria.

	Road types		
Road distance	**unpaved**	**secondary asphalted**	**primary asphalted**
< 50 m	6	6	6
50-100 m	6	6	6
> 100 m	6	6	6

During our selection, we added a constraint of maintaining a minimum distance of 500 m between any two selected wetlands to provide independent observations, since the movements of most species studied are generally lower than 500 m (Semlitsch and Bodie 2003). During the study, certain sites were abandoned following disturbances (urbanization, forest clearing), and in such cases, sites were replaced by other sites of the same road type x distance category.

2.3.2 Amphibian populations in the study area

To determine the impact of roads on amphibian populations, this project targeted populations of anurans present in both study areas. We selected two species of frogs from the *Ranidae* family, the Green Frog (*Lithobates clamitans*) and the Mink Frog (*Lithobates septentrionalis*), and one species among *Hylidae,* the Spring Peeper (*Pseudacris crucifer*).

2.3.3 Sampling by call surveys

Because male anurans call during the breeding season and the calls of each species are easily identifiable, we used call surveys to detect anurans at each site. We conducted call surveys at stations on the perimeter of wetlands. Our sampling effort increased with the size of the wetland, with two call survey stations sampled for the first 25 m^2 of a site area, and with an additional station each time the wetland doubled in size (Adams *et al.* 1997; Mazerolle *et al.* 2014). The number of call survey stations for the study sites ranged from 2 to 11 stations.

2.3.4 Description of site visits

Sampling was conducted between 4 pm and midnight, as it is normally the most active period of the anurans under study (Oseen and Wassersug 2002). To facilitate sampling, sites were grouped into clusters of proximate sites. We randomly selected these clusters to sample them in a random sequence to avoid visiting a given site at the same time of day. During a visit to a site, we listened for three minutes at each sampling station to detect the species studied. We did not sample during strong wind (> 12 km/h) or heavy rain. Sites were visited between three and four times per year.

2.3.5 Sampling variables

For each visit to a site, we measured the water temperature (± 0.1°C) of the pond at every two sampling stations to compute an average water temperature for each site and visit. For example, we took 5 water temperature samplings for a site with 10 sampling stations. We also recorded the occurrence of rain during call surveys.

Because the study areas are at different latitude and the study spanned three years, we used the number of days elapsed after snow melt (SnowDay) to yield a comparative measure of date for each area. Indeed, we expected amphibians to respond to environmental factors such as the number of days since snow melt rather than the actual Julian date. We obtained the SnowDay data from the Environment Canada climate archives (Government of Canada [Online]) where they kept information about the last day of snow track for both region. We used the city of Montebello (Plaisance national Park area) and the locality of Mont-Brun (Aiguebelle national Park area) to get the information from the archives (Figure 2.1).

We measured car traffic during each survey at each station. A mean number of vehicles per 3 minutes at each station (sum of vehicles heard/number of stations) has been calculated to get a unique value of traffic for each site visit.

2.3.6 Site and landscape variables

First, we determined the wetland types (pond, marsh, lake, or other) by following the definitions from Ducks Unlimited Canada [Online] and Bazoge *et al.* (2014) to ensure that all sites were wetlands recognized by an official organization. We measured pH and electrical conductivity at each site once during season 2013 and twice during season 2014 (pHTestr®30, Oakton® instruments, ±0.01 pH accuracy; ECTestr^{TM} 11 Plus, Oakton® instruments, 0 - 2000 µS range). In July 2013, we characterized the peripheral vegetation (plants and trees) in a 30 m buffer around the ponds by establishing 30 x 3 m transects divided into three sections of 10 x 3 m. Then, we established three different height strata (0-30 cm, 30 cm-3 m, and > 3 m). Our peripheral vegetation sampling efforts increased with the size of the pond, with three transects for the first 25 m^2 of a site area, and with an additional transect each time it doubled in size. The number of transects per site varied between 3 and 5, depending on the size of ponds. We classified vegetation cover of the different groups using the following five classes: < 5%, [5% - 25%], [25% - 50%], [50 - 75 %] and > 75%. We summarized the vegetation data with principal components analysis (PCA) and used the first scores as explanatory variables in our analyses. We established buffers within a 500 m radius around wetlands - corresponding to the maximal dispersal ability of species under consideration - using ARCGIS software Version 10.2® to quantify the proportion of forest cover types (deciduous / conifer / mixed) (Semlitsch and Bodie 2003). We also determined road density (km/km^2) and the proportion of wetland cover around ponds in these 500 m buffers as they describe the landscape, by using ARCGIS software again.

Table 2.2 Summary of the sites and call surveys variables.

Site variables	**Sampling variables**
• PCA scores vegetation in a 30 m buffer	• Presence/Absence of species
• Size of the pond (ha)	• Water temperature (°C)
• Road distance (m)	• Rain (1/0)
• Road type (unpaved /secondary asphalted / primary asphalted)	
• Proportion of forest cover of each type (deciduous, conifer, mixed) in a 500 m buffer	• Car traffic
• Road density (km/km^2) in a 500 m buffer	• SnowDay
• Proportion of wetlands cover in a 500 m buffer	
• pH	
• Conductivity	

2.3.7 Statistical analyses

All analyses were performed in R Version 3.2.2 (R Development Core Team 2015). We centered and reduced all variables to zero mean and unit standard deviation for analysis. We computed the Pearson product moment correlation coefficient (*r*) among variables and we avoided including pairs of variables with $|r| > 0.7$ in the same model.

Dynamic occupancy models

We used dynamic occupancy models to estimate the probabilities of initial occupancy, extinction, colonization, and detection at the sampled sites for the three seasons (MacKenzie *et al.* 2003). We formulated different models according to our hypotheses (Table 2.3 – 2.6). We tested our hypotheses separately for each species. We obtained the maximum likelihood estimate of the parameters using the unmarked package in R (Fiske *et al.* 2011).

Dynamic occupancy models assume that 1) the occupancy state (ψ) is the same between the first and last visit to a site during a season (year), 2) the probability of occurrence is explained by the explanatory variables of interest, 3) the probability of detecting a species (p) during a visit is adequately modeled by the variables of interest, and that 4) the detections are independent of each other (MacKenzie *et al.* 2006; Mazerolle *et al.* 2007). Dynamic occupancy models estimate the probabilities of extinction (ε) and colonization (γ) at sites between two sampling seasons to describe the change in occupancy across seasons and these parameters can be modeled using explanatory variables with a logit link.

We compared the candidate models using the Akaike information criterion and used multimodel inference implemented in the AICcmodavg package (Akaike 1973; Burnham and Anderson 2002; Mazerolle 2006, Mazerolle 2015). We used a parametric bootstrap approach based on the chi-square statistic with 5000 bootstrap samples to assess the goodness of fit for the top-ranked models for each species implemented in the AICcmodavg package (Mazerolle 2015).

2.3.8 Biological hypotheses

Hypothesis 1: Road distance

The composition of the landscape surrounding wetlands is often associated with the probability of occupancy of amphibians according to the specific biology of the species. For many species, forest cover around a moist environment also plays a role as important as the wetland itself (Guerry and Hunter 2002). In addition, some authors mention that ecological effects of road on wildlife can extend beyond 100 m, depending on the type of road and combined effects of different environmental factors (Forman and Alexander 1998; Forman 2000; Forman and Deblinger 2000). We believe that colonization probability is greater at larger distances from a road than close to roads, and that extinction probability is greatest close to roads. We hypothesize that these relationships are mitigated by the amount of forest cover around wetlands. Indeed, if all the essential elements in the anuran life cycle are nearby wetlands, adults are less likely to migrate far from roads. As mentioned by Guerry and Hunter (2002), the wetland is mainly used for reproduction, while upland habitats are used for foraging and hibernation. Thus, the distance between the two environments is important. This assumption is also based on the high mortality rate that occurs during the breeding season (van Gelder 1973; Oldham 1966).

Hypothesis 2: Road type

As Hels and Buchwald (2001) suggest, the mortality risk of amphibians crossing a road increases with the traffic intensity. Moreover, roads can increase the probability of crossing when they separate different habitat components of the amphibian life cycle. We hypothesize that colonization probability is lowest in ponds next to roads with high traffic (main road) than low traffic (unpaved road). We also hypothesize that the extinction probability is greatest in ponds next to roads under high traffic than low traffic. We expect that these relationships are mitigated by the amount of forest cover around the wetland. For example, although it is predicted that the extinction probability will be greater in wetlands near a main asphalt road, this effect may be lessened by the forest cover. Although we predict that the extinction probability is greater in the presence of a main asphalted road compared to an unpaved road, the effect should decrease with distance. For example, the effect of a main paved road could extend further into the forest than an unpaved road. Thus, the probability of extinction should decrease with distance from the road, but this relationship may be weaker for unpaved roads given the less negative impact on the quality of the environment for this road type.

Hypothesis 3: Detection probability

As suggested by Oseen and Wassersug (2002), anurans can vary the frequency and intensity of their calls with some environmental variables. Factors such as the water temperature, wind, rain, and car traffic may reduce detectability of species during call surveys. Furthermore, the number of days elapsed following snow melt (SnowDay) should also influence the singing of amphibians during the season as each species has a specific breeding period. All these parameters are chose as the best to describe the detection probability of the species under study and will be included into the different models.

Table 2.3 Hypotheses and models related to the initial occupancy probability.

Models	**Hypotheses**
ψ(.)	The initial occupancy probability is constant.
ψ(Rd_Dist + Rd_Typ)	The initial occupancy probability varies with road distance and additive effect of road type.
ψ(Rd_Dist + Rd_Typ + Rd_Dist:Rd_Typ)	The initial occupancy probability varies with road distance x road type interaction.
ψ(pH + Conductivity + Per_Veg)	The initial occupancy probability varies with additive effects of pH, conductivity, and peripheral vegetation.

