Wildlife science inclusive of local priorities and knowledge co-production: moose habitat selection in the Adapted Forestry Regime of Eeyou Istchee, Northern Quebec

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ABSTRACT (ENGLISH)

Inclusion of Indigenous knowledge about wildlife populations and their habitats can inform wildlife research, while also increasing local engagement and support for wildlife conservation decisions. Boreal forest land use and forestry practices have direct and indirect impacts on ecosystems and Indigenous communities. Eeyou Istchee, the Cree traditional territory in Northern Quebec, Canada, includes areas that are significantly impacted by forestry activities. Concerns have been raised about the impact of these forestry activities on moose, a wildlife species that is vitally important to Cree culture and food security. The Adapted Forestry Regime (AFR) was enacted in 2002 to better integrate Cree concerns and community participation in forestry practices and management. Included within this regime was the identification of Sites of Special Wildlife Interest to the Cree (25% areas), where forestry would be specially managed to reduce negative impacts of logging on wildlife, including moose. Twenty years after implementation of the AFR, moose habitat quality has not been assessed. The objective of this thesis is to advance understanding about the inclusion of experiential wildlife knowledge into quantitative habitat analyses. I approach this objective by contributing a systematic review of the methods, successes, and limitations defining past attempts at experiential wildlife knowledge inclusion and through a case study evaluating the effects of an adapted forest regime on moose habitat selection informed by Cree knowledge.

Chapter 1 presents a systematic review of methods reported in peer-reviewed literature to interweave local, expert, and Indigenous knowledge into quantitative modeling in wildlife analyses. This kind of knowledge interweaving can help to increase applicability, trust, and equity in wildlife science and management while also potentially increasing accuracy and transferability. We reviewed 49 articles and reported on the methodologies employed in knowledge holder selection, their stages of involvement, knowledge elicitation, modeling processes, bias and uncertainty management, and validation. We conclude with six key identified benefits, limitations, and recommended improvements for future analyses that interweave knowledge into quantitative science.

Chapter 2 assesses moose habitat selection in the AFR informed by Cree expert knowledge retrieved from semi-structured interviews in the form of habitat relationships that were used to

determine the variables explored in the model; land cover, elevation, distance to water, road density, and 25% areas were chosen for analysis based on recurring topics brought up by Cree experts that aligned with available data. We performed home range analysis, Generalized Linear Model analysis to assess habitat selection, and Resource Selection Function analyses to assess how moose used habitat features relative to availability. We ran models for mid-summer and mid-winter from 2018 to 2021 for 38 female moose fitted with GPS collars. Moose selected for 25% areas (special interest sites with wildlife-focused logging management) in both seasons. In summer, moose selected small islands, thinned forests (regenerating stands after forestry disturbance that have had brush cutting recently performed), coniferous forest with fir, and flood zones, while in winter moose selected mixedwood and deciduous forest. In both seasons, moose selected midland and upland terrain while avoiding lowlands. Moose tended to use sites regenerating post-forestry either similarly to, or more than sites regenerating from natural disturbance, although selection was less than for preferred intact stands.

Through these analyses, I provide the first assessment of moose use of the 25% areas and quantify use of logged stands in the AFR, informed by and reflective of Cree Knowledge, highlighting the importance of a multi-season and multi-knowledge approach to assess the influence of an adapted forestry regime on the evolution of moose habitat quality. By illustrating how Cree knowledge can inform a quantitative analysis of moose habitat selection related to a local knowledge priority, this thesis represents a step towards a knowledge co-production approach that can improve the credibility, saliency, and legitimacy of research findings.

