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DANS LE SUD DU QUÉBEC

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DE LA MAÎTRISE EN BIOLOGIE

PAR
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AVANT-PROPOS

Deux articles scientifiques en anglais composent le corps principal de ce mémoire. Le premier concerne l'identification de populations de bernaches du Canada à l'aide de la morphométrie et des analyses isotopiques. Le second illustre le rôle de la chasse comme outil de contrôle de la population du sud du Québec. Les deux articles seront soumis à des revues scientifiques. Je suis premier auteur et mon directeur Jean-François Giroux ainsi que Jean Rodrigue du Service Canadien de la Faune sont co-auteurs pour les deux articles. Richard Cotter, Jack Hughes et Ted Nichols sont également co-auteurs sur l'article concernant l'identification de populations. Les données du premier article ont été récoltées lors des opérations de baguage de bernaches du Canada effectuées par le Service Canadien de la Faune dans le nord et le sud du Québec entre 1996 et 2004. J'ai principalement récolté les données pour le second article durant l'été et l'automne 2004. Une introduction générale entame le mémoire et porte sur les réalités modernes et l'implication de la chasse en gestion des populations animales de même que sur la bernache du Canada dans la voie migratoire de l'Atlantique. Une conclusion générale illustre les points principaux relatés dans les deux articles et souligne les implications futures associées à mon mémoire. Un appendice relatif à des données non-publiées portant sur les signatures isotopiques de différents sites d'échantillonnage est présenté en toute fin de mémoire.

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RÉSUMÉ

Les systèmes écologiques se modifient sous l'influence des activités anthropiques. Alors que certaines espèces sauvages sont défavorisées par ces modifications, d'autres en tirent avantage et s'adaptent avec succès. Au cours des dernières décennies, de nombreuses populations d'oies ont vu leurs effectifs augmenter de façon exponentielle suite, entre autre, à des changements dans l'utilisation du territoire agricole. La bernache du Canada résidente (*Branta canadensis maxima*) s'est particulièrement bien adaptée aux habitats anthropiques après sa réintroduction dans le nord-est des États-Unis. Considérant les conflits causés par la surabondance de cette sous-espèce de bernache dans plusieurs états et provinces, son expansion préoccupe les gestionnaires d'ici. Les activités cynégétiques étant traditionnellement utilisées dans la gestion des populations de sauvagine, l'implication de la chasse sur la population nicheuse nouvellement établie dans le sud du Québec est le sujet principal de ce mémoire. Le premier chapitre a pour objectif le développement d'un outil de caractérisation des populations présentes dans la voie migratoire de l'Atlantique afin d'estimer de façon plus précise leurs paramètres démographiques. Deux sous-espèces sont principalement représentées (*B. c. interior* et *B. c. maxima*), lesquelles sont distinctes morphologiquement. De plus, les populations existent et muent sous différentes latitudes. En utilisant la morphométrie et les signatures isotopiques des plumes primaires ($\delta^{13}\text{C}$, $\delta^{15}\text{N}$), il est possible d'établir la population source des individus. Des modèles ont d'abord été développés afin de caractériser les oiseaux lors des opérations de baguage permettant ainsi une meilleure estimation des taux de survie de ces populations. Nos résultats permettent également de classer les oiseaux récoltés durant la chasse d'automne. Alors qu'il subsiste une hétérogénéité de populations pendant la migration automnale, l'estimation de la contribution de chacune d'elles dans la récolte améliorera l'estimation des taux de mortalité due à la chasse. Le deuxième chapitre vise à décrire la dispersion des bernaches à l'automne afin d'évaluer le rôle potentiel de la chasse comme outil de contrôle de la croissance de la population de bernaches résidentes qui nichent dans la région péri-urbaine de Montréal. En utilisant des méthodes de capture-marquage-recapture, nous avons documenté leur distribution post-reproduction et démontré que les individus se déplacent dans des secteurs où la décharge d'arme à feu est interdite et cela avant même l'ouverture de la saison de chasse. Les individus quittent les sites de reproduction et d'élevage situés dans la région de Varennes et de Repentigny, et se réfugient à Laval et Terrebonne. Nos résultats montrent qu'après la dispersion, les milieux naturels demeurent l'habitat principalement utilisé par les bernaches et que ces dernières résident sur le territoire jusque tard en automne. La gestion actuelle du territoire limite considérablement l'utilisation de la chasse sportive traditionnelle comme moyen de contrôle de la population résidente du sud du Québec. Des stratégies alternatives telles que la chasse contrôlée limitée dans le temps et l'espace devront être considérées. Par contre, l'implantation de telles mesures nécessiteront une sensibilisation du public à la problématique et l'établissement de politiques visant à optimiser la valeur globale de cette espèce faunique.

Mots clés: Aménagement de la faune, bernache du Canada, *Branta canadensis*, chasse, isotopes stables, morphométrie, péri-urbain, refuge, voie migratoire de l'Atlantique

INTRODUCTION GÉNÉRALE

L'expansion de la population humaine change les habitats, altère les fonctions écologiques et réduit la biodiversité. Néanmoins, l'étalement urbain favorise de nombreuses espèces animales qui survivent et se reproduisent dans ces environnements en tirant avantage de la disponibilité des ressources qu'on y retrouve (Ankney 1996; Boal et Mannan 1998; Conover et Conover 2003; DeStefano et DeGraaf 2003; Marzluff *et al.* 2001; Ticer 1998). Selon la répartition et la taille qu'ont ces populations, elles peuvent profondément affecter l'intégrité des écosystèmes (Jano *et al.* 1998; Tafangenyasha 2001) en plus d'imposer des coûts substantiels aux sociétés humaines (Filion 1998; Patterson 1991; Wagner *et al.* 1997; Yodzis 2001). Conover (1997) estime que le coût annuel des dommages causés par la faune dans les 100 plus grandes villes américaines atteint près de 4 milliards de dollars. D'un autre côté, nos sociétés retirent des bénéfices associés à la présence de la faune. Par exemple, les ornithologues et les chasseurs contribuent au développement économique et plus particulièrement autour des régions urbanisées (Atlantic Flyway Council 1999; Service canadien de la faune 2005).

La présence d'une espèce faunique n'a pas que des conséquences *socio-économiques* ou *récréatives*. Citons, entre autres, la valeur *écologique* qui est le bénéfice que l'espèce apporte au fonctionnement de l'écosystème en interagissant avec les autres organismes et l'environnement physique. Il y a aussi la valeur *scientifique* représentant le rôle de l'espèce sur l'avancement du savoir humain incluant la création et la dissémination de la connaissance et, aussi, la valeur *historique*, c'est à dire le rôle qu'a joué l'espèce autrefois. Les implications découlant de la présence d'une espèce indigène en danger ne sont pas les mêmes que pour une espèce surabondante en expansion. Face à la complexité des implications de la faune sur nos sociétés contemporaines, le défi en aménagement des populations animales est d'élaborer des stratégies de gestion visant à optimiser la valeur de la faune (Decker *et al.* 1987; Gilbert et Dodds 2001).

La gestion des populations se définit comme étant la conservation, l'exploitation et le contrôle des populations animales ou végétales (Shea 1998). Les ressources naturelles ont été exploitées de façon variable au cours de l'histoire, allant d'une exploitation chasseur-cueilleur à une exploitation intensive des ressources à haute valeur commerciale. L'homme a

done fait subir différentes modifications aux milieux naturels telles que les extinctions ou introductions d'espèces (Donald et Greenwood 2001; Manchester et Bullock 2000) mais aussi l'explosion démographique de certaines populations sauvages (Frederiksen *et al.* 2001; Gauthier *et al.* 2005; Wagner *et al.* 1997). Chez ces dernières, la transformation du territoire permet aux individus d'adapter leur utilisation de l'habitat. Ils sont alors en meilleure condition physique, ce qui fait augmenter leur taux de reproduction et de survie. Devant les problèmes engendrés par la surabondance et l'expansion d'une population, plusieurs mesures de contrôle peuvent être envisagées, allant de la prévention à l'abattage massif (Ankney 1996; Boyd et Canadian Wildlife Service. 2000; Leader-Williams *et al.* 2001; McComb *et al.* 2001).

Rôle de la chasse en gestion des populations

La chasse sportive est un exemple de solution traditionnellement utilisée par les gestionnaires pour contrôler les populations en expansion. Bien que contestée par les organismes humanitaires, elle est socialement acceptée et génère des bénéfices socio-économiques. De plus, le contrôle par la chasse permet de récolter une certaine proportion d'individus adultes et est donc particulièrement efficace pour modifier la dynamique de population d'espèces opportunistes qui vivent longtemps (Menu *et al.* 2002; Schultz *et al.* 1988).

En contre partie, les activités cynégétiques sont de plus en plus réglementées afin d'éviter la surexploitation de certaines espèces (Anonyme 1979, 1994) mais cela contribue d'autant plus à la croissance de plusieurs populations sauvages (Ankney 1996; Madsen 1991). Toutefois, des exemples démontrent qu'en modifiant les modalités de chasse, il est possible d'atteindre les objectifs qui visent à minimiser les problèmes engendrés par ces populations surabondantes. Citons l'amendement à la Convention sur les oiseaux migrateurs permettant la chasse de printemps de la grande oie des neiges (*Chen caerulescens*; Canadian Wildlife Service 2001) et qui a considérablement affecté la dynamique de ces populations. En fait, les taux de reproduction ont diminué étant donné que les adultes arrivaient sur les sites de nidification en moins bonne condition physique suite au dérangement occasionné par cette chasse exceptionnelle en période prénuptiale. Notons, aussi, les saisons hâtives et tardives de chasse à la bernache du Canada (*Branta canadensis*) instaurées dans le nord-est des États-

Unis et qui sont en partie responsables de la stabilisation de la population de bernaches résidentes (Atlantic Flyway Council 1999; Heusmann 1999). Étant donné les patrons de migration qui diffèrent entre les bernaches migratrices et résidentes (*B. c. interior*, *B. c. maxima*; Malecki *et al.* 2001), les saisons spéciales de chasse ont permis de récolter une plus grande proportion de bernaches résidentes sans toutefois affecter le statut de la sous-espèce migratrice moins abondante (Hindman *et al.* 2003).

Afin de comprendre comment la chasse sportive influence la dynamique des populations, il est primordial d'intégrer l'écologie comportementale à ce type de gestion de la faune (Sutherland 1996; 1998). La répartition et l'abondance des animaux sont influencées par la compétition pour les ressources (Sutherland 1983) ainsi que par le dérangement (Béchet *et al.* 2004; Hill et Frederick 1997) et le risque de prédation (Houston 1993; Lima et Dill 1990). À ce titre, la chasse joue donc un rôle prédominant dans l'utilisation de l'habitat par les animaux. Il a souvent été démontré que les animaux évitent les zones dérangées (Bélanger 1989; Bélanger et Bédard 1990; Riddington *et al.* 1996; Sutherland et Crockford 1993; Tuite 1984). Or, les milieux urbains sont des zones à haut niveau de dérangement mais où la pression de chasse y est peu élevée. Les animaux peuvent donc y retrouver un compromis adaptatif entre le risque de prédation et le taux de dérangement. Ceci rehausse l'importance d'étudier l'écologie comportementale pour mieux comprendre comment la gestion par la chasse affecte la dynamique des populations, cet aspect étant particulièrement pertinent dans les régions façonnées par l'étalement urbain.

Système à l'étude

La bernache du Canada (*Branta canadensis*) est une espèce très diversifiée et cosmopolite avec de nombreuses populations dans chacune des quatre voies migratoires de l'Amérique du Nord (Bellrose 1980). Dans la voie migratoire de l'Atlantique, deux sous-espèces sont principalement représentées et, bien qu'elles partagent les mêmes aires d'hivernage, elles ont des sites de nidification distincts. On retrouve d'abord la sous-espèce migratrice (*B. c. interior*) dont les plus grandes densités d'individus nicheurs se situent dans la péninsule d'Ungava (Malecki et Trost 1990). Cette sous-espèce est traditionnellement observée sous nos latitudes lors des migrations du printemps et de l'automne.

La sous-espèce résidente (*B. c. maxima*) est pour sa part définie comme étant constituée par les bernaches nichant au sud du 48° parallèle et à l'est du 80° méridien (Atlantic Flyway Council 1999). Cette sous-espèce se distingue de la sous-espèce migratrice par sa morphologie et son comportement (Hanson 1965). Les bernaches résidentes sont issues de programmes de relocalisation instaurés dans les années 1960. Les taux de survie et de recrutement de cette population sont favorisés par le fait que les individus utilisent fréquemment des environnements urbains où la chasse est restreinte. De plus, elle réside sous des latitudes méridionales où la production primaire est élevée, ce qui permet un meilleur succès de nidification, un développement accéléré ainsi qu'une maturation sexuelle hâtive (Bellrose 1980; Hanson 1965; Nelson et Otting 1998).

Devant la diversité qui existe chez la bernache du Canada, les gestionnaires font face au défi d'estimer correctement les paramètres démographiques de chacune des populations (Hindman *et al.* 2003). Avant les années 1990, le statut des populations de bernaches dans la voie migratoire de l'Atlantique était évalué à l'aide d'inventaires effectués sur les aires d'hivernage. Cette méthode a conduit à une estimation biaisée de l'augmentation du nombre de bernaches résidentes étant donné que les deux sous-espèces hivernent en sympatrie. Des taux de récolte élevés ont réduit le taux de survie de la population des bernaches migratrices (Hestbeck 1995). De plus, un faible taux de recrutement (saisons de reproduction difficiles en 1994 et 1995; Harvey et Bourget 1997), a accentué le déclin de la sous-espèce migratrice (Hindman *et al.* 1996). La fermeture de la saison régulière de chasse fut imposée en 1995 pour permettre à la sous-espèce migratrice de regagner un statut moins vulnérable.