ψ(Landscape)	The initial occupancy probability varies with additive effects of road density and wetland proportion within a buffer of 500 m.
ψ(ForCov_F + Rd_Typ)	The initial occupancy probability varies with additive effects of deciduous forest type cover within a buffer of 500 m and road type.
ψ (ForCov_M + Rd_Typ)	The initial occupancy probability varies with additive effects of mixed forest type cover within a buffer of 500 m and road type.
ψ(ForCov_C + Rd_Typ)	The initial occupancy probability varies with additive effects of coniferous forest type cover (buffer 500 m) and road type.
ψ(ForCov_F + Rd_Typ + ForCov_F:Rd_Typ)	The initial occupancy probability varies with interactive effects of deciduous forest type cover within a buffer of 500 m and road type.
ψ(ForCov_M + Rd_Typ + ForCov_M:Rd_Typ)	The initial occupancy probability varies with interactive effects of mixed forest type cover within a buffer of 500 m and road type.
ψ(ForCov_C + Rd_Typ + ForCov_C:Rd_Typ)	The initial occupancy probability varies with interactive effects of coniferous forest type cover within a buffer of 500 m and road type.
ψ(ForCov_F + Rd_Dist)	The initial occupancy probability varies with additive effects of deciduous forest type cover within a buffer of 500 m and road distance.
ψ(ForCov_M + Rd_Dist)	The initial occupancy probability varies with additive effects of mixed forest type cover within a buffer of 500 m and road distance.
ψ(ForCov_C + Rd_Dist)	The initial occupancy probability varies with additive effects of coniferous forest type cover within a buffer of 500 m and road distance.

ψ(ForCov_F + Rd_Dist + ForCov_F:Rd_Dist)	The initial occupancy probability varies with interactive effects of deciduous forest type cover within a buffer of 500 m and road distance.
ψ(ForCov_M + Rd_Dist + ForCov_M:Rd_Dist)	The initial occupancy probability varies with interactive effects of mixed forest type cover within buffer of 500 m and road distance.
ψ(ForCov_C + Rd_Dist + ForCov_C:Rd_Dist)	The initial occupancy probability varies with interactive effects of coniferous forest type cover within a buffer of 500 m and road distance.
ψ(Size + Rd_Dist)	The initial occupancy probability varies with additive effects of pond area and road distance.
ψ(Size + Rd_Dist + Size:Rd_Dist)	The initial occupancy probability varies with interactive effects of pond area and road distance.

Table 2.4 Hypotheses and models related to the probability of colonization.

Models	Hypotheses
γ(.)	The colonization probability is constant.
γ(Rd_Dist + Rd_Typ)	The colonization probability varies with additive effects of road distance and road types.
γ(Rd_Dist + Rd_Typ + Rd_Dist:Rd_Typ)	The colonization probability varies with interactive effects of road distance and road type.
γ(pH + Conductivity + Per_Veg)	The colonization probability varies with additive effects of pH, conductivity, and peripheral vegetation.
γ(Landscape)	The colonization probability varies with additive effects of road density and wetland proportion within a buffer of 500 m.
γ(ForCov_F + Rd_Typ)	The colonization probability varies with additive effects of deciduous forest type cover within a buffer of 500 m and road type.

γ (ForCov_M + Rd_Typ)	The colonization probability varies with additive effects of mixed forest type cover within a buffer of 500 m and road type.
γ(ForCov_C + Rd_Typ)	The colonization probability varies with additive effects of coniferous forest type cover within a buffer of 500 m and road type.
γ(ForCov_F + Rd_Typ + ForCov_F:Rd_Typ)	The colonization probability varies with interactive effects of deciduous forest type cover within a buffer of 500 m and road type.
γ(ForCov_M + Rd_Typ + ForCov_M:Rd_Typ)	The colonization probability varies with interactive effects of mixed forest type cover with a buffer of 500 m and road type.
γ(ForCov_C + Rd_Typ + ForCov_C:Rd_Typ)	The colonization probability varies with interactive effects of coniferous forest type cover within a buffer of 500 m and road type.
γ(ForCov_F + Rd_Dist)	The colonization probability varies with additive effects of deciduous forest type cover within a buffer of 500 m and road distance.
γ(ForCov_M + Rd_Dist)	The colonization probability varies with additive effects of mixed forest type cover within a buffer of 500 m and road distance.
γ(ForCov_C + Rd_Dist)	The colonization probability varies with additive effects of coniferous forest type cover within a buffer of 500 m and road distance.
γ(ForCov_F + Rd_Dist + ForCov_F:Rd_Dist)	The colonization probability varies with interactive effects of deciduous forest type cover within a buffer of 500 m and road distance.
γ(ForCov_M + Rd_Dist + ForCov_M:Rd_Dist)	The colonization probability varies with interactive effects of mixed forest type cover within a buffer of 500 m and road distance.
γ(ForCov_C + Rd_Dist + ForCov_C:Rd_Dist)	The colonization probability varies with interactive effects of coniferous forest type cover within a buffer of 500 m and road distance.

γ(Size + Rd_Dist)	The colonization probability varies with additive effects of pond area and road distance.
γ(Size + Rd_Dist + Size:Rd_Dist)	The colonization probability varies with interactive effects of pond size and road distance.

Table 2.5 Hypotheses and models related to the extinction probability.

Models	Hypotheses
ϵ (.)	The extinction probability is constant.
ϵ(Rd_Dist + Rd_Typ)	The extinction probability varies with additive effects of road distance and road type.
ϵ(Rd_Dist + Rd_Typ + Rd_Dist:Rd_Typ)	The extinction probability varies with interactive effects of road distance and road type.
ϵ(pH + Conductivity + Per_Veg)	The extinction probability varies with additive effects of pH, conductivity, and peripheral vegetation.
ϵ(Landscape)	The extinction probability varies with additive effects of road density and wetland proportion within a buffer of 500 m.
ϵ(ForCov_F + Rd_Typ)	The extinction probability varies with additive effects of deciduous forest type cover within a buffer of 500 m and road type.
ϵ(ForCov_M + Rd_Typ)	The extinction probability varies with additive effects of mixed forest type cover within a buffer of 500 m and road type.
ϵ(ForCov_C + Rd_Typ)	The extinction probability varies with additive effects of coniferous forest type cover within a buffer of 500 m and road type.
ϵ(ForCov_F + Rd_Typ + ForCov_F:Rd_Typ)	The extinction probability varies with interactive effects of deciduous forest type cover within a buffer of 500 m and road type.

ϵ(ForCov_M + Rd_Typ + ForCov_M:Rd_Typ)	The extinction probability varies with interactive effects of mixed forest type cover within a buffer of 500 m and road type.
ϵ(ForCov_C + Rd_Typ + ForCov_C:Rd_Typ)	The extinction probability varies with interactive effects of coniferous forest type cover within a buffer of 500 m and road type.
ϵ(ForCov_F + Rd_Dist)	The extinction probability varies with additive effects of deciduous forest type cover within a buffer of 500 m and road distance.
ϵ(ForCov_M + Rd_Dist)	The extinction probability varies with additive effects of mixed forest type cover within a buffer of 500 m and road distance.
ϵ(ForCov_C + Rd_Dist)	The extinction probability varies with additive effects of coniferous forest type cover within a buffer of 500 m and road distance.
ϵ(ForCov_F + Rd_Dist + ForCov_F:Rd_Dist)	The extinction probability varies with interactive effects of deciduous forest type cover within buffer of 500 m and road distance.
ϵ(ForCov_M + Rd_Dist + ForCov_M:Rd_Dist)	The extinction probability varies with interactive effects of mixed forest type cover within a buffer of 500 m and road distance.
ϵ(ForCov_C + Rd_Dist + ForCov_C:Rd_Dist)	The extinction probability varies with interactive effects of coniferous forest type cover within a buffer of 500 m and road distance.
ϵ(Size + Rd_Dist)	The extinction probability varies with additive effects of pond size and road distance.
ϵ(Size + Rd_Dist + Size:Rd_Dist)	The extinction probability varies with interactive effects of pond size and road distance.

Table 2.6 Hypotheses and models related to the detection probability.

Models	**Hypotheses**
p(.)	The detection probability is constant.
p(Wat_Temp + Rain + Trafic)	The detection probability varies with additive effects of water temperature, rain, and car traffic.
p(SnowDay + SnowDay2 + year + year:SnowDay2)	The detection probability varies with interactive effects of quadratic SnowDay and year.

2.4 Results

The full data set consisted of 110 sites. During the first year of the study, 90 sites (45 sites/region) were sampled according to the selection criteria with 5 replicates for each combination of road distance and road type (Table 2.1). We added one site per category in 2013 for a total of 108 sites (54 sites/region). In 2014, we replaced 1 site in each region as some site owners did not want to grant access anymore.

Table 2.7 Range and mean of site covariates ; Size (ha), Road distance (m), Proportion of coniferous forest cover in a 500 m buffer (ForCov_C), Proportion of mixed forest cover in a 500 m buffer (ForCov_M), Proportion of deciduous forest cover in a 500 m buffer (ForCov_D), Road density (km/km^2) in a 500 m buffer, Proportion of wetland cover (WetCov) in a 500 m buffer, pond pH and water Conductivity, based on total of 110 sites.

Parameter	Range	Mean
Size (ha)	[0.003 – 1.08]	0.1894
Road distance (m)	[1 – 248]	86.18
ForCov_M	[0 – 0.72]	0.2503
ForCov_C	[0 – 0.67]	0.1307
ForCov_D	[0 – 0.91]	0.2557
Road density (km/km^2)	[1.04 –3.95]	1.703
WetCov	[0 – 0.69]	0.1074
pH	[5.41 – 8.34]	7.064
Conductivity (µS/m)	[28.14 – 808.04]	163.91

2.4.1 Observations across years

For the Green Frog, 44% (40/90) of sites had at least one detection in 2012, whereas 42% (45/108) had at least one detection in 2013 and 47% (51/108) in 2014. For the Mink Frog, 4.44% of sites had at least one detection for 2012, 9.25% of sites had at least one detection in 2013, and 7.4% in 2014. For the Spring Peeper, 44.4% of sites had at least one detection in 2012, 50% in 2013, and 51.8% in 2014 (Figure 2.2). We excluded the Spring Peeper data of 2012 from the analysis because we missed the peak season of the Spring Peeper in that year.

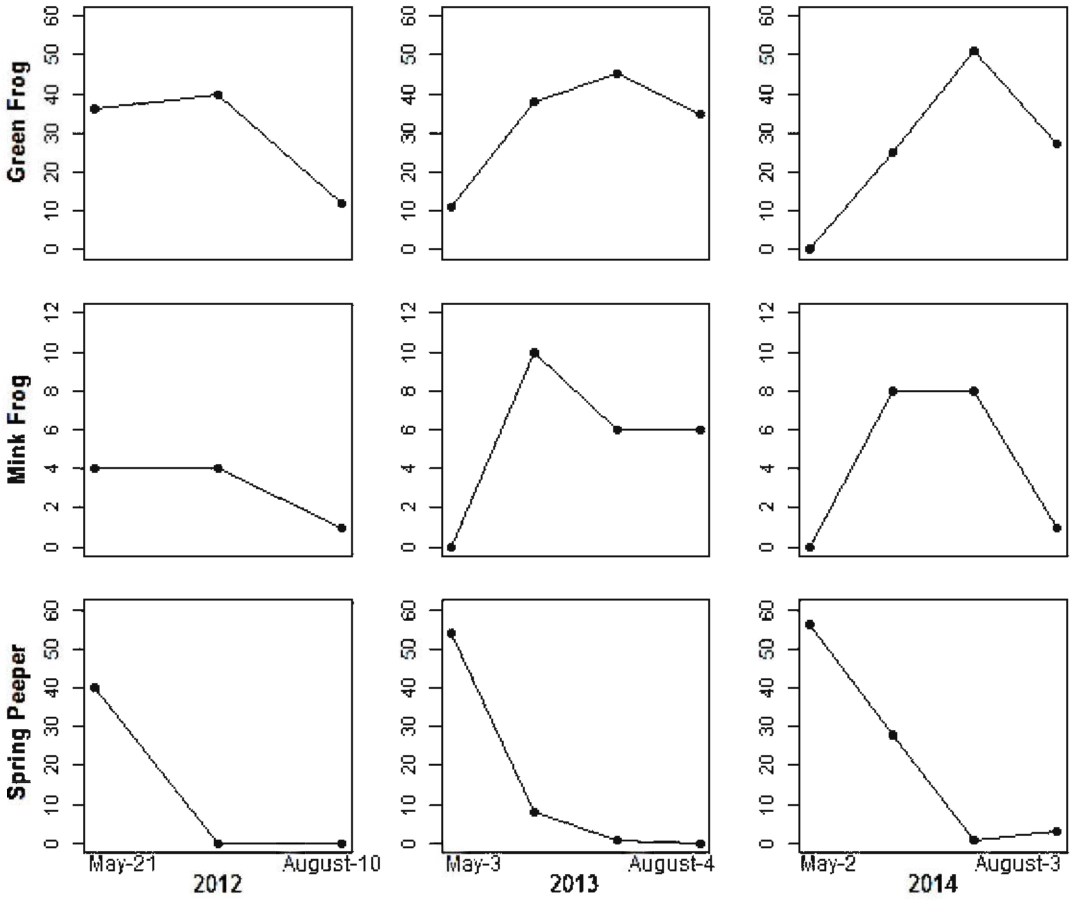

Figure 2.2 Observations of species by year with first and last sampling dates.

2.4.2 Green Frog

Two models had most of the support for the Green Frog, with a combined Akaike weight of 0.95 (Table 2.8). The first model ($\omega_i = 0.68$) considered additive effects of pH, conductivity, and peripheral vegetation on extinction probability and interactive effects of SnowDay and year on probability of detection, while there were no variables on either initial occupancy probability or colonization probability. The second model ($\omega_i = 0.27$, Δ AIC$_c$=1.87) included interactive effects of coniferous forest type cover within a buffer of 500 m and road type on the extinction probability and interactive effects of SnowDay and year on probability of detection, with no variables on initial occupancy probability and colonization probability.

Table 2.8 Top two dynamic occupancy models based on Akaike information criterion for small samples (AICc), showing the distance between each model (ΔAICc), the number of estimated parameters (K), and Akaike weights (ω_i) on the Green Frog data from sites in the vicinity of Aiguebelle and Plaisance National Parks, Québec, Canada (2012 - 2014).

Models	K	AICc	ΔAICc	ω_i
ψ (.) γ (.) ϵ (pH+Conductivity+Per_Veg) p(SnowDay+SnowDay^2+year+year:SnowDay^2)	13	1022.03	0.00	0.68
ψ(.)γ (.)ϵ(ForCov_C+Rd_Dist+ForCov_C:Rd_Dist) p(SnowDay+SnowDay^2+year+year:SnowDay^2)	13	1023.90	1.87	0.27

Results from model-averaged parameter estimates (Table 2.9) show that the extinction probability decreased with pH (95% CI: -4.47, -0.6, Figure 2.3). Extinction probability also increased with the proportion of coniferous forest cover in a 500 m buffer (95% CI: 0.31, 1.16, Figure 2.4). Finally, Green Frog detection probability increased with a quadratic effect of SnowDay (95% CI: 0.34, 0.85, Figure 2.5). The year 2014 have been different from 2013, maybe because the year 2014 had a lot more of precipitation and a colder winter temperature.

Table 2.9 Model-averaged parameter estimates for initial occupancy probability (ψ), colonization probability (γ), extinction probability (ϵ) and detection probability (p) on the Green Frog from sites in the vicinity of Aiguebelle and Plaisance National Parks, Québec, Canada (2012 - 2014). A 95% unconditional confidence interval excluding 0 indicates that the variable has an effect on a parameter.

Parameter	Estimate	SE	Lower 95% CI	Upper 95% CI
Extinction probability (ϵ)				
pH	-2.53	0.99	**-4.47**	**-0.6**
Conductivity	-1.32	1.62	-4.49	1.85
Per_Veg	0.4	0.39	-0.36	1.16
ForCov_C	1.56	0.63	**0.31**	**2.8**
Rd_Dist	0.73	0.95	-1.13	2.59
ForCov_C: Rd_Dist	1.33	0.82	-0.28	2.95
Detection probability (p)				
SnowDay	0.59	0.13	**0.34**	**0.85**
SnowDay2	-0.72	0.13	**-0.97**	**-0.46**
SnowDay2:Year2013	-0.16	0.24	-0.63	0.3
SnowDay2:Year2014	-0.91	0.34	**-1.56**	**-0.25**

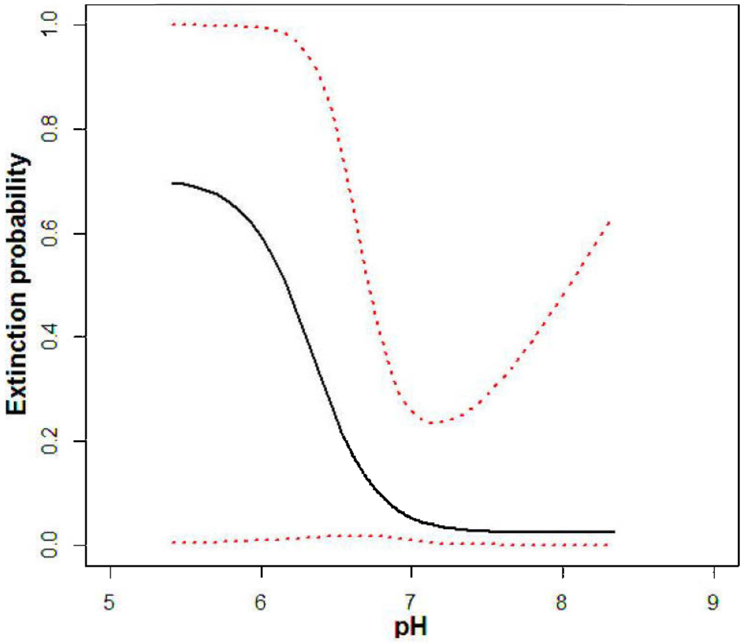

Figure 2.3 Green Frog extinction probability decreasing with pH. Dotted lines denote 95% CI resulting from model-averaged predictions.