Depuis la réouverture de la chasse en 1999, les politiques visent à réduire le nombre de bernaches résidentes tout en maintenant le statut de la sous-espèce migratrice stable. Les inventaires sont maintenant effectués durant la période de nidification ce qui permet une meilleure connaissance du statut de la sous-espèce migratrice. De plus, des saisons spéciales de chasse permettent de récolter une plus grande proportion de bernaches résidentes sans toutefois affecter la population migratrice. D'autre part, la migration de mue des bernaches résidentes est un phénomène d'importance qui vient compliquer l'estimation des différents paramètres démographiques. Des bernaches résidentes sous-adultes ou ayant échouées leur nidification migrent avant la mue pour rejoindre des habitats de la forêt boréale et du Nouveau-Québec (Abraham *et al.* 1999; Salomonsen 1968). Lors de la migration automnale,

trois populations sont donc présentes sous nos latitudes; les bernaches résidentes qui ont nichées dans la région, les bernaches migratrices qui ont nichées dans le nord du Québec et les bernaches résidentes qui ont muées dans le nord du Québec. Ainsi, la récolte de chasse est hétérogène ce qui rend problématique l'estimation des taux de récolte de chacune des populations.

Dans la voie migratoire de l'Atlantique, la population totale de bernaches résidentes est estimée à plus de 1 million d'individus dont une proportion importante effectue une migration de mue dans le nord du Québec (Atlantic Flyway Council 1999; Comité sur la sauvagine du service canadien de la faune 2004). Actuellement, les objectifs du plan d'aménagement adopté par l'Atlantic Flyway Council (1999) visent à réduire l'effectif à 650 000 individus en récoltant environ 400 000 bernaches résidentes annuellement (Hindman *et al.* 2003). Cependant, ces objectifs sont difficiles à atteindre étant donné le nombre décroissant de chasseurs (Heusmann 1999). Afin d'élaborer des stratégies adéquates de contrôle, davantage d'études sur la dynamique de population de la bernache résidente sont nécessaires notamment dans les régions où celle-ci prend de l'expansion.

Dans le sud du Québec, les premiers individus nicheurs de la sous-espèce résidente ont été recensés au début des années 1990 sur les îles de Varennes (Québec, 45°40' N, 73°27' W). Plus de 3000 individus ont été dénombrés dans le sud-ouest du Québec lors de relevés aériens effectués à l'été 2003 (J. Rodrigue, SCF, comm. pers.). Le taux de croissance élevé de cette nouvelle population ($\lambda=1,41$; Giroux *et al.* 2001) semble démontrer que l'habitat de reproduction n'est pas limitant. Afin de prévenir les conflits vécus en Ontario et dans de nombreux états américains (problèmes de santé publique liés à la concentration de fiente, nuisance sur les terrains de golf et les parcs, déprédation des cultures et danger sur les sites aéroportuaires), des mesures spéciales devront être adoptées par les gestionnaires du Québec. Afin d'orienter des stratégies de gestion de façon adéquates, il est nécessaire de mieux connaître le comportement des bernaches établies dans le sud du Québec et ce, en relation avec les différents paramètres qui affectent la dynamique de population.

Objectifs du mémoire

Le premier objectif de ce mémoire est de caractériser les différentes populations de bernaches dans la voie migratoire de l'Atlantique. Puisqu'il existe un dimorphisme de taille

entre les bernaches résidentes et migratrices, la morphométrie devrait permettre de classer les individus selon leur sous-espèce. De plus, étant donné que les différentes populations muent dans des habitats distincts, les ratios isotopiques de carbone et d'azote contenus dans les plumes primaires devraient varier entre ces populations.

Le deuxième objectif vise à évaluer le rôle potentiel de la chasse sportive pour contrôler la croissance de la population de bernaches résidentes établie dans le sud du Québec. La répartition des individus qui nichent dans la région péri-urbaine de Montréal devrait être influencée par le gradient de pression de chasse qu'on y retrouve. De plus, l'utilisation de l'habitat durant la chasse devrait être influencée par le risque de prélèvement.

CHAPITRE 1 : IDENTIFICATION DE POPULATIONS DE BERNACHES DU CANADA
À L'AIDE DE MESURES MORPHOMETRIQUES ET D'ISOTOPES STABLES

Ce chapitre sera soumis pour publication sous le titre suivant :

Identifying populations of Canada geese using morphometric measurements and stable isotopes.

Matthieu Beaumont , Jean-François Giroux, Jean Rodrigue, Richard Cotter, Jack Hughes,
and Ted Nichols

Abstract: Population status of Canada geese (*Branta canadensis*) in the Atlantic Flyway has evolved through time and managers need demographic information and harvest assessment for each population. We used a combination of morphometric measurements and isotopic signatures of feathers ($\delta^{15}\text{N}$, $\delta^{13}\text{C}$) to assign individual Canada geese according to their subspecies and molting sites. We first compared different sub-samples of birds caught in the same area to determine those that were the most representative by providing the best discriminant functions. We distinguished >89% of after hatching year (AHY) *B. c. interior* and *B. c. maxima* using skull measurements alone. *B. c. interior* from the west and east coasts of Ungava peninsula were also well associated to their respective breeding area (>84% with 2 or 3 morphological variables). The ^{15}N isotope was more useful than ^{13}C to discriminate geese from different molting areas. For AHY individuals, $\delta^{15}\text{N}$ of feathers was greater for *B. c. maxima* that bred and molted in southern Quebec (>8.3‰) than for those that molted on the west coast of the Ungava peninsula. For hatching year (HY) individuals, $\delta^{15}\text{N}$ of feathers was greater for *B. c. maxima* (>7.6‰) than for *B. c. interior*. Our study provides an economical and reliable tool for identifying subspecies and populations of Canada geese during large-scale banding operations and harvest surveys.

Key words: Atlantic Flyway, *Branta canadensis interior*, *Branta canadensis maxima*, Canada goose, discriminant function, molt migration, morphometric, populations, Quebec, stable isotopes

Introduction

In the Atlantic Flyway, Canada goose populations mix on wintering areas in north-eastern United States but have distinct breeding ranges. The migratory Canada geese (*Branta canadensis interior*) nest throughout the Nunavik (northern Quebec) with concentrations along the East and West coasts of Ungava peninsula (Malecki and Trost 1990). The temperate goose population (*B. c. maxima*) is defined by its breeding range south of the 48° N latitude and east of 80° W longitude (Atlantic Flyway Council 1999). However, a portion of *B. c. maxima* sub adults and failed breeders migrate to northern latitudes up to the *B. c. interior* breeding range to molt (Hanson 1965; Abraham et al. 1999). In addition, local breeding populations of *B. c. maxima* are becoming established in southern Quebec and are harvested in fall along with the returning moult migrant *B. c. maxima* and the migratory *B. c. interior* (Giroux et al. 2001).

Before the 1990s, the status of the Atlantic Flyway population of Canada geese was assessed by surveys conducted throughout the wintering area (Hindman and Ferrigno 1990; Atlantic Flyway Council 1999). These assessments were biased by the increase and expansion of *B. c. maxima* geese. Concomitantly, poor breeding conditions and high harvest rates caused the decline of the migratory Canada goose population (*B. c. interior*) leading to closure of sport hunting in 1995 (Hindman et al. 1996). Following the increase in the number of *B. c. interior* breeding pairs, restrictions have now been lifted but managers need tools to accurately estimate demographic parameters of each population.

Increases in population size have brought new hunting measures aimed at controlling overabundant resident geese while preventing over harvest of the less abundant migratory subspecies (Atlantic Flyway Council 1999). To achieve this goal, it is essential to precisely characterize the geese banded on summer range because harvest and survival rates are assessed from these marked populations. Furthermore, estimation of the contribution of each population in the harvest according to regional units and through time during the hunting period is required to adapt regulations aimed at achieving management objectives.

Biologists have collected measurements of Atlantic Flyway Canada geese during banding operations for a long time. The morphological differences among subspecies or stocks of Canada geese have often been exploited to refine management capabilities (Moser and Rolley 1990; Mercendino et al. 1994). More recently, stable isotope analyses of feathers and

molecular techniques have been suggested for identifying subpopulations of waterfowl (Caccamise et al. 2000; Hobson et al. 2000; Pearce et al. 2000; Hebert and Wassenaar 2001; Scribner et al. 2003). While accurate genetic characterization is difficult when populations are panmictic (Cronin *et al.* 1996), environmental variations in isotopic signatures might be used to trace molting origin of bird. Stable isotope ratios in the tissues of an animal depend on its diet and its habitat use (Deniro and Epstein 1981). Keratin tissues, like feathers, are inert after formation and their isotopic signature can thus reveal the habitat types where they were grown (Mizutani et al. 1990).

Our main objective was to build classification models that would allow managers to associate an individual goose to its population within the Atlantic Flyway. We wanted to develop a technique that combined both morphometric measurements and stable isotope analyses of primary feathers. We used a large data set of measurements taken on birds captured in southern and northern Quebec along with isotope signatures of primary feathers collected on molting birds in the same areas. We first determined the most representative samples that provide the best discriminant functions that separate (1) AHY *B. c. interior* from *B. c. maxima* and (2) AHY *B. c. interior* from the West and East coasts of Ungava peninsula. Because accurate assignment to a subspecies relies on correct sex determination and precise measurements (Rasmussen et al. 2001), we evaluated the extent of measurement and sexing errors in these large data sets. Finally, using nitrogen and carbon isotope ratios of primary feathers, we wanted to separate individuals of *B. c. maxima* breeding in southern Quebec from those molting in Nunavik and to associate hatching year (HY) geese to their respective subspecies or hatching grounds.

Methods

Study area

Molting Canada geese were captured during banding operations in three areas (Fig. 1). Two areas were located in Nunavik and are characterized by coastal tundra habitats. These areas are spread along the West (WU) and East (EU) coasts of Ungava peninsula (58°25' to 60°40' N and 67°10' to 77°50' W). *B. c. interior* breeds on both coasts (Malecki and Trost 1990; Bordage and Plante 1993), but molting *B. c. maxima* are found mainly on the WU coast (Harvey and Rodrigue, 2002). The third area is located east of Montreal in southern Quebec

(SQ; 45°40' N, 73°27' W) and involves a recently established breeding population of *B. c. maxima* (Giroux et al. 2001). The SQ study area includes both natural and urban habitats that are described in details by Doiron (2006).

Sample collection

Banding operations took place between 1996 and 2004 in WU and EU and between 1999 and 2004 in SQ. Geese were driven into corral traps by ground crews assisted by boats or helicopters. All individuals were aged (AHY and HY) based on plumage and sexed by cloacal examination. Skull, culmen and tarsus bone were measured by experienced measurers using vernier callipers (± 0.1 mm) on a sample of birds (Dzubin and Cooch 1992). Not all birds were measured and not all morphological structures were collected on measured birds. Primary feathers in SQ were sampled in 2003 and 2004, whereas those in WU and EU were collected only in 2003. We removed approximately 80 mm of the distal end of the posterior vane on the newly grown eighth primary feather (see Caccamise et al. 2000).

Sample and model selection

Three independent samples were generated to determine the best samples that would minimise the presence of molting *B. c. maxima* in the WU and EU *B. c. interior* data sets. The first sample referred to as “known *interior*” included birds first banded as HY in WU and EU and recaptured in subsequent years as AHY. Because the sample size of “known *interior*” was relatively small, a second sample was generated by considering catches composed of at least 75% HY birds. This sample referred to as “family groups” was based on the observations of Hanson (1965) and Didiuk (1979) that molt migrants *B. c. maxima* generally avoid flocks of *B. c. interior* with goslings. Finally, we pooled the rest of the birds measured during the study to compose the “all birds” sample. ANOVAs were performed to evaluate whether measurements of each morphological variable differed between the 3 independent samples for the WU and EU data sets.

According to Giroux et al. (2001), measurements taken on SQ geese could include a few *B. c. interior* birds, potentially reducing the accuracy of the models established to discriminate between *B. c. maxima* and *B. c. interior*. We thus compared measurements taken on geese captured more than once in SQ (“recaptured” sample) assumed to be *B. c. maxima*

with the rest of the data set (“all birds” sample). In the “known *interior*” and “recaptured” samples, we discarded data from birds that were sexed differently in successive captures (see below).

Analyses of variance were performed to compare measurements of *B. c. interior* geese between breeding grounds (WU and EU) for each sample. The same was done to compare *B. c. maxima* and *B. c. interior* measurements using the “recaptured” and “known *interior*” samples, respectively. Comparisons were made on males and females separately because of sexual dimorphism (Hanson 1951). Conditions for normality of the distribution and homoscedasticity were met for all analyses (Kolmogorov-Smirnov tests; $P > 0.05$).

We performed discriminant function analyses (DFA) on morphometric measurements (skull, culmen, and tarsus) using different samples to separate *B. c. maxima* and *B. c. interior* AHY. We first compared the “known *interior*” and the “family groups” samples of WU and EU with the “recaptured” sample of SQ. Next, we compared the “all birds” samples for the 3 areas. The same was done to separate *B. c. interior* geese from each coast of the Ungava peninsula (WU vs. EU). Because skull is the most commonly taken measurement during banding operations and because it has been considered as the best univariate predictor in other studies (Moser and Rolley 1990; Merendino et al. 1994), we did not generate discriminant functions based solely on culmen or tarsus. The percentages of birds well discriminated by the various DFA models were used to evaluate their accuracy.