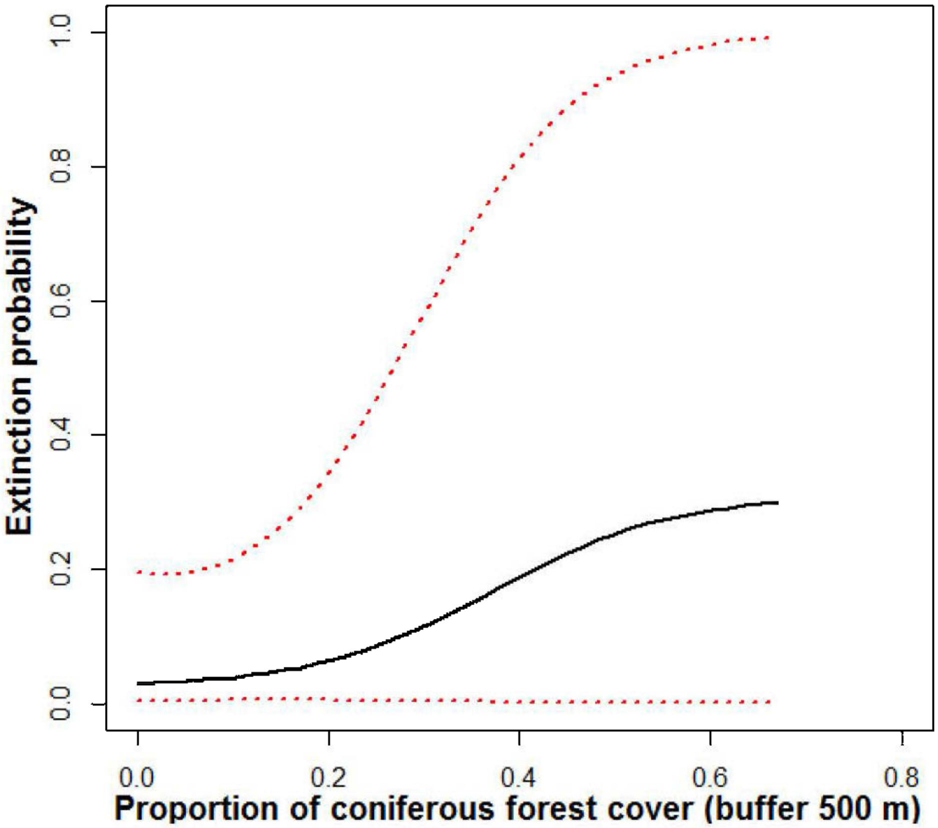

Figure 2.4 Green Frog extinction probability increasing with proportion of coniferous forest cover within buffer of 500 m. Dotted lines denote 95% CI resulting from model-averaged predictions.

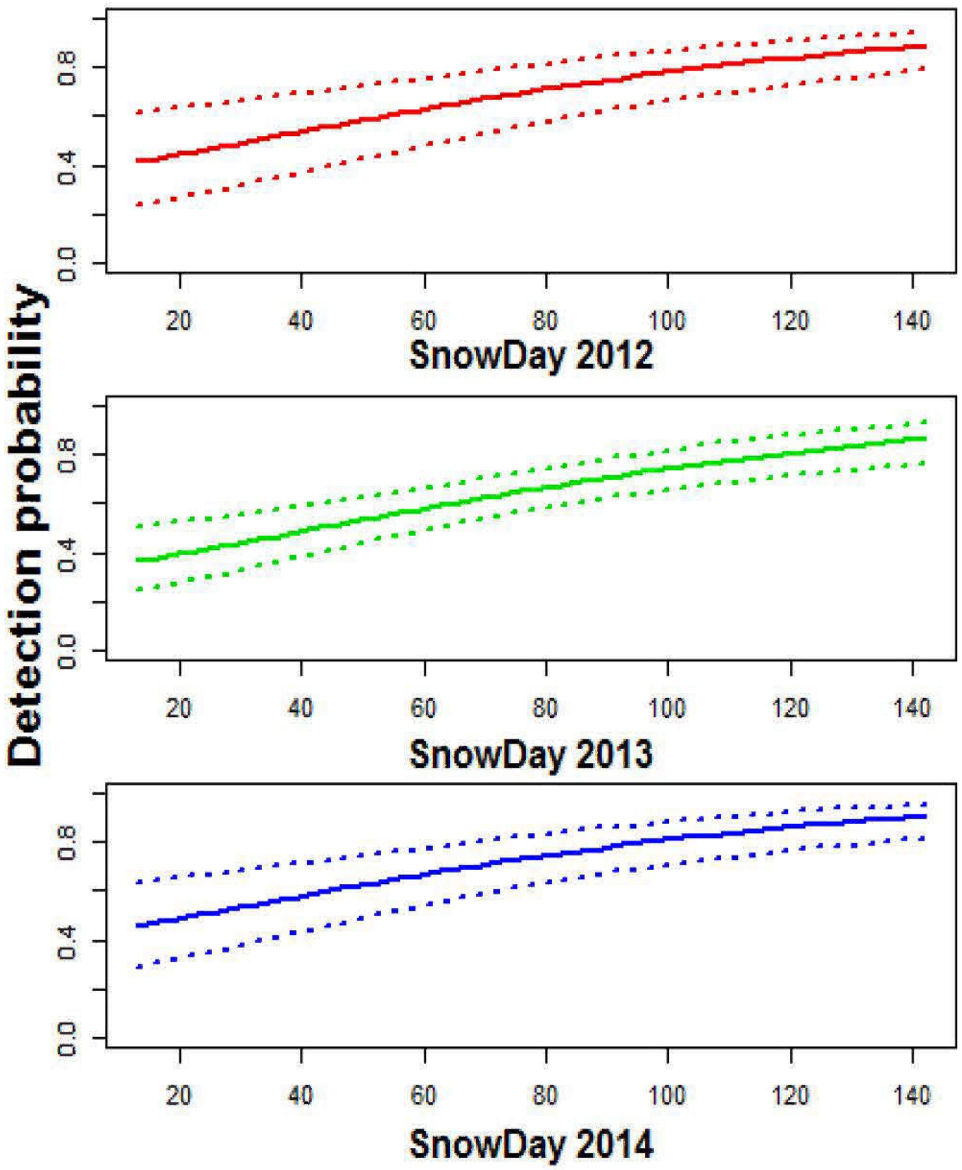

Figure 2.5 Green Frog detection probability increasing with SnowDay. Dotted lines denote 95% CI resulting from model-averaged predictions.

2.4.3 Mink Frog

Six models had most of the support for the Mink Frog, with a combined Akaike weight of 0.91 (Table 2.10). The first model ($\omega_i = 0.27$) considered interactive effects of pond size and road distance on extinction probability, interactive effects of SnowDay and year on detection probability, while there were no variables on probabilities of initial occupancy and colonization. The second model ($\omega_i = 0.25$, ΔAICc = 0.10) included interactive effects of pond size and road distance on colonization probability, interactive effects of SnowDay and year on detection probability, while there were no effects of variables on initial occupancy probability and extinction probability. The third model ($\omega_i = 0.14$, ΔAICc = 1.28) considered additive effects of pond size and road distance on colonization probability, interactive effects of SnowDay and year on detection probability, without any effects on initial occupancy probability and extinction probability. The fourth model ($\omega_i = 0.09$, ΔAICc = 2.09) considered additive effects of road distance and the proportion of mixed forest cover within a buffer of 500 m on colonization probability, interactive effects of SnowDay and year on probability of detection, but did not include variables on either initial occupancy probability or extinction probability. The fifth model ($\omega_i = 0.09$, ΔAICc = 2.24) considered interactive effects of road distance and the proportion of mixed forest cover within a buffer of 500 m on colonization probability, interactive effects of SnowDay and year on detection probability, but no effect on initial occupancy probability and extinction probability. Finally, the sixth model ($\omega_i = 0.07$, ΔAICc = 2.66) considered additive effects of road distance and the proportion of mixed forest cover within a buffer of 500 m on initial occupancy probability, interactive effects of SnowDay and year on probability of detection, while there were no effects of variables on colonization probability and extinction probability.

Table 2.10 Top six dynamic occupancy models based on Akaike information criterion(AICc), showing the distance between each model (ΔAICc), the number of estimated parameters (K), and Akaike weights(ω_i) on the Mink Frog data from the from sites in the vicinity of Aiguebelle and Plaisance National Parks, Québec, Canada (2012 - 2014).

Models	K	AICc	ΔAICc	ω_i
ψ (.) γ (.) ϵ (Size + Rd_Dist + Size:Rd_Dist) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	13	369.93	0.00	0.27
ψ (.) γ (Size +Rd_Dist + Size:Rd_Dist) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	13	370.04	0.10	0.25
ψ (.) γ(Size +Rd_Dist) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	12	371.22	1.28	0.14
ψ (.) γ(ForCov_M +Rd_Dist) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	12	372.03	2.09	0.09
ψ(.) γ(ForCov_M +Rd_Dist + ForCov_M:Rd_Dist) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	13	372.17	2.24	0.09
ψ(Size + Rd_Dist) γ(.) ϵ (.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	12	372.60	2.66	0.07

Multimodel inference indicated that Mink Frog colonization probability increased with pond size (95% CI: 0.09, 2.24, Figure 2.6, Table 2.11). We found no other variable influencing probabilities of initial occupancy, colonization, extinction, or detection (Table 2.11).

Table 2.11 Model-averaged parameter estimates for initial occupancy probability (ψ), colonization probability (γ), extinction probability (ϵ), and detection probability (p) on the Mink Frog data from sites in the vicinity of Aiguebelle and Plaisance National Parks, Québec, Canada (2012 - 2014). A 95% unconditional confidence interval excluding 0 indicates that the variable has an effect on a parameter.