Measurement and Sexing Errors

Using birds measured twice during different years, we estimated among- and within-bird component of the total variance using a Model II ANOVA for each morphological variable, area, and sex. The total variance among measurement values includes the variance due to measurement error (ME) and the variance associated to size variation of the morphological structures (Bailey and Byrnes 1990). The percentage of the total variance among measurement values explained by the ME was calculated using the formula presented by Bailey and Byrnes (1990): $\% \text{ ME} = [s^2_{\text{within}} / (s^2_{\text{among}} + s^2_{\text{within}})] \times 100$. The rate of error in sex assignment was established by comparing the sex attribution for individuals captured more than once and this was compared among the 3 areas with a χ^2 test.

Isotope analyses

Feather samples were cleaned of surface oils by floating them during 48 hours in a 2:1 mixture of methanol/chloroform, rinsing twice with methanol and drying for 24 hours under a hood. Between 0.6 and 0.75 mg of feather material was combusted using a Carlo-Erba auto analyser (model NA1500 series 2). The resulting CO₂ and N₂ gas from the samples was separated chromatographically and introduced in a VG Optima triple collector isotope-ratio mass-spectrometer via an open split. Stable carbon (¹³C) and nitrogen (¹⁵N) isotope ratios were expressed in delta (δ) notation as the deviation in parts per thousand (‰) from the PDB (Pee Dee Belemnite) and the AIR (atmospheric nitrogen) standards, respectively. Using internal laboratory and isotopic primary standards, sample repeatability for δ¹³C and δ¹⁵N was established at ±0.2‰.

We first compared δ¹³C and δ¹⁵N between years, sexes, and ages for SQ *B. c. maxima* as well as between areas, sexes, and ages for *B. c. interior* with ANOVAs. We also compared isotope signatures of feathers between sexes of WU AHY *B. c. maxima*, between SQ and WU of *B. c. maxima* AHY, and between the 2 subspecies of HY geese using t-tests. DFAs were performed to first discriminate *B. c. maxima* AHY between SQ and WU and secondly between HY birds of each subspecies.

Results

Sample and model selection

We measured 6,101 and 3,846 *B. c. interior* AHY geese in WU and EU, respectively between 1996 and 2004 and 618 *B. c. maxima* AHY in SQ between 1999 and 2004. Not all measurements were taken on all birds, thus sample sizes varied for the different analyses. Means of the variables measured for the “known *interior*” sample were 11 times out of 12 lower than those for the “family groups” and “all birds” samples (Table 1). These differences, however, were not statistically significant except for culmen of EU females ($F=3.431$; $df=1, 725$; $P=0.033$).

Samples for the SQ *B. c. maxima* were compared for each variable and sex but no difference was observed between the “all birds” and “recaptured” samples ($P>0.05$). However, means of measures for the “recaptured” sample were consistently greater than for the “all birds” sample (Table 1).

ANOVAs performed on all samples of *B. c. interior* confirmed that AHY geese were significantly smaller in WU than in EU for both sexes ($P < 0.0001$ for all tests; Table 1). Based on the “family groups” samples, mean skull measurements of EU females and males were larger than those of WU geese by 3.5 and 3.0 mm, respectively. Comparisons made with the “recaptured” and “known *interior*” samples indicated that AHY *B. c. maxima* were larger than geese from WU and EU ($P < 0.0001$ for all tests; Table 1). Mean skull measurements of *B. c. maxima* were 12.4 and 13.5 mm longer than for WU females and males, respectively while they were 9.2 and 10.7 mm longer than for EU females and males, respectively.

DFA models using morphometric measurements generally provided good discrimination among groups (Table 2). However, classifications performed with the “known *interior*” samples were usually more accurate to classify the 2 subspecies while those based on the “family groups” samples were better to separate EU and WU *B. c. interior*. The smaller difference between WU and EU *B. c. interior* than between the 2 subspecies resulted in a lower proportion of individuals being classified with accuracy. Using 3 morphometric variables instead of 2 or 1 gave, on average, a better classification (93.4, 91.3, and 87.1%, respectively). On the other hand, DFAs based on skull and culmen were sometimes less accurate than those using skull only.

To select equations that would minimize the presence of *B. c. maxima* in the *B. c. interior* data set, we chose the “known *interior*” sample as the most parsimonious selection except to separate EU and WU birds for which the “family groups” samples were used (Table 3). For *B. c. maxima* data set, the “recaptured” sample was chosen for the DFA analysis based on the same principle. The various equations can be used in Nunavik during banding operations to separate subspecies on the respective breeding sites and in southern Quebec to characterize the harvest composition both at the subspecies level and for the 2 breeding northern sites. The number of variables to be measured depends on the time available and the accuracy required (Table 3).

Measurement and sexing errors

A total of 58 birds were measured twice at WU and EU. In SQ, the sample size was too limited to carry out the analysis ($n < 5$). On average, % ME was greater for culmen followed

by tarsus and skull. It was also greater for females than males in both areas and greater in EU than WU, especially for females (Table 4). The measurement error of skull for experienced observers was usually <2 mm for live *B. c. interior*.

A total of 750, 507, and 192 birds were captured and sexed two or more times during banding operations in WU, EU, and SQ, respectively. Different sex attribution was recorded for 54 of these birds (3.7%). There were a lower percentage of mistakes for *B. c. interior* in Nunavik (1.6 and 3.7% in WU and EU, respectively) than for *B. c. maxima* in SQ (12.0%; $\chi^2=43.44$, $df=2$; $P<0.0001$).

Isotope analyses

We analysed ^{13}C and ^{15}N in newly grown primary feathers of 43 *B. c. maxima* HY collected in SQ as well as of 20 *B. c. interior* HY sampled in WU and 20 in EU. We also analysed primary feathers of 45 and 17 *B. c. maxima* AHY geese that have molted in SQ and WU, respectively. Finally, we analysed 22 and 19 *B. c. interior* AHY geese that have been trapped in WU and EU, respectively.

In SQ, no difference was found between years ($\delta^{13}\text{C}$: $F=0.715$; $df=1$, 87; $P=0.401$; $\delta^{15}\text{N}$: $F=0.202$; $df=1$, 87; $P=0.654$) or sexes ($\delta^{13}\text{C}$: $F=0.388$; $df=1$, 87; $P=0.535$; $\delta^{15}\text{N}$: $F=0.262$; $df=1$, 87; $P=0.611$) for *B. c. maxima*. However, primary feathers of AHY were more enriched in ^{13}C than those of HY ($F=7.657$; $df=1$, 87; $P=0.007$; Fig. 2). No such difference was observed for $\delta^{15}\text{N}$ ($F=1.959$; $df=1$, 87; $P=0.166$). In Nunavik, no difference was found between sexes for AHY *B. c. maxima* ($\delta^{13}\text{C}$: $t=0.382$; $df=16$; $P=0.708$; $\delta^{15}\text{N}$: $t=0.083$; $df=16$; $P=0.935$) nor for *B. c. interior* ($\delta^{13}\text{C}$: $F=0.203$; $df=1$, 80; $P=0.653$; $\delta^{15}\text{N}$: $F=0.154$; $df=1$, 80; $P=0.696$). On the other hand, area and age significantly affected delta values of *B. c. interior* feathers both for ^{13}C (area: $F=18.786$; $df=1$, 80; $P<0.001$; age: $F=65.005$; $df=1$, 80; $P<0.0001$) and ^{15}N (area: $F=12.395$; $df=1$, 80; $P=0.007$; age: $F=18.813$; $df=1$, 80; $P<0.001$). However, the patterns were not consistent for ^{13}C (Fig. 2). AHY were less enriched than HY in EU with no such difference in WU.

Heteroscedasticity of $\delta^{13}\text{C}$ in feather samples between southern and northern birds (SQ: $SE=0.331$; WU and EU: $SE=0.051$) precluded the use of parametric statistics (e.g. DFA) to discriminate these groups. For this reason, we only compared $\delta^{15}\text{N}$ between SQ and WU AHY *B. c. maxima* and found that feathers from SQ geese were more enriched in ^{15}N than

those of WU ($t=8.833$; $df=60$; $P<0.0001$). DFA model using $\delta^{15}\text{N}$ correctly assigned 95% of AHY *maxima* to their respective molting area and $y > 0$ in the equation $y = -8.3 + \delta^{15}\text{N}$ indicates AHY *B. c. maxima* from SQ.

Although delta values of feathers were significantly different between *B. c. interior* captured in WU and EU, the DFA had a poor discriminatory power (<75% with 2 isotopes) for both age groups. Hence, we only compared $\delta^{15}\text{N}$ between the 2 subspecies for HY and found that feathers of *B. c. maxima* had significantly higher values than of *B. c. interior* ($t=7.780$; $df=81$; $P<0.0001$). Individuals were well associated to their respective subspecies with 90% of the birds being well classified while $y > 0$ in the equation $y = -7.6 + \delta^{15}\text{N}$ designates a HY *B. c. maxima* reared in SQ.

Finally, $\delta^{13}\text{C}$ in feathers of *B. c. maxima* AHY captured in WU were significantly greater than those of *B. c. interior* ($t=2.934$; $df=37$; $P=0.006$) while no such difference was found for $\delta^{15}\text{N}$ ($t=1.264$; $df=37$; $P=0.216$).

Discussion

Representative samples are required to properly characterize different populations of birds based on morphometric measurements. We had identified the molt migration of *B. c. maxima* into the *B. c. interior* breeding range in Nunavik as a potential problem to adequately characterize *B. c. interior* individuals captured on the West and East coasts of the Ungava peninsula. By limiting our analyses to recaptured AHY birds that had been banded as HY in northern Quebec (referred as “known *interior*”), we reduced our sample size but improved the accuracy of DFA models suggesting that the full data set was indeed contaminated by molt migrant *B. c. maxima*. Comparisons of means also support this contention. Moser and Rolley (1990) and Merendino et al. (1994) attempted to overcome this problem by considering individuals captured twice in more than 3 years and females with a brood patch as *B. c. interior*. We did not use these criteria because failed breeders (with a brood patch) can also migrate to these latitudes (Abraham et al. 1999). With the increasing effort at controlling the establishment of resident populations at southern latitudes by adding Canada goose eggs, this should result in a greater number of failed breeders (possibly the same individuals in successive years) that migrate to northern Quebec to molt.

Hanson (1965) and Didiuk (1979) argued that molting *B. c. maxima* did not share the same habitats as family groups of *B. c. interior*. This is supported by the difference in isotopic signatures of birds captured in the Ungava peninsula, at least for ^{13}C , which suggests different habitat use or food habits during the molt of these two subspecies. However, our samples of geese captured in family groups were not as helpful as the “known *interior*” samples to discriminate between *B. c. interior* and *B. c. maxima*. Moreover, the mean values of the morphometric measurements were often larger for this sample than for the “known *interior*” suggesting the presence of *B. c. maxima* in these groups. The “family groups” sample gave, nevertheless, the best DFA models to classify *B. c. interior* from the respective coast of the Ungava peninsula, possibly because of the larger sample size.

The *B. c. interior* subspecies is composed of several morphologically distinct populations (Leafloor and Rusch, 1997; Dickson, 2000). Clinal size variation has also been shown for *B. c. interior* in the Mississippi Flyway (Moser and Rolley, 1990). Detailed knowledge of geographic variation in body size is thus needed to improve effectiveness of classification at the Flyway level. Although Canada geese captured in SQ were comparable in size to *B. c. maxima* from southern Ontario and the Mississippi Flyway (Moser and Rolley 1990; Merendino et al. 1994), geographical differences still exist among populations of *B. c. maxima* within a Flyway. However, characterization of this subspecies must not be based at the regional scale because harvest of summer-banded individuals indicated interchange across regions (Moser and Rolley, 1990).

Population discrimination using morphometric measurements is subject to diverse biases (Leafloor and Rusch 1997; Leafloor et al. 1998; Thompson et al. 1999; Rasmussen et al. 2001). The imprecision of a measurement depends on the morphological variable and the experience of the measurer. To investigate measurement errors (ME), our analyses were based on homogenous samples established for each sex class in each region. This reduced the among-birds variation and estimated more accurately the relative contribution of the within-birds variance associated to the measurers. Moreover, our data were collected in the field on live birds by different experienced measurers, which is different than measuring dead immobile birds as done by Rasmussen et al. (2001). Nevertheless, we found similar magnitude of variability within goose measurement than Rasmussen et al. (2001). Although it would be preferable to have the same measurers each year, this is rarely possible when the

banding programs extend over many years and involve several agencies. Variables with low % ME such as skull should be privileged for morphometric classification to reduce the likelihood of type II error (Toft 1983).

This study is the first to investigate the rate of error in sex attribution of Canada goose. More than 10% of geese recaptured in SQ were sexed differently compared to less than 4% in the Ungava peninsula. This might be due to the robustness of *B. c. maxima* that make manipulations of the birds more difficult. Brown and Brown (2002) found error of 11% in sex assignment for HY mute swans (*Cygnus olor*) banded in UK and the bias was the same for males and females. This bias does not influence the accuracy of our DFA models because we only included recaptured geese that had been sexed identically. However, failure in sex assignment may have consequences on classification when the measurements fall within the range of overlap between *B. c. maxima* females and *B. c. interior* males. For these birds, subspecies classification is more likely to be false if individuals are wrongly sexed. Our data indicated that tolerance intervals (TI) set at an α -level of 95% overlap by 10 mm for skull measurements of *B. c. maxima* females (lower TI=116.1 mm) and *B. c. interior* males (higher TI=126.1 mm).

The isotopic signatures carried in primary feathers of a bird reflect the environment in which it molted and this can be used to trace source of elements (Mizutani et al. 1992; Hobson et al. 2000). However, like Graves et al (2002) recently found in black-throated blue warblers (*Dendroica caerulescens*), enrichment in Canada goose is age-specific and must be considered when using stable isotope to delineate geographic origin of birds. Different patterns of isotopic fractionation between AHY and HY during keratin synthesis may explain this finding but a distinct diet between goslings and adults may be also important. During the rearing period on the west coast of the Ungava peninsula, AHY and HY Canada geese consumed different varieties of plants and different plant parts (Cadieux et al. 2005). Compare to AHY that consumed high-energy plant part, the diet of goslings is generally with higher nitrogen concentration.

The ^{13}C isotope has a limited capacity to act as a general marker to link individual geese to their molting area. Unlike Wassenaar and Hobson (2000; 2001), this finding may not be due to a similar contribution of C_3 and C_4 plants between geographic areas, but rather to the

greater variability of the SQ site and the use of both anthropic and natural habitats by broods of *B. c. maxima* along the St-Lawrence river (Doiron 2006).

Conversely, different nitrogen isotope concentrations in goose feathers reveal different sources of nitrogen. Higher $\delta^{15}\text{N}$ in the feathers of SQ geese may reflect the entry of excess fertilizer nitrogen into local river systems (Heaton 1986; Kendall 1998; Hebert and Wassenaar 2001). Multiple isotope analyses have been proposed to trace the origin of birds (Chamberlain et al. 1997; Caccamise et al. 2000; Wassenaar and Hobson 2001; Hebert and Wassenaar 2005). However, hydrogen and sulfur isotopic analyses of complex organic structure like keratin need extensive and expensive laboratory manipulations and may be less suitable for large scale sampling.

Management implications

Canada goose management objectives in the Atlantic Flyway have evolved during the second half of the 20th century following the expansion of the temperate subspecies (*B. c. maxima*). The effective management of this species requires demographic information and harvest estimates for each population (Hindman et al. 2003). Our study shows the effectiveness of using morphometric measurements and isotope signatures to properly identify the population source of an individual Canada goose.

First, our results can be used to discriminate molting *B. c. maxima* and breeding *B. c. interior* AHY geese captured in the Ungava peninsula region. Using only skull measurements, the most commonly measure taken during banding operations, we can now accurately assign >89% of the geese to their subspecies. Skulls larger than 118 and 125 mm for females and males, respectively, indicate *B. c. maxima*. More measurements can be taken to increase accuracy but the time to process the birds will lengthen the banding operations, which can affect goose survival (Menu et al. 2001).

Our results are also useful to estimate the contribution of each population in the harvest. Morphometric measurements classifying subspecies of harvested AHY geese might be used during fall at southern latitudes. Morphometric measurements are also valuable to distinguish the population source (west and east coasts of Ungava peninsula) of harvested *B. c. interior* geese (models' accuracy >84% when using 2 or 3 variables). For AHY *B. c. maxima*, $\delta^{15}\text{N}$ of primary feathers collected on harvested geese in southern Quebec could help to distinguish

ones that have bred in this region from those that have molted in Ungava peninsula region (models' accuracy >90% using $\delta^{15}\text{N}$). However, Caccamise et al. (2000) found that inland *B. c. maxima* trapped in New Jersey were less enriched in stable isotopes than *B. c. maxima* trapped in southern Quebec. Hence, precaution must be taken when using isotopic signature to manage sub-populations of resident Canada geese. For HY geese, $\delta^{15}\text{N}$ of primary feathers could be helpful to classify harvested geese at the subspecies level.

To establish regulations and achieve specific management objectives, Part Surveys based on wings and tails of bagged waterfowl are used to investigate population contribution according to regional units and through time. However, to assess subspecies and sexes of individual Canada geese, collection of head and rump of birds would have to be added to this survey, which may be difficult to implement. On the other hand, several sampling stations distributed throughout different hunting districts could be deployed for specific and punctual surveys. Once a representative sample is achieved, the combined use of morphometric measurements and isotope analyses should be cost-effective to differentiate populations of Canada geese.

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Table 1. Mean skull, culmen, and tarsus length (mm) of after hatching year male and female Canada geese captured on the West (WU) and East coast (EU) of Ungava peninsula, 1997-2004 and southern Quebec (SQ), 1999-2004. See the text for description of the samples.

Subspecies (area) Sample	Female									Male								
	Skull			Culmen			Tarsus			Skull			Culmen			Tarsus		
	x	se	n	x	se	n	x	se	n	X	se	n	x	se	n	x	se	n
<i>B. c. interior</i> (WU)																		
Known <i>interior</i>	111.1	0.25	124	48.6	0.33	86	78.4	0.46	85	117.6	0.44	78	50.7	0.57	47	85.1	0.67	47
Family groups	111.7	0.11	490	48.6	0.12	164	78.8	0.23	162	118.0	0.17	443	51.4	0.22	137	83.9	0.30	137
All birds	111.6	0.08	2597	48.8	0.35	667	78.7	0.13	659	117.8	0.08	2369	51.7	0.51	538	84.4	0.16	529
<i>B. c. interior</i> (EU)																		
Known <i>interior</i>	114.3	0.46	48	51.4	0.38	42	81.3	0.51	42	120.4	0.66	35	54.2	0.50	30	88.0	0.58	30
Family groups	115.2	0.37	105	52.8	0.34	53	81.9	0.49	42	121.0	0.39	92	55.4	0.42	37	88.6	0.58	33
All birds	114.6	0.09	1872	52.1	0.21	633	82.0	0.15	622	120.8	0.09	1694	55.0	0.22	586	88.3	0.17	578
<i>B. c. maxima</i> (SQ)																		
Recaptured	123.5	0.42	86	55.8	0.26	85	88.8	0.39	78	131.1	0.50	67	59.4	0.33	67	95.9	0.57	44
All birds	123.4	0.27	256	55.7	0.16	256	88.0	0.30	152	130.7	0.33	209	59.1	0.22	206	95.4	0.46	94

Table 2. Percent of Canada geese correctly classified to their subspecies or breeding area using discriminant function analyses based on 3 different samples of *B. c. interior* captured on the West (WU) and East coast (EU) of Ungava peninsula and two samples of *B. c. maxima* captured in southern Quebec (SQ).

Variable	Female (%)				Male (%)			
	<i>Maxima</i> vs. <i>Interior</i>	<i>Maxima</i> vs. <i>Int. WU</i>	<i>Maxima</i> vs. <i>Int. EU</i>	<i>Int. WU</i> vs. <i>Int. EU</i>	<i>Maxima</i> vs. <i>Interior</i>	<i>Maxima</i> vs. <i>Int. WU</i>	<i>Maxima</i> vs. <i>Int. EU</i>	<i>Int. WU</i> vs. <i>Int. EU</i>
Sample of <i>B. c. interior</i> *								
Skull-Culmen-Tarsus								
Known <i>interior</i>	94.6	98.1	93.3	68.5	95.0	98.9	93.2	68.8
Family groups	94.0	97.9	87.4	88.7	92.5	97.8	93.5	84.1
All birds	86.3	96.5	82.0	77.6	87.2	96.1	85.2	76.3
Skull-Culmen								
Known <i>interior</i>	94.4	97.7	88.2	68.0	94.4	97.4	90.7	67.5
Family groups	89.1	95.6	84.8	85.3	90.0	94.2	86.5	81.0
All birds	86.8	95.1	81.8	76.4	87.2	93.8	85.1	75.5
Skull								
Known <i>interior</i>	94.6	98.1	88.8	68.6	93.9	94.5	89.2	63.7
Family groups	91.9	95.1	86.9	70.4	94.0	95.9	89.3	66.2
All birds	90.4	94.5	87.6	67.9	90.9	93.9	89.2	67.2

*Known *interior* and Family groups samples of *B. c. interior* were compared with the recaptured sample of *B. c. maxima* while the All birds samples of both subspecies were compared

Table 3. Selected equations for classifying subspecies of AHY Canada geese based on skull (SK), culmen (CU), and bone tarsus (TA) measurements. SQ, WU, and EU indicate Southern Quebec, West coast, and East coast of Ungava peninsula, respectively.

Classification (use) ^a	
Female	Male
<i>Maxima</i> vs. <i>interior</i> (To be used in SQ for harvest)	
$y = -234.25 + 1.43*SK + 0.02*CU + 0.77*TA$	$y = -170.68 + 0.65*SK + 0.30*CU + 0.80*TA$
$y = -208.05 + 1.64*SK + 0.27*CU$	$y = -157.67 + 1.16*SK + 0.23*CU$
$y = -118.16 + SK$	$y = -124.88 + SK$
<i>Maxima</i> vs. <i>WU interior</i> (To be used on WU during banding)	
$y = -293.60 + 1.59*SK + 0.45*CU + 1.00*TA$	$y = -239.66 + 1.25*SK + 0.38*CU + 0.70*TA$
$y = -245.05 + 1.77*SK + 0.71*CU$	$y = -199.88 + 1.39*SK + 0.50*CU$
$y = -117.18 + SK$	$y = -124.35 + SK$
<i>Maxima</i> vs. <i>EU interior</i> (To be used on EU during banding)	
$y = -196.95 + 1.00*SK + 0.06*CU + 0.89*TA$	$y = -188.25 + 0.69*SK + 0.52*CU + 0.77*TA$
$y = -163.34 + 1.29*SK + 0.19*CU$	$y = -150.51 + 1.11*SK + 0.20*CU$
$y = -118.64 + SK$	$y = -125.39 + SK$
<i>EU interior</i> vs. <i>WU interior</i> (To be used in SQ for harvest)	
$y = -87.66 + -0.31*SK + 1.99*CU + 0.28*TA$	$y = -89.14 + -0.13*SK + 1.04*CU + 0.57*TA$
$y = -63.13 + -0.17*SK + 1.63*CU$	$y = -87.06 + 0.35*SK + 0.85*CU$
$y = -114.11 + SK$	$y = -118.23 + SK$

^a Where $y > 0$ means *B. c. maxima* for the first 3 series of models and EU *B. c. interior* for the last series.

Table 4. Variance components estimates for Canada geese morphological variables measured (mm) on AHY females and males captured on the west (WU) and east (EU) coasts of Ungava peninsula, 1996-2004.

Variable	Variance component	WU		EU	
		Females (n=10)	Males (n=12)	Females (n=17)	Males (n=19)
Skull	Among geese	2.83	9.89	9.13	10.33
	Within geese	0.71	2.25	2.04	1.30
	% measurement error	19.9	18.5	18.3	11.2
Culmen	Among geese	2.77	5.76	2.96	3.19
	Within geese	0.78	2.22	3.00	2.26
	% measurement error	21.9	27.8	50.3	41.5
Tarsus	Among geese	5.87	15.13	5.03	10.55
	Within geese	3.10	2.13	2.29	1.15
	% measurement error	34.6	12.3	31.3	9.8



Fig. 1. Location of sampling areas in northern and southern Quebec. *B. c. interior* geese were captured along the west (WU) and east coasts (EU) of Ungava peninsula whereas AHY *B. c. maxima* geese were captured in southern Quebec (SQ) and WU. Stars represents capture sites.

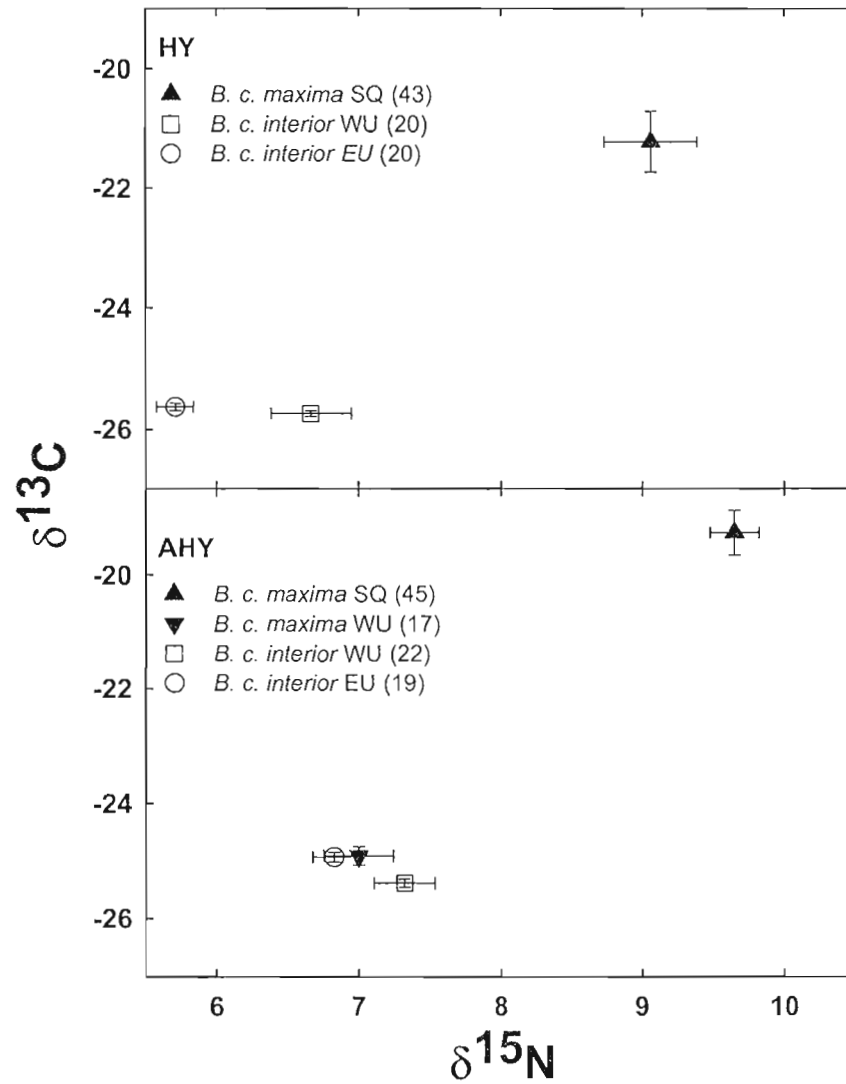


Fig. 2. Mean ($\pm$ SE) delta values for carbon ($\delta^{13}\text{C}$) and nitrogen ($\delta^{15}\text{N}$) isotopes of feathers collected from hatching year (HY) and after hatching year (AHY) Canada geese in Quebec, 2003-2004. Values are plotted according to subspecies (*B. c. maxima* and *B. c. interior*) and summer range (SQ = Southern Quebec; WU = west coast of Ungava peninsula; EU = east coast of Ungava peninsula). Sample size shown in parentheses.