Parameter	Estimate	SE	Lower 95% CI	Upper 95% CI
Occupancy (ψ)				
Size	3.98	2.97	-1.85	9.81
Rd_Dist	1.33	1.63	-1.86	4.53
Size: Rd_Dist	-	-	-	-
Colonization (γ)				
Size	1.17	0.55	**0.09**	**2.24**
Rd_Dist	-0.06	1.56	-3.11	2.99
Size:Rd_Dist	-1.5	1.3	-4.05	1.04
ForCov_M	-7.76	7.13	-21.72	6.21
Rd_Dist	0.03	1.15	-2.21	2.28
ForCov_M:Rd_Dist	8.91	10.73	-12.13	29.95
Extinction(ϵ)				
Size	-	-	-	-
Rd_Dist	1.72	3.3	-4.76	8.19
Size:Rd_Dist	5.46	6.83	-7.93	18.85
Detection (p)				
SnowDay	-0.13	0.33	-0.78	0.51
SnowDay2	-0.28	0.35	-0.97	0.41
SnowDay:Year2013	-1.07	0.67	-2.37	0.24
SnowDay:Year2014	-1.49	0.84	-3.14	0.15

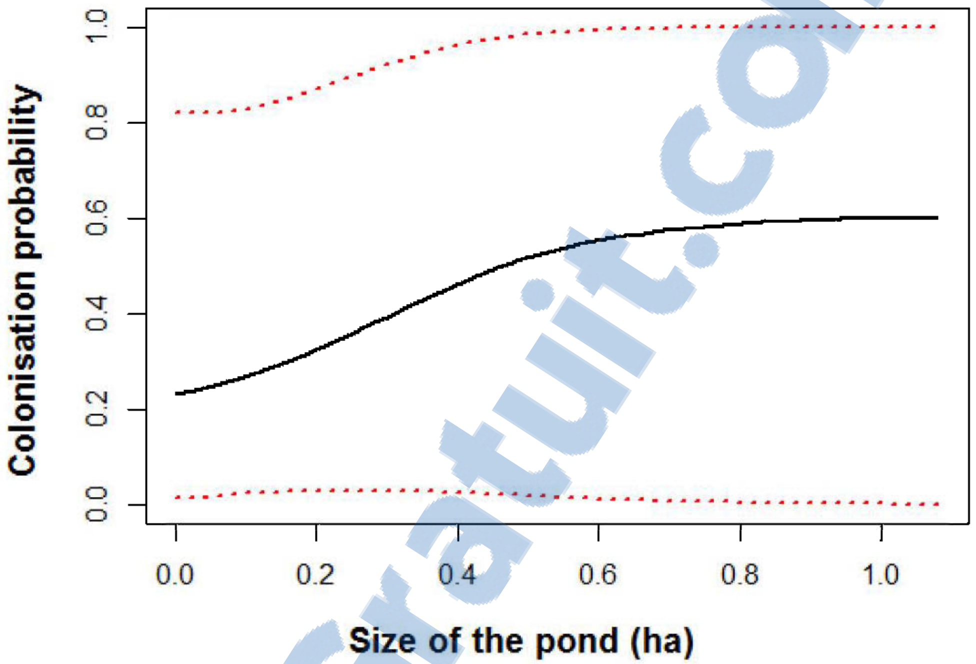

Figure 2.6 Mink Frog colonization probability increasing with pond size. Dotted lines denote 95% CI resulting from model-averaged predictions.

2.4.4 Spring Peeper

Five models had most of the support for the Spring Peeper, with a combined Akaike weight of 0.98 (Table 2.12). The first model ($\omega_i = 0.42$) considered interactive effects of road distance and proportion of mixed forest cover within a buffer of 500 m on initial occupancy probability, interactive effects of SnowDays and year on probability of detection, with no effect of variables on colonization probability and extinction probability. The second model ($\omega_i = 0.25$, ΔAICc = 1.00) considered interactive effects of road distance and proportion of deciduous forest cover within a buffer 500 m on initial occupancy probability, interactive effects of SnowDay and year on probability of detection, but no variables on colonization probability and extinction probability. The third model ($\omega_i = 0.13$, ΔAICc = 2.29) considered additive effects of road density and wetland cover within a buffer of 500 m on extinction probability, interactive effects of SnowDay and year on probability on detection, without any variable on initial occupancy probability and colonization probability. The fourth model ($\omega_i = 0.13$, ΔAICc = 2.31) included additive effects of road density and wetland cover within a buffer of 500 m on initial occupancy probability, interactive effects of SnowDay and year on probability on detection, but did not include variables on colonization and extinction probabilities. The fifth model ($\omega_i = 0.05$, ΔAICc = 4.32) contained additive effects of pH, conductivity, and peripheral vegetation on initial occupancy probability, interactive effects of SnowDay and year on probability of detection, without variables on colonization probability and extinction probability.

Table 2.12 Top five dynamic occupancy models based on Akaike information criterion (AICc), showing the distance between each model (ΔAICc), the number of estimated parameters (K), and Akaike weights(ω_i) on the Spring Peeper data from sites in the vicinity of Aiguebelle and Plaisance National Parks, Québec, Canada (2013 - 2014).

Models	**K**	**AICc**	**ΔAICc**	**ω_i**
ψ(ForCov_M +Rd_Dist + ForCov_M:Rd_Dist) γ(.) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	11	494.96	0.00	0.42
ψ(ForCov_F +Rd_Dist + ForCov_F:Rd_Dist) γ(.) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	11	495.96	1.00	0.25
ψ(.) γ(.) ϵ(Landscape) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	10	497.25	2.29	0.13
ψ(Landscape) γ(.) ϵ (.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	10	497.27	2.31	0.13
ψ(pH+Conductivity+Per_Veg) γ(.) ϵ(.) p(SnowDay+SnowDay^2 + year + year:SnowDay^2)	11	499.28	4.32	0.05

Model averaging indicated that occupancy probability varied with interactions between mixed forest cover within a buffer of 500 m and road distance (95% CI: 0.11, 1.78, Table 2.13). For a road distance of 50 m, the occupancy probability decreased slowly with mixed forest cover (Figure 2.7). For 100 and 150 m distances from road, the occupancy probability increased with mixed forest cover. Occupancy probability also varied with interactions between deciduous forest cover within a buffer of 500 m and road distance (95% CI: -2.5,-0.1, Figure 2.8). For a road distance of 50 m, the occupancy probability increased slowly with deciduous forest cover, whereas at 100 and 150 m from road, the occupancy probability decreased with deciduous forest cover. Finally, Spring Peeper detection probability decreased with SnowDay (95% CI: -3.32, -1.66, Figure 2.9).

Table 2.13 Model-averaged parameter estimates for occupancy probability (ψ), colonization probability (γ), extinction probability (ϵ), and detection probability (p) on the Spring Peeper data from sites in the vicinity of Aiguebelle and Plaisance National Parks, Québec, Canada (2013 - 2014). A 95% unconditional confidence interval excluding 0 indicates that the variable has an effect on a parameter.

Parameter	Estimate	SE	Lower 95% CI	Upper 95% CI
Occupancy (ψ)				
ForCov_M	0.44	1.55	-2.6	3.49
Rd_Dist	0.43	0.55	-0.66	1.51
ForCov_M:Rd_Dist	0.94	0.43	**0.11**	**1.78**
ForCov_F	-0.18	0.3	-0.77	0.42
Rd_Dist	0.22	0.44	-0.64	1.09
ForCov_F:Rd_Dist	-1.3	0.61	**-2.5**	**-0.1**
Rd_Density	0.2	0.34	-0.47	0.87
WetCov	0.61	0.56	-0.49	1.71
pH	-0.05	0.33	-0.7	0.6
Conductivity	-0.24	0.39	-0.99	0.52
Per_Veg	0.36	0.24	-0.11	0.84
Extinction (ϵ)				
Rd_Density	-0.33	0.79	-1.89	1.22
WetCov	-0.92	0.96	-2.81	0.97
Detection (p)				
SnowDay	-2.49	0.42	**-3.32**	**-1.66**
SnowDay2	0.53	0.5	-0.44	1.5
SnowDay2:Year2014	-0.79	0.51	-1.8	0.21

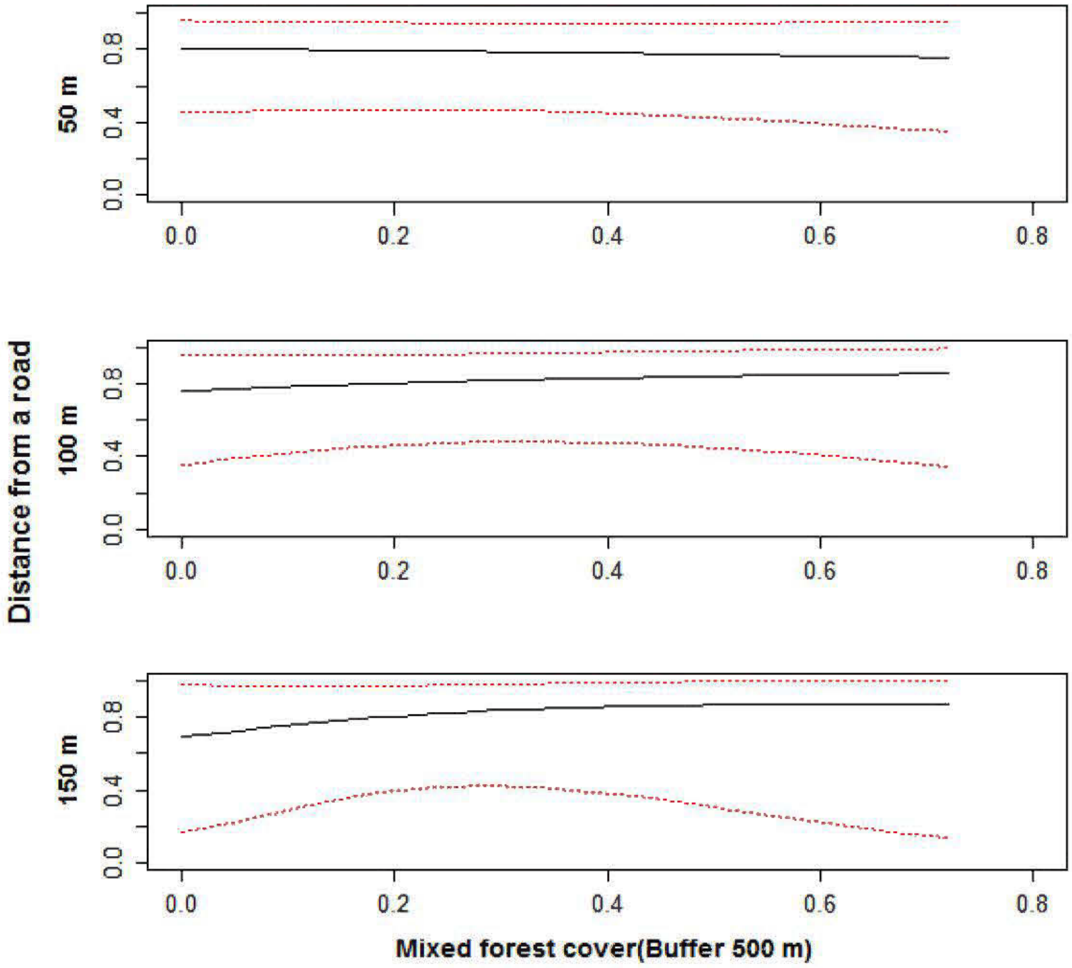

Figure 2.7 Spring Peeper occupancy probability with interactions between mixed forest cover within a buffer of 500 m x Road distance. Dotted lines denote 95% CI resulting from model-averaged predictions.