CHAPITRE 2 : DISPERSION POST-REPRODUCTIVE DES BERNACHES DU CANADA
RESIDENTES DANS LE SUD DU QUEBEC

Ce chapitre sera soumis pour publication sous le titre suivant :

Post-breeding dispersal of resident Canada geese in southern Quebec.

Matthieu Beaumont, Jean-François Giroux, Jean Rodrigue

Abstract: Individual behavior that reduces vulnerability to sport harvest can affect population dynamics and must be considered when developing management plan. Resident Canada geese (*Branta canadensis maxima*) have increased steadily throughout the Atlantic flyway and have become a nuisance in some parts of their range. The objective of our study was to determine dispersal, movements, and habitat use during the post-breeding period of resident Canada geese recently established in southern Quebec. The ultimate goal was to determine the potential of hunting as a measure to prevent this population of becoming over-abundant. We tracked a sample of geese fitted with radio-collars or conventional alphanumeric collars throughout the fall in 3 zones characterized by different habitats and hunting pressure. Before the hunting season, geese left the breeding area where hunting was allowed to reach a suburban environment where firearm discharge was totally or partially restricted. This post-breeding dispersal occurred when juveniles were approximately 3 months old. Distribution of radio-collared geese and flock surveys showed that few birds used the hunting zone. Local movements among zones with different hunting pressure were limited once migrant geese from northern breeding populations reached the study area. Radio-collared geese used mainly natural habitats ($75.4 \pm 2.6\%$), followed by urban ($14.4 \pm 2.7\%$) and agricultural habitats ($10.3 \pm 0.8\%$). They were located in $73.8\% \pm 6.2\%$ of the time in areas where hunting was prohibited. Geese that attended juveniles were more prone to use areas where firearm discharge was restricted than geese that have abandoned or lost their brood. This study showed that under the prevailing regulations, the potential role of hunting to manage resident Canada geese in southern Quebec is limited.

Keywords: *Branta Canadensis maxima*, Canada goose, habitat use, hunting disturbance, movement, Quebec, suburb ecology

Introduction

The choice of foraging habitats by animals is assumed to be an active process with the goal of maximizing their net rate of energy intake (Stephens and Krebs 1986). Individuals face trade-offs between predation risks and foraging opportunities (Lima and Dill 1990). They should thus adopt optimal strategies that maintain energy balance and reduce probability of mortality. In refuge systems, where groups of individuals disperse radially from a central living place to acquire food (Hamilton and Watt 1970), mobile populations have the option of moving to a more favorable environment when foraging costs become greater than gains (Frederick and Klaas 1982). Behaviors of individuals that directly affect vital parameters such as survival may thus influence population status. For that matter, movements of individuals in reaction to sport harvest may enhance survival and be a leading feature in population dynamics (Madsen 1995).

Over the past 40 years, waterfowl managers have favored hunting to control the growth of several goose populations in North America (Ankney 1996; Johnson 1997). In the Atlantic flyway, the demographic expansion of resident Canada geese (*Branta canadensis maxima*) has been recently stabilized, in part because of higher harvest resulting from special hunting seasons in several states and provinces (Atlantic Flyway Council 1999; Heusmann 1999; Hindman et al. 2003). However, the ability of this subspecies to exploit urban environments increases conflicts with humans and may prevent achievement of management goals because hunting activities are often prohibited or restricted around cities (Conover and Chasko 1985; Ankney 1996). It is thus essential to investigate the role of hunting on movements and habitat use by resident geese in these environments to refine control strategies and to help managers in taking decisions.

Resident Canada geese have expanded their range in northeastern America and are now breeding in southern Quebec (Giroux et al. 2001). The rapid growth of a population recently established near Montreal emphasizes the importance of collecting information to develop strategies that would aim at limiting population growth and minimizing conflicts experienced in other regions. In this study, we examined the post-breeding dispersal, fall distribution, local movements, and habitat use of resident Canada geese established in Quebec. The area is characterized by urban, suburban, and rural settings where different hunting regulations and

municipal by-laws about firearm discharge prevail. In addition, a special early hunting season has been introduced few years ago by the Canadian Wildlife Service to specifically control resident Canada geese. We took advantage of this spatiotemporal variation in hunting activity to evaluate its effect on the post-breeding dispersal of geese. Ultimately, we wanted to assess the potential of sport hunting as a control measure for this increasing population.

Methods

Study area

The study area encompassed approximately 415 km² within the region of urban sprawl that occurred in the last 20 years around Montreal (Fig. 1). It was characterized by urban and suburban developments encroaching into agricultural lands. We divided the area into 3 zones according to hunting activity and land use. The East zone included the nesting, rearing, and molting sites of Canada geese that breed at Varennes (Giroux et al. 2001). It was located northeast of Montreal along the St-Lawrence River (45°39'30" N, 73°27'30" W) and included several islands that were used as pasture until the end of the 1990's. This has limited the growth of trees and shrubs and maintained an extensive herbaceous cover (Lapointe et al. 2000). The surrounding mainland, especially along the south shore of the River was composed of 40% of agricultural lands dominated by small grain cereal and corn fields, the latter being used to feed livestock. Waterfowl hunting was allowed throughout the area except in the urban portions of Montreal, Repentigny, Varennes, and Boucherville and started on the fourth Saturday of September until mid December. The special early goose season was limited to agricultural lands and began 3 weeks before the regular season.

The Central zone was located at the junction of des Prairies and des Mille Îles Rivers (Fig. 1). Agricultural lands represented <30% of this zone and fields were mainly cultivated to produce human food (sweet corn and vegetables). Hunting was allowed along the south shores of des Prairies River on the island of Montreal and in all agricultural lands within the municipality of Lachenaie. Firearm discharge was prohibited elsewhere including the entire Ile Jesus (Laval) by municipal by-laws.

The West zone was characterized by extensive urban developments with high human density. However, the Mille Îles River that crossed the zone was dotted with many islands that constituted natural habitats. Agricultural lands were scattered on Ile Jesus and in the most

northerly portion of the zone. Hunting was not allowed throughout the West zone because municipal by-laws prohibited discharge of firearms.

Canada geese that breed in southern Quebec winter in north-eastern United States (J. Rodrigue, unpubl. data). Cold temperatures and freeze-up of most water bodies in winter prevent the geese to stay all year round. The geese arrive on their breeding sites at the end of March but their departure date was unknown. In addition, molt migrants from other southern resident populations join the breeding flocks during the summer. They arrive in the first week of June (Giroux et al. 2001) but their departure date was also unknown. Other molt migrants pass through the area in June and continue to more northern sites, up to Ungava peninsula (Rodrigue and Harvey 2002). During the fall, a portion of Canada geese (*B. c. interior*) that breed in Nunavik (northern Quebec) stage in southern Quebec (Malecki et al. 2001) along with returning molt migrant giant Canada geese.

Marking and tracking geese

In the spring 2004, the 4 islands of Varennes (111.5 ha) were searched for Canada goose nests. Using a scoop net or a bow-net trap, we captured a sample of nesting females and equipped them with radio-transmitters affixed to neck collars (total weight: 56 ± 0.8 g), which represented $<1.5\%$ of body weight (Demers et al. 2003). Nests were monitored until hatch to determine success and brood size of each marked female. In early July 2003 and 2004, we captured pre-fledged juveniles and molting adults using corral traps. Conventional alphanumeric plastic neck collars were put on a sample of adult females that had a brood patch. However, the presence of a brood patch does not mean that a female had bred in southern Quebec. Molt migrant geese that were failed breeders from other resident populations could also have a brood patch.

Starting in mid-July, when the geese were still molting, we located the radio-marked geese every day by telemetry using a Yagi antenna mounted on a vehicle or from a boat. When geese started to disperse, we searched each zone twice a week. Survey effort among zones was balanced between morning and afternoon periods. Tracking ended when the last radio-tagged goose left the area. Radio-collared geese were assigned to a zone on a weekly basis. Whenever, a bird was located in 2 zones during the same week, we associated it to the zone where it had not been observed the previous weeks to maximize information about

distribution. Residence time of each radio-collared goose during the hunting season was the number of days that elapsed between the opening of the special hunting season and the date of last location in the study area.

Radio-tagged geese were located by triangulation or visual contact and the locations were noted on 1:20,000 topographical maps. For each location, we recorded habitat according to 3 broad categories. Natural habitats were mainly riversides and islands that have not been modified by humans. Urban habitat included man-made habitats such as parks, golf courses, ponds, and riversides that had been transformed by human activities. Finally, agricultural habitats included fields with different crops. We also noted whether the geese were located in areas where hunting or firearm discharge was allowed or not.

During the surveys, we also searched for neck-collared females with a spotting scope (25-60 x) in all flocks regardless of the presence of radio-tagged females. We recorded the number of individuals in each flock and computed the sum for each week and zone. Finally, groups of hunters spotted during the surveys were recorded, summed for the whole season and standardized by the number of days with surveys in each zone.

Statistical analyses

Survival and movement probabilities of collared geese were estimated using multi-state capture-recapture models with program MARK version 4.1 (Brownie et al. 1993; White and Burnham 1999). For multi-state models, capture histories reveal both encounters and locations (state). In this study, the first capture consisted of observations of neck-collared geese during the molting period and subsequent captures were the observations made during the following periods (see below). Maximum likelihood estimates were obtained for: p^r_t = the probability that an individual was detected in zone r at time t given that the individual was alive at time t , ϕ^r_t = the probability that an individual alive in zone r at time t survived and did not permanently emigrate from the study area until $t+1$ and ψ^{rs}_t = the probability that an individual in zone r at time t be in zone s at time $t+1$ given that the individual survived until $t+1$. We assumed that movement probabilities between t and $t+1$ followed a first-order Markovian process, i.e. they only depended on the region at time t .

To minimize the number of parameters to be included in the models, we grouped recaptures into 8 3-week periods. July 24 was set as the end of the molting period (M) based

on the hatching date of the first nest and a 70-day fledging period (Hanson 1965). Two post molting periods (PM1 and PM2) were set before the beginning of the special hunting season (SS) that spanned between 6 and 24 September. The regular hunting season opened on 25 September and 4 3-week periods were established (RS1, RS2, RS3, and RS4). A bird seen in more than 1 zone for a given period was associated to the zone where it was most often encountered. When ties occurred, the bird was associated to the zone where it has not been encountered during the previous period to maximize information about movements. We considered the probability of collar loss to be null during the study because of its short duration.

The most general model including ϕ , p , and ψ as function of time ($t=7$), zone ($z=3$) and collar types ($c=2$) was coded $\phi_{z \times t \times c}$, $p_{z \times t \times c}$, and $\psi_{z \times t \times c}$, respectively. Geese tagged with conventional and radio collars were coded distinctively giving full encounter probability ($p' = 1$) of radio-collared birds. To reduce the number of parameters, we tested models without group constraints on ϕ and ψ and models in which p was function or not of t and/or s . Our main interest was to investigate the effect of hunting on ϕ and ψ using models that had biological significance. Numerous models were computed using constraints relevant to the hunting regulations that changed among periods and zones.

Goodness-of-fit tests were calculated using the program U-Care 2.02 (Lebreton 2003), which does not separate multi-state tests but give associated statistic values. We used the Akaike's Information Criterion (AIC_c) to select the best approximating model (lowest AIC_c value; Burnham and Anderson 1998). We also used AIC_c weight ($AIC_c \omega$), which represents the weight of evidence in support of each model in the candidate set given the data.

We used compositional analyses to investigate how habitat use by radio-collared geese was affected by hunting (Acbischer et al. 1993). Availability was defined as the proportion of each habitat (natural, urban, and agricultural) within the entire study area. This was established from a digitized Landsat thematic map (TM, 30-m pixel) using ArcMap 9.1 for Windows (ERSI 2005). We excluded forests from natural habitat and urban settlements from the urban habitat because geese do not use these habitats. We also calculated the areas where hunting was allowed and prohibited within the natural and agricultural habitats (hunting being always prohibited in urban habitats). Habitat use was defined as the proportion of radio locations in each habitat for each individual. To satisfy unit-sum constraint and independency

conditions, the sampling unit was the log-ratio of the proportion of locations of each individual (Aebischer et al. 1993).

We first tested that habitat use differed significantly from random use before and during the hunting season. For each period, we ranked habitats from the most to the least exploited by geese and tested the significance of the associated Wilk's lambda (λ) by randomization as recommended by Aebischer et al. (1993). We then used a MANOVA to investigate whether habitat use by radio-collared geese changed after the opening of the hunting season. Our second objective was to detect habitat preference during the hunting season by considering five habitat categories (natural with and without hunting, agricultural with and without hunting and urban without hunting) in a compositional analysis. We also used randomization tests to determine the significant differences between ranks of the habitats used by geese. We used a paired *t*-test to compare the percentage of locations of individuals in hunted areas between the natural and agricultural habitats to evaluate whether geese were more susceptible to encounter hunters when using one of these habitats. Finally, a compositional analysis was used to compare habitat use by females that attended a brood and those that either abandoned or lost their broods. A Student *t*-test was also used to compare the percentage of locations in areas where hunting was prohibited between attending and non-attending females. All compositional analyses and randomization tests were performed with a modified SAS program written by Ott and Hovey (http://nhsbig.inhs.uiuc.edu/habitat_use/bycomp.sas). Angular transformations were applied to percentage data.

Results

In 2004, 134 nests were located on the Varennes islands and we captured 19 females that were fitted with a radio-collar. Four females left the study area before the molting period (between June 6 and June 12) and 3 returned to the study area during the fall. The fourth one was shot in Massachusetts in November. Among these 4 females, one had its nest preyed upon and the others abandoned their young early during the brood-rearing period, a common phenomenon in this population (Doiron 2006). Young that had been web-tagged at hatching were captured in July in banding drives that did not include the parents (J-F Giroux, unpubl. data). Because we did not succeed in putting all radios during the nesting period, we replaced

conventional collars by radio collars of 6 females during the banding drives. These females had been previously located on a nest (all were successful in hatching eggs) and observed regularly during the brood rearing and molting periods. Among the 21 radio-collared females that molted in the study area, 1 had its nest preyed upon, 12 reared ≥ 1 young and 8 abandoned or lost their brood. A total of 1044 locations were obtained including 508 visual contacts (48.7%) with an average of 47.1 locations per female (SE = 1.5; min = 29; max = 58) and a mean interval of 3.3 days between each location.

In 2003 and 2004, we banded 288 and 544 molting adults and fitted 91 and 124 females with conventional collars, respectively. We observed 150 females during the fall 2004 for a total of 1727 locations or 13.3 per female (SE = 0.46; min = 1; max = 36).

During the hunting season, we recorded seven times more hunting parties in the East (0.34/day) than in the Central zone (0.05/day). As expected, no hunter was observed in the West zone throughout the fall. Although we could not distinguish between those that were hunting geese and ducks, most waterfowl hunters in southern Quebec will shoot at geese if they have the opportunity. During our surveys, we did not encounter any hunter during the special early season but we know that some hunters were active. In 2004, 7 birds banded in 2003-2004 were recovered during the special season and 33 during the regular season within the study area, mostly in the East zone (92.5%).

Dispersion and migration chronology

During the molt and the beginning of the post-molt periods, Canada geese were concentrated in the East zone (Fig. 2a). These included birds that bred at Varennes and those that arrived there during the summer to molt. The 21 remaining radio-collared geese were all located in this zone (Fig. 2b). Their numbers decreased rapidly by mid-August when the birds dispersed towards the Central and West zones. This occurred when the juveniles were approximately 3 months old (Table 1), well before the opening of the special hunting season in early September.

Concomitant to the dispersal of resident geese to the West and Central zones, arrival of *B. c. interior* and *B. c. maxima* from northern Quebec increased the total number of birds in these two zones (Fig. 2a). The regular hunting season began at this time and the maximum number of birds was recorded about 3 weeks later. Very few geese used the East zone during

the hunting period compared to numbers observed in the Central and West zones. Finally, a greater number of geese including radio-tagged birds used the West than the Central zone during the second half of the hunting season (RS3 and RS4). Only 1 radio-tagged goose was shot during the study and this occurred during RS1 in the East zone. This goose had previously spent the PM1, PM2, and SS periods in the Central zone.

The chronology of migration departure was much extended than the post-molt dispersal (Table 1). The first radio-tagged geese left the area in early October and the last one on 19 December. On the other hand, the post-molt dispersal occurred during a period of just over 1 month. These birds were not located further south in Quebec during sporadic ground tracking throughout the area. Residence time during the hunting season lasts on average 88 days, which represents 78% of the duration of the whole hunting season (112 days)

Movements

When the data on radio and conventional collar locations were grouped into 8 periods of 3 weeks, 1026 observations were considered including 370 in the East, 357 in the Central, and 299 in the West zones. For radio collared birds, we are confident that the regrouped encounters were representative of each zone use because individuals were located on average 91.7% of time in the same zone during a given period. Although ties in the number of locations in more than one zone never occurred, individuals were located in three zones during a same period at only nine occasions. For conventional collared geese, 85.4% of regrouped encounters occurred in a single zone, 14.6% in 2 zones and none in 3 zones. Moreover, ties in the number of locations for 2 zones occurred in only 5.9% of the regrouped encounters.

The general model fitted the data well ($\chi^2_{102} = 89.3$; $p > 0.05$). The two best models indicated that detection probability (p) was time and group dependent while apparent survival probability (ϕ) and movement probability (ψ) were related to hunting regulations (Table 2). Time constraints on ϕ indicated different survival probability after the opening of the hunting season ($\phi_{zh}: \phi_{1-2-3}^r \neq \phi_{4-5-6-7}^r$) whereas a weak preference was shown between the model with full zone constraint on ϕ ($\phi_i: \phi_i^{East} \neq \phi_i^{Central} \neq \phi_i^{West}$) compared to one with hunting zone constraint on ϕ ($\phi_{th}: \phi_i^{East} \neq \phi_i^{Central-West}$). This last model estimated that ϕ for the East zone was 98.5% (SE = 0.01) before the hunting season and 64.5% (SE = 0.16) during the hunting

season while ϕ for the Central and the West zone together was 99.9% (SE = 0.01) and 96.1% (SE = 0.01) before and during the hunting season, respectively.

The best model suggested that constraint on ψ due to hunting regulations implied only movements toward the East zone and suggested that ψ was constant throughout the study until the last period ($\psi_{zh}: \psi_{1-2-3-4-5-6} \neq \psi_7$ for $\psi^{Central \rightarrow East}$ and $\psi^{West \rightarrow East}$) whereas no constraint on other movements was selected. Estimates of ψ indicated increasing movements to the Central and West zones before the opening of the regular hunting season (Fig. 3a). Thereafter, ψ remained low except for a small increase during RS3 indicating some movements from the Central to the West zone. When the opposite transitions were considered, movements from the West to the Central zone occurred at the beginning of the hunting season. Movements towards the East zone were negligible during the whole period except during RS4 when some geese returned to their breeding and molting sites before departing for migration (Fig. 3b).

Habitat use

During the fall, radio-collared geese were predominantly located in natural habitats ($75.4 \pm 2.6\%$), followed by urban ($14.4 \pm 2.7\%$) and agricultural habitats ($10.3 \pm 0.8\%$). Geese using natural habitats were observed resting on des Mille Îles and des Prairies Rivers and were often feeding in shallow water dominated by submerged vegetation along the shores of the rivers or around the numerous islands that dotted these rivers. Urban habitats used by geese included lawns on private properties and city parks located along riversides, corporate managed landscapes surrounding petro-chemical plants, and golf courses. In agricultural habitats, geese concentrate their feeding on spilled grains in stubble and plowed corn fields or small grain cereals. Harvest of corn fields in the Central zone (sweet corn for human use) started in mid August and was completed by late September. In the East zone, harvest of corn for grain started in late September and ended in December.

Compositional analysis showed a significant departure from random use of habitats during the fall both before ($n = 21$; Wilk's $\lambda = 0.332$; randomized $P < 0.001$) and during the hunting period ($n = 24$; Wilk's $\lambda = 0.353$; randomized $P < 0.001$). Natural and urban habitats were both preferred over agricultural lands during the two periods. However, habitat use by geese slightly changed after the opening of the hunting season with geese spending more time

in agricultural habitats (4.7 vs. 15.0%) and less in urban habitats (20.9 vs. 9.2%; MANOVA, $F_{2,38} = 5.49$; $P = 0.003$).

During the hunting season, radio-collared geese were located mainly in areas where hunting was prohibited ($73.2\% \pm 6.2\%$). For this period, significant departure from a random use was observed ($n = 24$; Wilk's $\lambda = 0.096$; randomized $P < 0.001$; Fig 4a). Urban habitats and non-hunting areas in natural habitats were the most preferred while agricultural lands and areas accessible to hunters in natural habitats were the least preferred. Radio-tagged geese were more prone to be in a hunting area when using agricultural ($40.6 \pm 8.8\%$) than natural habitats ($26.6 \pm 6.3\%$, $t_{23} = 2.18$; $P = 0.020$).

Geese that had attended a brood during the rearing period and that were most likely still accompanied by juveniles after the post-breeding dispersal did not use habitats randomly during the hunting season ($n = 12$; Wilk's $\lambda = 0.053$; randomized $P < 0.001$). They preferentially used no hunting areas in natural and urban habitats (Fig. 4b). Females that had abandoned or lost their brood also showed preferential use of some habitats during the hunting season ($n = 12$; Wilk's $\lambda = 0.116$; randomized $P < 0.001$). Although they greatly preferred no hunting areas in natural and urban habitats, they also used hunted areas in natural habitats to a greater extent than their availability (Fig. 4c). Attending females were more often located in areas where hunting was prohibited than non-attending ones ($82.0 \pm 7.9\%$ vs. $64.8 \pm 9.2\%$; $t_{23} = 2.54$; $P = 0.019$).

Discussion

In this study, we showed the limited potential role of hunting to manage the resident Canada geese established in southern Quebec. Dispersal toward a suburban environment reduced the risk of mortality of individuals by limiting the effect of hunting. In our study area, hunting pressure was higher around the main breeding site than few km away where municipal by-laws restricted firearm discharge. The Varennes islands are a popular duck hunting area and few banded geese were recovered by hunters from this area despite its limited use by geese during the fall. Nevertheless, the intensity of goose hunting in the whole area was limited as shown by our index of hunter's encounter. Our results confirm the contention expressed by the Atlantic Flyway Council (1999) that harvest of Canada geese in urban and suburban areas is generally limited.

Strong behavioral response to hunting disturbance has been shown in many species of waterfowl (Madsen 1998; Madsen 2001; Béchet et al. 2003; Bregnballe et al. 2004). Madsen (1998) found that shooting was the most disturbing human activities that affect distribution of waterfowl in Denmark. In migrating snow geese (*Anser caerulescens atlanticus*), an increased of backward movements was observed after the implementation of a spring conservation hunt in Quebec (Béchet et al. 2003). This supports the idea that birds have the ability to assess the risk of being preyed upon or shot and that they can incorporate this information into their decision making (Lima and Dill 1990; Blumstein and Bouskila 1996). In this study, however, the shift by resident Canada geese from a relatively high to a low hunting pressure area occurred several weeks before the opening of the hunting season indicating no direct effect of hunting. Individuals may thus use prior knowledge of pay-offs acquired from early experience and tradition (McNamara et al. 2006; Valone 2006).

On the other hand, changes in resource availability may also influence the distribution of birds in fall. Compared to the high hunting pressure area where corn is harvested later, sweet corn in the low hunting pressure area is collected in early fall providing some resources for geese before the opening of the hunting season. After the migration peak, few movements occurred and geese stayed in areas where hunting pressure was low. Movements took place toward the high hunting pressure area at the end of the hunting season, which coincided with improved feeding opportunities in stubble and ploughed corn fields and with the presence of fewer hunters.

Several goose species feed on crops (Frederick and Klaas 1982; Alisaukas 1988; Hill and Frederick 1997) and significance of this resource for population dynamics have been clearly established (Shimada 2002; Gauthier et al. 2005; Tombre et al. 2005). However, individuals in our study were more likely to be in contact with hunters when using agricultural than natural habitats. Geese were often observed feeding in marshes and shallow water in natural habitats and this occurred throughout the day. The decision of geese to minimize their predation risk (hunters' encounters) might have consequences on individual foraging opportunities.

Herbivory in geese implies a high proportion of time devoted to feeding (Frederick and Klaas 1982; Giroux and Bedard 1990; Ely 1992; Gawlik and Slack 1996). Although natural habitats provide green vegetation that is less digestible than spilled grains (McDonald 1995),

selective pressure may not require a high rate of energy intake by resident geese at this stage of their annual cycle. These large birds may be able to maintain body reserves during a long staging period. Moreover, the distance to the wintering grounds in northeastern United States is relatively short (≈ 250 -400 km) and contrary to the prenuptial period, energy reserves may not need to be maintained at the maximum level during winter (Witter and Cuthill 1993).

Undisturbed feeding sites providing abundant food can also be found in urban habitats (Conover and Kania 1991). Although nuisance problems in southern Quebec are currently limited (J. Rodrigue, unpubl. data), the preferential use of this habitat may generate increased conflicts with human interests considering that the current limited effect of hunting is not likely to restrain further increase of the population size. Moreover, considering that hatching year birds are more vulnerable (Chapman et al. 1969; Grieb 1970), the use of areas less accessible to hunters by females accompanied by juveniles may reduce hunting success, improve juvenile survival, and ultimately increase recruitment.