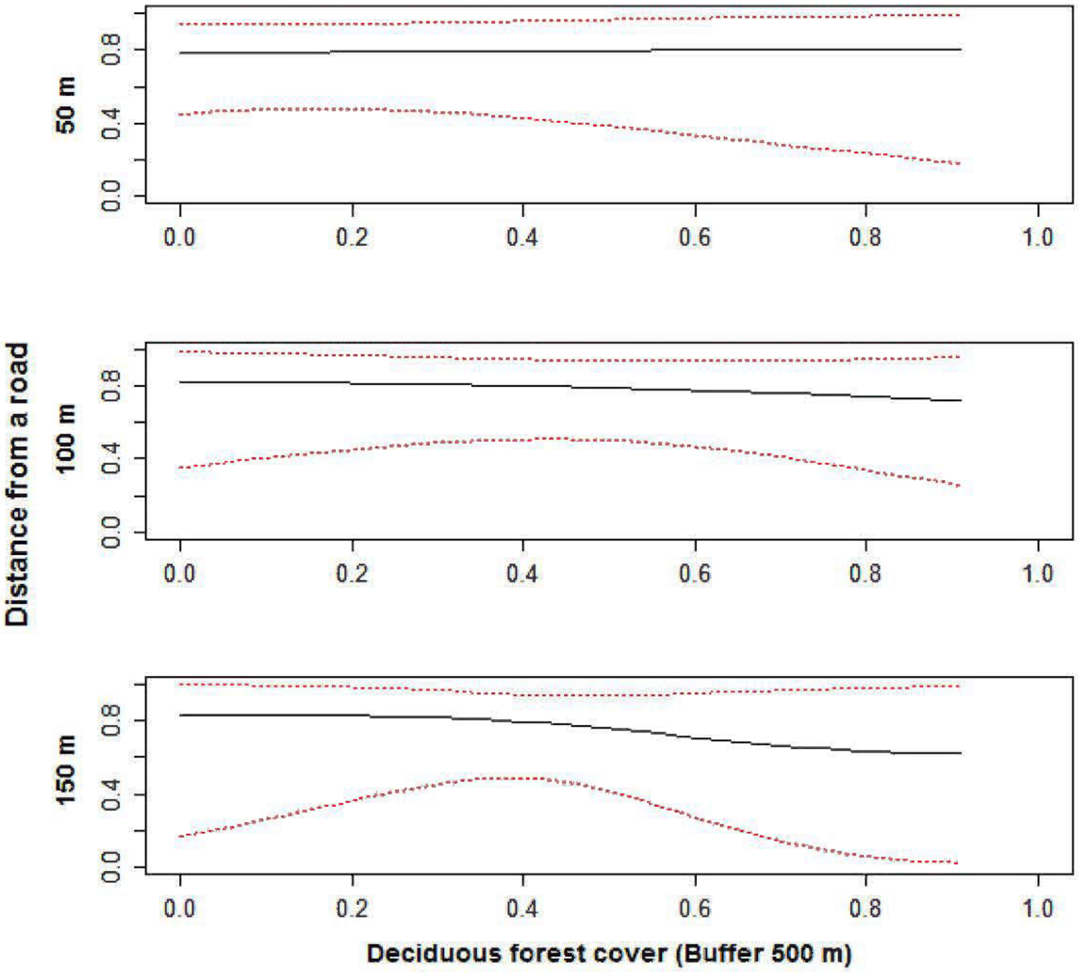

Figure 2.8 Spring Peeper occupancy probability with interactions between deciduous forest cover within buffer of 500 m x Road distance. Dotted lines denote 95% CI resulting from model-averaged predictions.

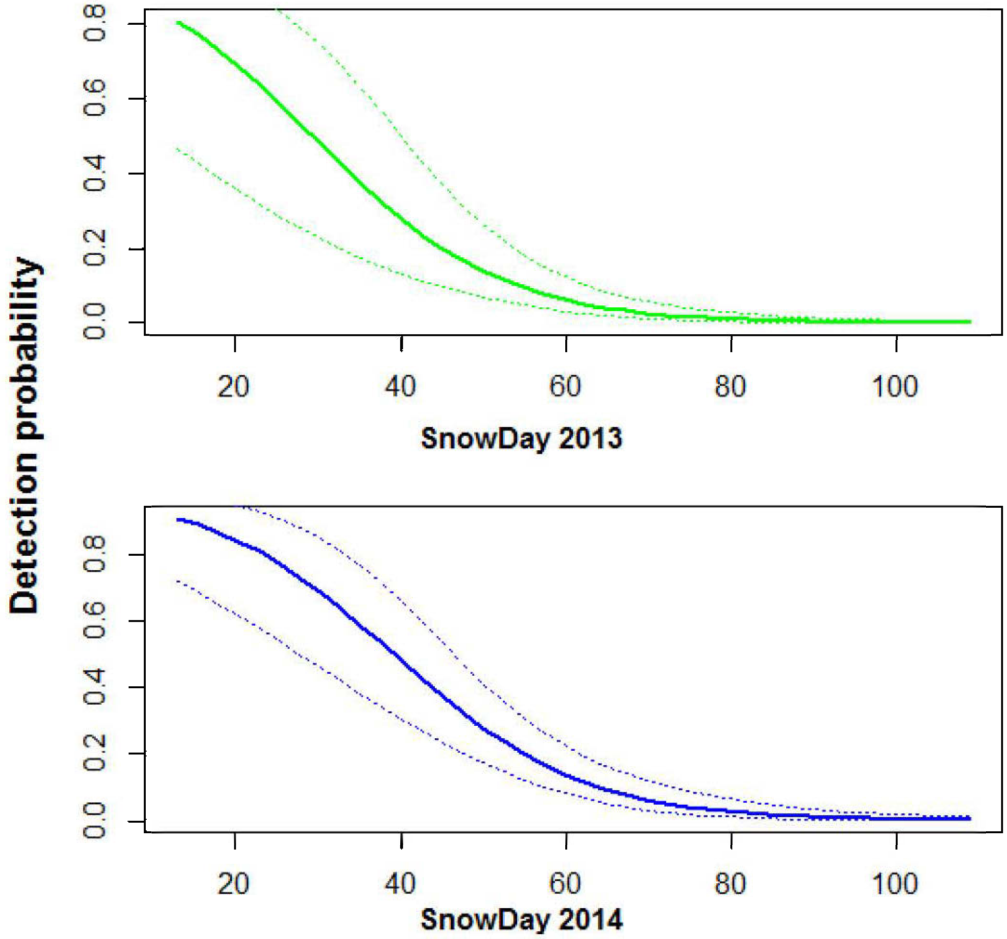

Figure 2.9 Spring Peeper detection probability decreasing with SnowDay. Dotted lines denote 95% CI resulting from model-averaged predictions.

2.5 Discussion

Our analysis suggests that the impact of roads varies with the species of anuran. We found that Green Frogs extinction parameter vary with wetland pH and the proportion of coniferous forest cover within a buffer of 500 m. The probability of extinction for that species increased with decreasing pH and also increased with coniferous forest cover within a buffer of 500 m. Both could be related as some authors suggest that the pH of the soil and water seem to become lower in coniferous forest stands than in deciduous stands due to a difference in the nitrogen atmospheric accumulation rate and limitation in temperate forest ecosystems (Aber *et al.* 1989; Ste-Marie and Paré 1999). Also, it has been shown in the past that roads, because of the generated pollution from deicing salt, change the quality of the environment by runoff waters and that could affect amphibian development and generally reduce the survival of individuals (Sanzo and Hecnar 2006; Karraker *et al.* 2008; Reeves *et al.* 2008; Karraker and Gibbs 2011). Some studies suggest that Green Frog acts differently in these circumstances than others amphibians. In these studies, Collins and Russell (2009) found that Green Frog had an intermediate sensitivity to sodium chloride concentration in ponds and that they were rarely found where the salinity was > 0.59 ppt. Karraker (2007) showed that Green Frog larval growth was unaffected by road salt and concluded that adults were relatively tolerant to road salt and in fact, that they were less sensitive than other North American amphibians. She also found that below 1500 µS, there was no significant difference in the survival rate and that a difference was only detectable at the 3000 µS treatment (Karraker, 2007). She attributed the result to a possible capacity of developmental and phenotypic plasticity (Karraker, 2007). As the range of conductivity in our sites was included between [28.14 – 808.04 µS] – with a mean of 163.91 µS – (Table 2.7) that could explain why we have not been able to link an effect of the conductivity on Green Frog occurrence, as this species seems to be able to deal with higher concentrations.

Some recent studies tried to link physicals parameters, such as noise generated by cars on roads, instead of chemical effects of roads on amphibian populations. Cosentino *et al.* (2014) suggested that the more important effect of roads on Green Frog is the noise from car traffic that comes into competition with males while calling. This also has been a conclusion from a study conducted in Eastern Ontario recently. Vargas-Salinas *et al.* (2014) found that Green Frog could change its calling behavior depending of the intensity of car traffic noise– calling less often when traffic noise intensity was higher and calling more often when traffic noise intensity was lower – as the call frequencies of that species is low and about the same as the traffic frequencies. However, Cunnington and Fahrig (2013) suggest that because of the ability of Green Frogs to compensate for traffic noise, mate attraction is not affected by traffic noise generated by roads near ponds. Even if recent studies propose the possibility that Green Frog had developed a behaviorial, developmental, and phenotypic plasticity, our results suggest that pH of the wetlands seems to be the more limiting parameter for that species, but it was not possible to clearly link it to a direct effect of roads (Karraker 2007; Collins and Russell 2009; Vargas-Salinas *et al.* 2014).