Management plans should be developed to reduce potential human-geese conflicts while preserving optimal value of this resource. It has been demonstrated that sport harvest can sometimes be efficient in controlling populations when specific regulations are established at a regional scale (Bregnballe et al. 2004). We have shown that residence time of resident Canada geese during the hunting period was long and extended until late fall, which should provide opportunities for some harvest. However, urban sprawl has affected hunting activities in suburbs, which currently limits the use of hunting as a management measure. Innovative regulations should be implemented to liberalize firearm discharge in well delimited areas and during restricted periods to increase the opportunities of harvesting resident Canada geese. As an example, high harvest rate was favored in Massachusetts during special hunting seasons in traditional sites as well as in atypical sites such as golf courses (Heusmann 1999). Waterfowlers' participation to the special early season for Canada geese in southern Quebec is currently unknown and their contribution to the harvest of resident birds needs to be estimated.

Conversely, we showed that migrant Canada geese were often mixed with resident birds when staging in southern Quebec. Precaution must thus be taken to protect less abundant or more vulnerable populations when drafting management plans (Hindman and Ferrigno 1990; Atlantic Flyway Council 1999). Difference in hunting vulnerability between

migrant and resident geese is likely to vary with regions, time of year, and the period that geese remained in the area (Leafloor et al. 1996; Lindberg and Malecki 1994; Schultz et al. 1988). We do not know whether migrant Canada geese and resident geese returning from their molt migration use the same habitats than the local resident geese tracked in this study. Resident geese that breed or molt in southern Quebec reached the low hunting pressure area before the arrival of northern populations and this may affect site selection by late arriving birds. While breeders from Ungava peninsula leave their summer sites in late September (Malecki et al. 2001), migration chronology of northern molting resident geese is unknown. The use of morphological and isotope analyses to establish the proportion of different populations in the harvest could help to refine management plans (Beaumont 2006). In addition, alternative control strategies that consider the behavioral ecology of the species must be developed and evaluated in relation with their effects on population dynamics.

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Table 1. Hatching, molting, and migration chronology of radio-collared resident Canada geese marked near Montreal, Quebec, 2004.

Event	n	Median date	Range
Hatching	23	20 May	11 May – 12 June
Post-molting dispersal	21	16 August	7 August – 11 September
Migration departure	23	3 December	9 October – 19 December

Table 2. Model selection of regional movements of resident Canada geese near Montreal, Quebec, 2004. ΔAIC_c , $AIC_c\omega$, and number of parameters (k) of the multi-state models are presented.

Model	k	ΔAIC_c	$AIC_c\omega$
$\phi_{z*th} p_{t*c} \psi_{zh*t}$	54	0	0.56
$\phi_{zh*th} p_{t*c} \psi_{zh*t}$	52	0.49	0.44

Model notation: ϕ = apparent survival; p = detection probability; ψ = movement probability; z = zone (East, Central, West); t = time; c = collar type; h refers to models where ϕ or ψ is time or spatial dependant in relation to hunting (See results).

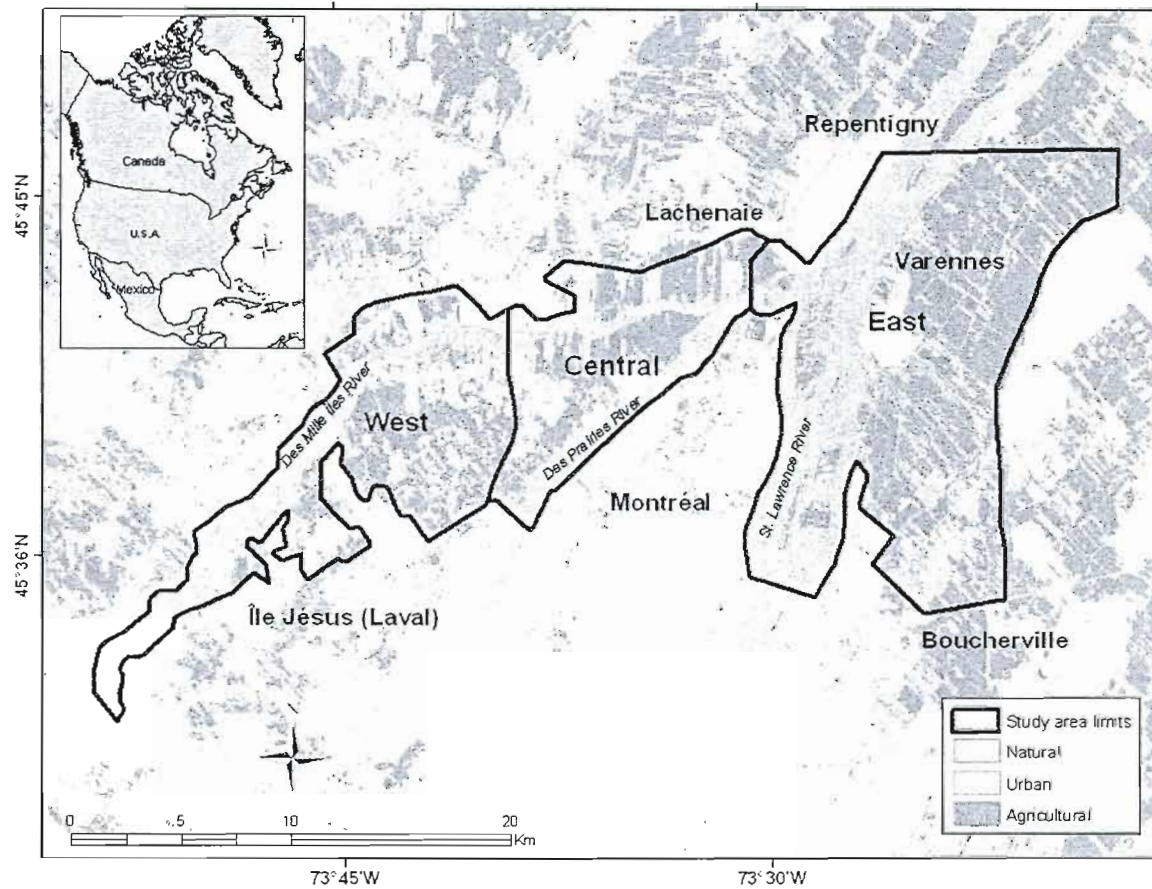


Fig. 1. Location of the West, Central and East zones where Canada geese were surveyed in the fall 2004 near Montreal, Quebec.

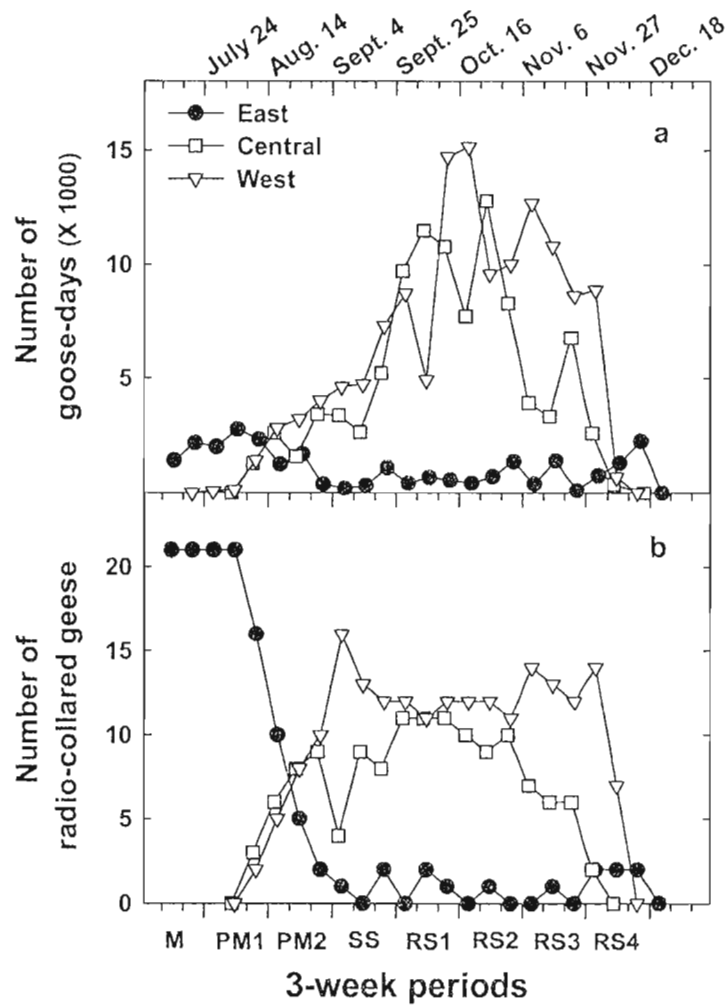


Fig. 2. a) Weekly counts of Canada geese and b) number of radio-collared resident Canada geese located each week in the West, Central, and East zones near Montreal, Quebec, 2004. See the Methods section for description of the periods.

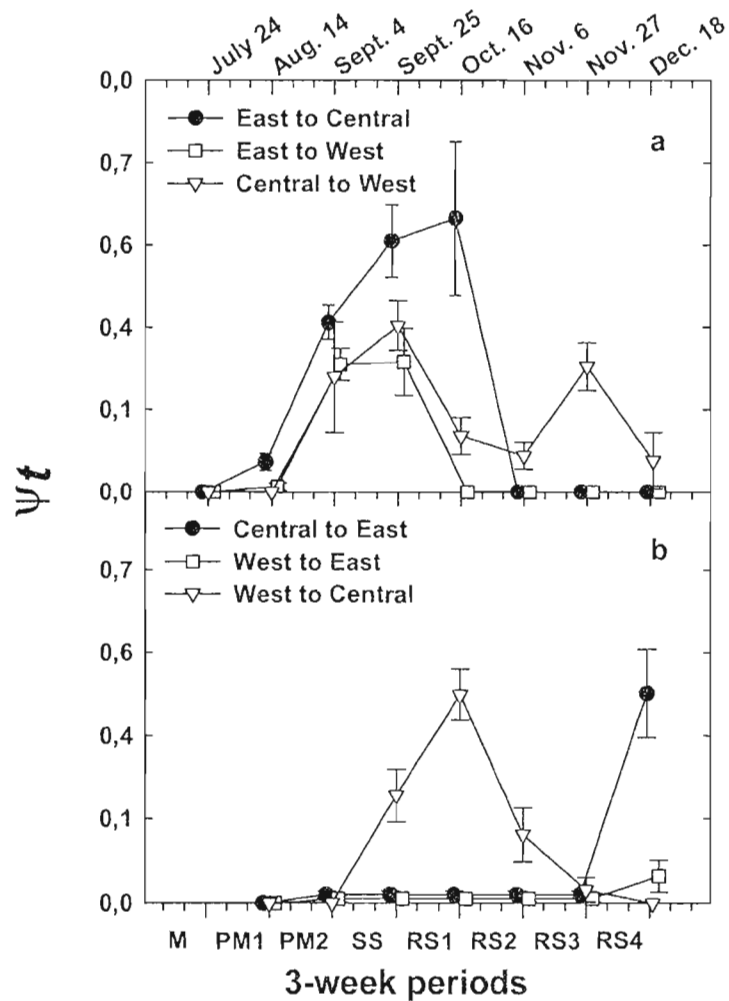


Fig. 3. Estimates (± 1 SE) of movement probability (ψ) of neck-collared Canada geese near Montreal, Quebec during 8 3-weeks period in 2004 along a) a decreasing and b) increasing hunting pressure gradient. Estimates are derived from the best model selected with AIC. See the Methods section for description of the periods.

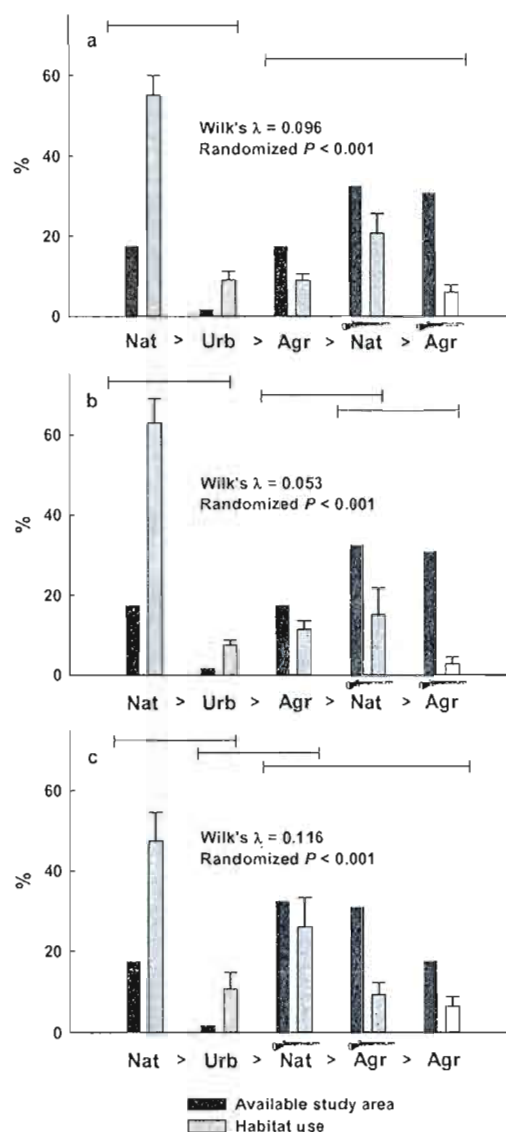


Fig. 4. Habitat use of radio-collared resident Canada geese during the hunting season near Montreal, Quebec, 2004 (a = all radio-collared, $n = 24$; b = attending female, $n = 12$; c = non-attending female, $n = 12$). Habitat to the left of the symbol > are selected proportionally over those to the right and non-significant differences between habitat is indicated by a connected line ($P > 0.05$, compositional analysis). Significant departure from random use is indicated by λ and randomized P -values. Nat = natural, Urb = urban, Agr = agricultural, firearm symbols represent hunting areas in natural and agricultural habitats.