Results for the Mink Frog show that pond size matters more than the other parameters we measured to explain the probability of colonization, as the latter increased with pond size. Mink Frogs are known as highly territorial and aquatic amphibians (Desroches and Rodrigue 2004; Elliott *et al.* 2009; AARQ [Online]; RÉCCAR [Online]). Some authors suggest that Mink Frogs could use rivers for juvenile dispersal (Desroches and Rodrigue 2004). Mortality on roads may be less important for that species, but this remains speculative. Seiler (2003) also suggests that territorial species and species that occur in small or isolated populations are more affected by the presence of roads. On the other hand, species with nocturnal movements may be less affected by roads than diurnal species (Hels and Buchwald

2001). Among all studied sites, the number of sites occupied by Mink Frog was the lowest. However, we found no effects of roads on the species.

Results for the Spring Peeper suggest that the effect of road distance on the occupancy probability depends on forest cover (road distance x forest cover) and also varies with the type of canopy. At a distance of 50 m from a road, the occupancy probability decreased with mixed forest cover, whereas for distances of 100 m and 150 m, the occupancy probability increased. In contrast, occupancy probability increased with the deciduous forest cover at 50 m from a road, whereas for distances of 100 m and 150 m, the occupancy probability decreased. Spring Peepers breed in temporary ponds and hibernate under leaves and logs in forests (Desroches and Rodrigue 2004; Elliott *et al.* 2009; AARQ [Online]; RÉCCAR [Online]). Roadsides favor the creation of ditches and channels in spring (Wemple *et al.* 1996; Forman and Alexander 1998). Spring Peepers may prefer that kind of site but may also have a preference for deciduous forest cover near the pond to fulfill the needs of adequate hibernation sites. It is also possible, as they are arboreal frogs, that it is easier for them to climb on deciduous rather than coniferous trees. At a distance of 50 m from a road, the occupancy probability increases for a deciduous forest cover, but decreases for a mixed forest cover. Similarly, for distances of 100 m and 150 m, occupancy probability decreased for deciduous forest cover, but increased for mixed forest cover. This result suggests that Spring Peeper select ponds close to a road except if that pond is surrounded by mixed forest because of the less important proportion of deciduous trees. The probability of occupancy will be greater for mixed forest far from the road (i.e. at distances of 100 m and 150 m). This could be due to a greater availability of hibernation sites increasing with the cover of deciduous trees. These results suggest that they could mostly optimize the distance between the pond for reproduction and the undergrowth for hibernation no matter if a road is close or not. As Ashley and Robinson (1996) proposed, some species of amphibians could be associated with roadside vegetation. It is possible that amphibians chose wetlands close to roads because of the woodland road section as both habitat characteristics are important (Mitchell 2000). Homyack *et al.* (2014) also suggested that roadside ditches could support occupancy of an important variety of amphibians in managed forest systems, across a range of years since disturbance but that the vehicle traffic would be limiting for most species because of the considerable mortality rate. On the other hand, Beebee (2013) suggested that arboreal species could remain a low mortality rate with an active behavior on roads. Finally, it is possible that because of it needs, Spring Peeper would prefer a roadside environment and like Green Frog they possibly adjusted their behavior to compensate the road activity.

The number of days elapsed since snow melt was the variable most often related to the probability of detection for two species out of three. For the Green Frog, the probability of detection, within the range observed, increased with SnowDay while it decreased for Spring Peeper. This result agrees with the biology of the species, as Green Frogs breed from June to July and sometime extend to August (Desroches et Rodrigue 2004; Elliott *et al.* 2009; RÉCCAR [En ligne]). Spring Peepers breed by the end of spring and into the end of June and sometimes July (Desroches et Rodrigue 2004; Elliott *et al.* 2009; RÉCCAR [En ligne]). In contrast, Mink Frogs breed in the middle of summer from June to August (Desroches et Rodrigue 2004; Elliott *et al.* 2009; RÉCCAR [En ligne]). The lack of relationship between the detection probability of Mink Frogs and the number of days since snow melt may be due to late-breeding habits of the species. Spring conditions may disturb the habits of early species with short breeding windows more so than late species. The sparseness of the Mink Frog data is also a potential explanation for the lack of response to the number of days since snow melt, as between 4.4% and 9.25% of sites had at least one detection.

Cosentino *et al.* (2014) suggest that there could be a small or positive effect of roads on anuran distribution of individual species and richness that were mostly controlled by road density and traffic. Other studies of road impacts on animals concluded that mortality was directly correlated with traffic volumes and speed limits, regardless of road type (Seiler 2003 ; Gunson *et al.* 2005; Rytwinski and Fahrig 2013). One problem remains in the definition of « high traffic volume » or « high speed limits » because these values vary among studies. Furthermore,in our study area, road density ranged between 1.04 to 3.95 km/km^2, with a mean of 1.703 km/km^2 (Table 2.7), whereas some authors report 2 to 40 km/km^2 (Rytwinski and Fahrig 2011). In their study, Rytwinski and Fahrig (2013) considered a road density of 1.86 km/km^2 as a low density, whereas this corresponds to our mean road density. Also, traffic volumes vary among countries and the areas where studies take place. Seiler (2003) suggest that traffic levels above 10,000 vehicles per day should be considered as a critical barrier for species. Considering that our busiest roads had a traffic volume about 166 cars/hour, even by extrapolation, we are far away from these numbers. If we compare with a study that focussed on small mammals and amphibians (i.e. Hels and Buchwald 2001), their models suggest that an average traffic volume of 4000 vehicles per day might kill most amphibians trying to cross a road but could present a marginal effect on animals having higher crossing speeds. Research on the effects of roads should always consider biology and behavior of species under studies as well as the landscape scale.

However, some studies showed that the impact of roads depends on species and gender. Puky (2006) suggests that because amphibian males and females spend different periods in different types of environments with a marked difference in the migration radius, the proportion of dead individuals on roads could be different. Other studies reported similar road effects on other taxa. For example, Benítez-López *et al.* (2010), in a meta-analysis of more than 600 articles, found that mammal and bird populations seem to avoid infrastructures from infrastructures such as roads, but that this displacement distance is linked to the habitat type and the species population. Birds and mammals seem to avoid larger distances from roads in open areas compared to woodland areas (Benítez-López *et a.l* 2010). They also concluded that animal size could play a role in the avoidance response as small-sized mammals were affected within few meters from infrastructures, whereas the abundance of large-sized mammals was reduced up to several hundred meters from infrastructures (McLellan and Shakleton 1989; Cameron *et al.,* 1992; Ortega and Capen 1999; Nellemann *et al.* 2003; Rytwinski and Fahrig 2013). Also, Benítez-López *et al.* (2010) showed that raptors were more abundant close to roads and other infrastructures than other bird taxa. Brown and Brown (2013) suggest that selection should also play a role on road avoidance and persistence of some populations of birds near roads. Seiler (2003) proposed that busy roads could simply repel animals, whereas quiet roads could be dangerously attractive. Rytwinski and Fahrig (2013) suggest that small-bodied species that avoid roads would be considered to be unaffected by increasing road density, in a mortality perspective at least.

The question of short-term persistence of populations near roads seems to be less critical as initially thought. But the new question is about how long populations would be able to persist and what would be the consequences on their genetics? Also, given that our study is based on roads that were created decades ago, it is possible that populations stabilized over time and are no longer linked by the exchange of individuals. However, Safner *et al.* (2011) reported that new roads could quickly reduce the movement between populations but that the genetic dissociation is a phenomenon acting over decades. Maintaining the genetic diversity of local populations and between populations remains primordial to ensure long term persistence of metapopulations. We agree with Rytwinski *et al.* (2015) when they suggest that road mitigation experiments must become a standard part of new road constructions because there is a lack of critical information on population response before and after new road projects.

2.6 Conclusion

Habitat degradation and fragmentation are considered as leading factors affecting the loss of biodiversity and amphibian declines by many authors (Knutson *et al.* 2001; Baillie 2004; Sanzo and Hecnar 2006; Fahrig and Rytwinski 2009). However, it is difficult to quantify that impact on wildlife as maybe some species find different ways to deal with roads. We found no impact of roads on colonization and extinction dynamics of wetland amphibians, but our results provide information for future experimentations (particularly on Green Frog and Spring Peeper populations). Our study spanned 3 years. Long term monitoring is primordial to assess the cumulative impacts of roads on animal populations as well as before and after construction studies to get a global idea of the situation. Some species of anurans are probably more resilient than we thought initially.

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CHAPITRE III

CONCLUSION GÉNÉRALE

Ce projet de recherche avait comme objectif de déterminer l’impact des routes sur la dynamique de colonisation et d’extinction des populations d’amphibiens en milieux humides.

Selon les différentes espèces à l’étude, nous avons obtenu les résultats suivants :

1. Pour la Grenouille verte; la probabilité d’extinction diminue avec le pH du milieu humide et augmente avec la proportion de résineux dans le couvert forestier. La probabilité de détection augmente avec le nombre de jours écoulés depuis la fonte des neiges et l’année 2013 diffère légèrement des années 2012 et 2014.

2. Pour la Grenouille du Nord; la probabilité de colonisation augmente avec la superficie du milieu humide.

3. Pour la Rainette crucifère; l'effet de la distance à la route sur la probabilité d’occurrence dépend du couvert forestier (interaction couvert x distance), mais diffère selon le type d’essence. Pour la distance de 50 m à une route, la probabilité d’occurrence diminue avec le couvert forestier de type mixte alors que pour des distances de 100 et 150 m, la probabilité d’occurrence augmente avec le couvert forestier de type mixte. Pour la distance de 50 m à une route, la probabilité d’occurrence augmente avec le couvert forestier de type feuillu alors que pour des distances de 100 et 150 m à une route, la probabilité d’occurrence diminue avec le couvert forestier de type feuillu. De plus, la probabilité de

détection de la Rainette crucifère diminue avec le nombre de jours écoulés depuis la fonte des neiges.

Aux échelles spatiales que nous avons considérées, nous n'avons trouvé aucun effet des routes sur les dynamiques de colonisations et d'extinctions pour les espèces étudiées. Concernant l'hypothèse sur la probabilité de détection, le nombre de jours écoulés depuis la fonte des neiges s'est avéré la meilleure variable pour deux espèces sur trois. Il est possible que le nombre de données de détection pour la Grenouille du Nord n'aie pas permis de lier le paramètre SnowDay à cette espèce.

Limites de l'étude

La détection de type présence/absence n'est pas un gage de la qualité ou de la quantité d'individus d'un milieu humide. Cela permet simplement de vérifier le nombre de sites occupés au cours des saisons, dans la perspective d'une meilleure compréhension de métapopulations d'une région. Par ailleurs, l'échantillonnage par station d'écoute ne permet pas de mettre en lumière la présence de femelles pour un site, puisque seuls les mâles chantent. Ainsi, un site occupé par un mâle solitaire d'une espèce donnée a le même poids qu'un autre occupé par une chorale d'individus sans tenir compte de la présence de femelles fécondes ni de la qualité des mâles présents.

Recommandations et pistes de recherches

Il est recommandé de poursuivre l'étude sur une plus longue période de temps afin de mieux mesurer les dynamiques aux échelles spatiales et temporelles puisque la présente étude s'est déroulée uniquement sur trois saisons de reproduction. Il est fort à parier que la poursuite de cette étude sur plusieurs années permettrait à la méthode des conclusions plus claires au fil des générations d'anoures, au moins en ce qui a trait à la dynamique de colonisation et d'extinction des amphibiens à l'étude.

La rainette crucifère semble, de par sa biologie et les résultats obtenus dans notre projet, une espèce pertinente à l'étude d'impact des routes sur les amphibiens. Il serait intéressant de réitérer l'étude en employant différentes zones tampons lors du choix des sites et au cours des analyses afin d'en raffiner les résultats.

Dans une prochaine étude, il serait pertinent de choisir à *priori* les sites en fonction de classes de couvert forestier et non de classes de distances à la route afin de démontrer plus précisément l'effet mitigé que peut induire le couvert forestier sur l'impact des routes. La Grenouille verte ainsi que la Rainette crucifère ont toutes deux démontré des effets selon le type de couvert forestier, laissant suggérer qu'une proportion plus élevée de conifères au sein du couvert forestier n'étaient pas souhaitées pour ces espèces. De plus, il est possible qu'on surestime le besoin en couvert forestier des espèces : un couvert forestier trop grand pourrait potentiellement s'avérer un facteur limitant la dispersion des espèces.

Par ailleurs, il pourrait également être intéressant de sélectionner des sites à *priori* en fonction de la quantité de route et quantité de milieux humides dans le paysage. Il est possible que nos étendues de variation concernant ces points n'étaient pas suffisamment large pour permettre de les faire ressortir dans les modèles. S'il avait été possible d'avoir certains sites n'ayant aucun milieu humide à proximité mais une très grande quantité de routes principales asphaltées il est possible que les résultats aient été nuancés.

Une étude plus large pourrait incorporer des échantillons génétiques de certains individus par capture-recapture afin de vérifier si les sites sont reliés entre eux par flux génétique. Il est possible que les populations des différents sites se soient différenciés ou non au fil du temps. Comme notre étude se base sur des routes ayant été créées il y a plusieurs décennies, il est envisageable que les populations se soient stabilisées avec le temps et ne soient plus liées entre elles par l'échange d'individus. La manière d'aborder la question de la préservation des étangs serait alors moins portée à être établie à l'échelle du paysage et deviendrait un peu plus complexe.

Le fait de ne pas avoir trouvé d'impact quantitatif des routes sur les amphibiens à l'étude ne veut pas nécessairement dire que l'effet est inexistant. Il est possible que certaines espèces aient finalement des flexibilités ou des capacités d'adaptation plus grandes que ce dont nous aurions pu croire au départ face aux changements. De récentes études s'attardent à des aspects originaux concernant la réaction des amphibiens aux routes et nous croyons que plus il y aura d'études sur des aspects variés et plus nous serons à même de comprendre la complexité de ces animaux.

ANNEXE A - CARACTÉRISTIQUES DES ANOURES DU QUÉBEC [1]

Famille	**Bufonidés**	**Hylidés**			
Espèce	Crapaud d'Amérique	Rainette versicolore	Rainette crucifère	Rainette faux-grillon de l'ouest	Rainette faux-grillon boréale
Longueur Mâle adulte (cm)	4.0 à 8.5	3.2 à 5.1	1.8 à 3.3	1.9 à 3.2	1.9 à 3.5
Longueur Femelle adulte (cm)	5.1 à 11.4	3.3 à 6.0	2.0 à 3.7	2.0 à 3.7	1.9 à 3.9
Longueur Têtard (cm)	1.0 à 2.8	3.0 à 5.0	1.0 à 2.0	1.0 à 2.3	1.0 à 2.3
Longueur métamorphe (cm)	0.7 à 1.2	1.3 à 2.0	0.9 à 1.4	0.7 à 1.1	0.7 à 1.3
Date de reproduction et ponte	Fin avril et mai (juin dans le nord)	Mai et juin	Mi-avril à fin juin	Mi-avril à mi-mai	Mai et juin
Nombre d'œufs	2000 à 20 000	400 à 2000 (paquets de 10 à 40)	200 à 1600 (paquets de 1 à 40)	150 à 800 (paquets de 10 à 75)	Probablement 150 à 800
Durée du stade larvaire (mois)	2 à 3	1.5 à 4	1.5 à 3.5	1 à 2	1 à 2
Maturité sexuelle (ans)	2 à 3	2 à 3	2	1 à 2	1 à 2
Longévité (ans)	4	Inconnue	2 à 4	2 à 4	2 à 4

1-Tiré de Desroches et Rodrigue (2004).

(ANNEXE A - SUITE)

Famille	**Ranidés**					
Espèce	<u>Grenouille des bois</u>	<u>Grenouille Léopard</u>	<u>Grenouille des marais</u>	<u>Grenouille verte</u>	<u>Grenouille du nord</u>	<u>Oua-ouaron</u>
Longueur Mâle adulte (cm)	3.4 à 6.5	5.1 à 8.0	4.2 à 6.4	5.2 à 9.8	4.6 à 7.1	8.5 à 18.0
Longueur Femelle adulte (cm)	3.4 à 8.3	6.0 à 11.1	4.8 à 8.7	5.8 à 10.8	4.8 à 7.6	9.4 à 20.3
Longueur Têtard (cm)	1 à 5	3.0 à 8.4	3.0 à 8.0	3.0 à 9.2	3.0 à 9.9	4.0 à 17.1
Longueur méta-morphe (cm)	1.4 à 1.8	1.8 à 3.1	1.9 à 2.7	2.3 à 3.8	2.9 à 4.0	3.1 à 5.9
Date de repro-duction et ponte	Avril et mai	Mi-avril à fin mai (juin dans le nord)	Mai et juin	Juin à août	Juin à août	Juin à août
Nombre d'œufs	600 à 3000	900 à 5200	780 à 3000	1400 à 5300	500 à 1700	3800 à 23500
Durée du stade larvaire (mois)	1 à 4	2 à 4	2 à 4	12 à 15	12 à 15	24 à 36
Maturité sexuelle (ans)	1 à 2 (4 à 6 dans la région de la Baie James)	2	2	2 à 3	2 à 3	3 à 5
Longévité (ans)	2 à 5 (jusqu'à 10 dans la Baie James)	4 à 5	4	inconnue	inconnue	8 à 9

ANNEXE B - CARACTÉRISTIQUES ENVIRONNEMENTALES 2012 À 2014[1]

			Région du Parc national de Plaisance [2] (Outaouais)	**Région du Parc national d'Aiguebelle [3] (Abitibi-Témiscamingue)**
Saison 2012	**Novembre 2011 à Avril 2012**	Précipitations sous forme de neige (cm)	152	264.4
	Novembre 2011 à Octobre 2012	Précipitations sous forme de pluie (mm)	844	855.6
	Novembre 2011 à Avril 2012	Température moyenne Hiver	-1.58	-3.63
	Avril 2011 à Octobre 2012	Température moyenne Été	15.6	10.95
Saison 2013	**Novembre 2012 à Avril 2013**	Précipitations sous forme de neige (cm)	184.4	354
	Novembre 2012 à Octobre 2013	Précipitations sous forme de pluie (mm)	1009.4	798.6
	Novembre 2012 à Avril 2013	Température moyenne Hiver	-3.07	-4.52
	Avril 2012 à Octobre 2013	Température moyenne Été	15.28	12.85
Saison 2014	**Novembre 2013 à Avril 2014**	Précipitations sous forme de neige (cm)	200.5	372.6
	Novembre 2013 à Octobre 2014	Précipitations sous forme de pluie (mm)	1103.3	790.4
	Novembre 2013 à Avril 2014	Température moyenne Hiver	-6.07	-10.4
	Avril 2013 à Octobre 2014	Température moyenne Été	15.47	12.55

1- Tiré de Environment Canada climate archives (Government of Canada [Online]
2- Ville de référence : Montebello
3- Ville de référence : Mont-Brun

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