CONCLUSION GÉNÉRALE

Afin de raffiner la gestion de la bernache du Canada dans la voie migratoire de l'Atlantique, nous avons produit une méthode d'identification de populations en utilisant la morphométrie ainsi que les isotopes stables. Dans un premier temps, il est possible d'identifier la population source de bernaches capturées sur les sites estivaux. Puisque les statuts sont établis à partir de ces populations, il est essentiel de bien caractériser les oiseaux lors des opérations de baguage. Dans un deuxième temps, il sera possible d'estimer la contribution spatio-temporelle de chacune des populations à la récolte automnale. Ultimement, les modalités de chasse pourront être optimisées afin d'atteindre les objectifs fixés par les gestionnaires.

Nous avons également documenté la distribution post-reproduction de bernaches qui nichent dans le sud du Québec. L'ensemble des individus qui nichent sur les îles de Varennes utilisent une zone refuge afin de se soustraire aux activités de chasse. Nous avons ainsi montré le rôle potentiel limité des activités cynégétiques afin de contrôler la croissance de cette population. Nous avons également quantifié l'utilisation des habitats utilisés par les individus et ce, en relation avec la chasse. Afin de limiter les problèmes vécus dans d'autres régions de la voie migratoire, nous suggérons que la chasse demeure un outil privilégié mais que des stratégies alternatives soient adoptées.

Gestion de la bernache du Canada dans la voie migratoire de l'Atlantique

Au cours des années 1990, la gestion de la bernache du Canada a subi des modifications importantes. Puisqu'il subsiste une hétérogénéité de populations sur les sites d'hivernage, les inventaires sont maintenant effectués sur les sites de nidification afin d'établir le statut de ces populations avec davantage de précision (Hindman *et al.* 2003). Nos travaux ont par contre mis en évidence la présence de migrateurs de mue de la sous-espèce résidente (*B. c. maxima*) sur les sites de nidification de la sous-espèce migratrice (*B. c. interior*). En comparant la morphométrie de différents échantillons d'oiseaux capturés dans la péninsule d'Ungava (référant à la sous-espèce migratrice), nous avons pu révéler une contamination de ces échantillons par la présence de bernaches résidentes en mue. Chez la bernache du Canada, le phénomène de migration de mue d'individus non-nicheurs est traditionnellement observé à

travers l'Amérique du Nord (Abraham *et al.* 1999; Davis 1985; Hanson 1965; Krohn 1979; Kuyt 1962; Lawrence 1998; Salomonsen 1968; Sterling 1967; Tacha 1991; Zicus 1981a, 1981b). Cependant, dans la péninsule d'Ungava, l'ampleur actuelle de ce phénomène n'a pas été étudiée. Étant donné que dans le sud de la voie migratoire, les conflits causés par la surabondance des bernaches résidentes engendrent une intensification des programmes de destruction des nids, le phénomène de migration de muc deviendra de plus en plus important. En plus de complexifier l'aménagement des populations subarctiques, l'augmentation de la présence de migrateurs de muc dans la péninsule d'Ungava peut résulter en un accroissement du niveau de compétition entre ces populations sympatriques (Abraham *et al.* 1999). Advenant une modification des taux de survie ou de recrutement découlant de ce contexte écologique particulier, la méthode de caractérisation des populations que nous avons développée améliorera l'estimation des différents paramètres démographiques.

Lors de la migration d'automne, les chasseurs du sud du Québec récoltent à la fois des bernaches migratrices en provenance de la péninsule d'Ungava et des bernaches résidentes qui ont niché dans la région ou qui ont mué plus au nord. Bien que ces différentes populations possèdent des statuts distincts, leurs taux de récolte par la chasse demeurent inconnus. Il sera maintenant possible de mieux estimer ces paramètres en utilisant les modèles que nous avons produits. Avec une enquête aléatoire et représentative de l'ensemble de la province, une analyse de la récolte serait réalisable de façon spatiale en considérant les districts de chasse comme unité d'échantillonnage. Dans la voie migratoire du Pacifique, certains états ont réparti des stations d'échantillonnage sur l'ensemble de leur territoire afin de recueillir des données morphométrique sur les oiseaux récoltés par les chasseurs (Subcommittee on Pacific population of western Canada geese 2000). Selon les patrons observés et les statuts des populations, les gestionnaires peuvent, de ce fait, mieux gérer la récolte à l'échelle régionale.

Nous avons montré que l'emploi de la morphométrie est sujet à divers biais; erreur de mesure et identification du sexe erronée. Pour ce qui est des ratios isotopiques, nous avons observé une hétérogénéité des valeurs moyennes pour différents sites d'échantillonnage situés à des latitudes semblables (*voir* Appendice I). Bien qu'il subsiste des variations dans les signatures isotopiques des plumes de bernaches dans la voie migratoire, les modèles ont été élaborés à l'aide d'échantillons représentatifs des régions où l'on retrouvait les plus fortes

densités d'individus. Néanmoins, afin d'améliorer le contrôle des taux de récolte ainsi que son influence sur la dynamique des populations, des politiques de gestion adaptatives permettraient de développer des modèles considérant l'effet des variations de l'environnement (Johnson 1999).

Stratégie pour le sud du Québec

Nos résultats montrent que les bernaches résidentes des îles de Varennes séjournent longtemps dans la région durant la chasse mais qu'elles sont concentrées dans les zones où la décharge d'armes à feu est interdite. L'évitement des zones de chasse par les individus est un comportement d'anti-prédation qui réduit les taux de mortalité et influence ainsi la dynamique de la population. L'étude du comportement des individus et des populations met donc en évidence l'importance des variables qui influencent les paramètres démographiques.

Sutherland (1996) suggère de lier le comportement animal à la dynamique des populations en développant des modèles basés sur la distribution idéale libre (Fretwell 1970). Il s'agit de mettre en relation le comportement des individus et les effets de densité-dépendance dans le but de les associer à la démographie; les réponses observées pouvant également être combinées à des caractères écologiques, physiologiques ou génétiques. Afin de comprendre comment certaines variables affectent les paramètres démographiques, il est nécessaire de quantifier en terme de coûts et bénéfices les décisions prises par des individus identifiables (Lomnicki 1980, 1988). Dans le sud du Québec, de plus en plus de bernaches sont marquées permettant ainsi un examen plus approfondi des variations comportementales entre les individus. Pour contrôler la croissance de cette population en augmentant les taux de mortalité, l'emploi de la modélisation en lien avec les activités cynégétiques s'avèrerait un outil novateur.

Subséquentement, une gestion intermittente et spatio-temporelle des activités de chasse pourrait être établie (*refuge design*; Bregnballe *et al.* 2004). Des modifications aux modalités de chasse pourraient être apportées dans le but d'influencer les patrons de mouvement des oiseaux et de favoriser le succès des chasseurs. Bien qu'une gestion encadrée de la chasse à l'échelle de la région métropolitaine de Montréal pourrait être envisagée, il serait complexe de permettre cette activité dans des endroits où elle n'est pas traditionnellement pratiquée ou dans des quartiers où elle a récemment été bannie; la chasse en milieu périurbain demeurant

une source de conflit fonctionnel et social (Fédération québécoise de la faune 2001). Des stratégies alternatives devront donc être adoptées et il est nécessaire de sensibiliser la communauté à la problématique de l'expansion de la bernache résidente. Pour ce faire, nous devons expliquer le phénomène ainsi que les conflits engendrés et les moyens dont nous disposons afin de réduire le risque que cette population devienne surabondante (Environnement Canada 2006a, 2006b). Et, au-delà des implications purement économiques, la communauté devrait être sensibilisée à la valeur philosophique et globale de la présence de cette faune afin de conserver son respect et son admiration.

APPENDICE I

SIGNATURES ISOTOPIQUES DES PLUMES PRIMAIRES RÉCOLTÉES SUR DES BERNACHES DU CANADA EN MUE

Dans le chapitre I, nous avons présenté les signature isotopiques de 8° primaires échantillonnées lors des opérations de baguage à Varennes ainsi que sur la côte Ouest et Est de la péninsule d'Ungava. Des échantillons additionnels ont également été récoltés à l'usine Dow Chemical (Québec, 45°43' N, 73°24' W) et à la Baie James (Québec, 52°17' N, 76°30' W). Ces échantillons ont été analysés conformément à la méthodologie décrite dans le chapitre I (*voir p 11*). Nos résultats montrent que les ratios isotopiques contenus dans les plumes des bernaches qui ont mué à la Dow Chemical et à la Baie James se distinguent de ceux des bernaches de Varennes ou de la péninsule d'Ungava.

Nous avons relevé une déplétion du ^{13}C pour les bernaches migratrices capturées à la Baie-James sur le réservoir hydroélectrique Opinaca (Fig. 1). Montgomery et al. (2000) ont rapporté une différence entre les valeurs de $\delta^{13}\text{C}$ trouvées dans le zooplancton des réservoirs par rapport à celui des lacs. Suite à l'augmentation des niveaux d'eau dans les réservoirs, il y a une augmentation de la matière en suspension et, conséquemment, un accroissement de la respiration microbienne (Kelly *et al.* 1997). Lors de la séquestration du carbone, les organismes autotrophes ont une préférence pour le ^{12}C , et le ratio isotopique de leurs tissus est plus faible que celui du CO_2 ambiant (Martincelli *et al.* 1991). La respiration étant, pour sa part, accompagnée d'un faible fractionnement isotopique, le CO_2 biogénique relâché est donc moins concentré en ^{13}C contribuant ainsi à la déplétion de cet isotope dans le milieu.

Dans le sud du Québec, des bernaches résidentes échantillonnées sur le terrain de l'usine Dow Chemical possédaient des signatures isotopiques distinctes de ce qui a été trouvé pour les bernaches de Varennes (Fig. 1). Les oiseaux utilisent des bassins recueillant l'eau issue des tours de refroidissement de l'usine pétrochimique (Raymond Paquin, comm. Pers., Dow Chemical). Lors de l'évaporation des molécules d'eau, les sels demeurent dans la tour et sont évacués par « Blowdown » ce qui rehausse les concentrations de sel dans les bassins et altère la biochimie de l'environnement (Almeida *et al.* 1989).

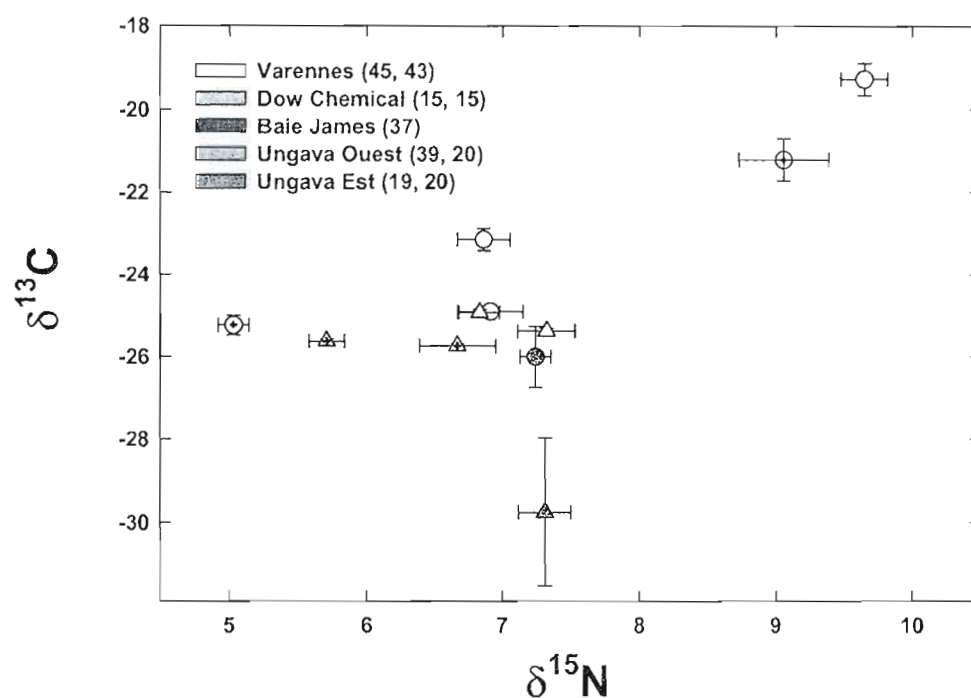


Fig. 1. Valeurs moyennes ($\pm$ SE) de $\delta^{15}\text{N}$ et $\delta^{13}\text{C}$ retrouvées dans les plumes primaires de bernaches du Canada échantillonnées sur les sites d'été de différentes populations en 2003 et 2004. Les cercles représentent les bernaches résidentes et les triangles, les bernaches migratrices. Les symboles pleins indiquent les adultes alors que les symboles avec une croix indiquent les juvéniles. Les tailles d'échantillon sont présentées entre parenthèse pour chaque site en ventilant les nombres d'adultes et juvéniles, respectivement.

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