RÉSUMÉ

Les connaissances autochtones sur la faune et leurs habitats peut éclairer la recherche, tout en renforçant l'engagement et le soutien communautaires dans les décisions en conservation de la faune. L'exploitation forestière a des répercussions sur les écosystèmes et les communautés autochtones. Eeyou Istchee, le territoire traditionnel des Cris dans le nord du Québec, au Canada, comprend des zones qui sont fortement touchées par l'exploitation forestière. Des préoccupations ont été soulevées quant à l'impact des activités forestières sur l'orignal, une espèce sauvage d'une importance vitale pour la culture et la sécurité alimentaire des Cris. Le régime forestier adapté (RFA) a été adopté en 2002 afin de mieux intégrer les préoccupations des Cris et leur participation aux pratiques et à la gestion forestières. Ce régime comprenait l'identification de sites d'intérêt faunique spécial pour les Cris (zones 25 %), où la foresterie serait gérée pour réduire les impacts sur la faune, y compris les orignaux. Vingt ans après l'implémentation du RFA, la qualité de l'habitat de l'orignal n'a pas été évaluée. L'objectif de cette thèse est de faire progresser l'inclusion des connaissances expérientielles dans les analyses quantitatives de l'habitat faunique.

Le chapitre 1 présente une revue systématique de la littérature évaluée par des pairs pour intégrer les connaissances locales, autochtones et d'experts dans des analyses quantitatives de la faune. Ce type d'imbrication des connaissances peut contribuer à accroître l'applicabilité, la confiance et l'équité dans la science et la gestion fauniques. Nous avons examiné 49 articles et présenté les méthodologies employées pour la sélection des détenteurs de connaissances, les étapes de leur participation, l'obtention des connaissances, les processus de modélisation, la gestion des biais et des incertitudes, et la validation. Nous concluons avec six avantages clés, des limites et des améliorations recommandées pour les futures analyses d'imbrication des connaissances.

Le chapitre 2 évalue la sélection de l'habitat de l'orignal dans le RFA en s'appuyant sur les connaissances des experts cris afin de guider la sélection des variables et l'élaboration du modèle. Nous avons effectué une analyse du domaine vital, un modèle linéaire généralisé pour évaluer la sélection de l'habitat et des analyses de sélection de Manly pour évaluer comment les orignaux utilisent les caractéristiques de l'habitat par rapport à la disponibilité. Nous avons exécuté des modèles pour le milieu de l'été et le milieu de l'hiver de 2018 à 2021 pour 38 orignaux femelles

équipées de colliers GPS. Les orignaux ont sélectionné les zones 25 % au cours des deux saisons. En été, les orignaux ont sélectionné les petites îles, les forêts éclaircies, les forêts de conifères avec sapin et les zones inondables, tandis qu'en hiver, les orignaux ont sélectionné les forêts mixtes et les forêts de feuillus. Au cours des deux saisons, les orignaux ont utilisé les terrains d’élévation moyenne et élevée, tout en évitant les terrains de basses élévations. Les orignaux avaient tendance à utiliser les sites après l'exploitation forestière de la même façon ou plus qu’après des perturbations naturelles, même si la sélection était moindre que pour les peuplements intacts préférés.

Nous fournissons la première évaluation de l'utilisation par les orignaux des zones 25 % et quantifions l'utilisation des peuplements exploités, informée par et reflétant le savoir cri, soulignant l'importance d'une approche multi-saisons et multi-connaissances pour évaluer l'influence du RFA sur l'évolution de la qualité de l'habitat des orignaux. En illustrant comment le savoir cri peut informer une analyse quantitative de la sélection de l'habitat de l'orignal liée à une priorité du savoir local, cette thèse représente un pas vers une approche de coproduction du savoir qui peut améliorer la crédibilité, la pertinence et la légitimité des résultats de recherche.

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CONTRIBUTION OF AUTHORS

This manuscript-based thesis contains two original research chapters: Chapter 1 has been published in the journal *Biological Conservation* prior to thesis submission, and Chapter 2 will be published post thesis submission. The contributions to the original research are as follows: E.R. Stern contributed to the study design, analysis, and writing of all chapters, and the writing of all thesis material; M.M. Humphries contributed to the study design, writing, and manuscript edits of Chapters 1 and 2; G. MacMillan contributed to thesis material edits, the study design and manuscript edits of Chapter 2; M. Landry-Cuerrier contributed to thesis material edits and the study design of Chapter 2; and V. Brodeur contributed to study design and data acquisition for Chapter 2.

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LIST OF ABBREVIATIONS

1. AFR: Adapted Forestry Regime
2. GPS: Global Positioning System
3. GIS: Geographic Information System
4. MCP: Minimum Convex Polygons
5. NSD: Net Squared Displacement
6. GLM: Generalized Linear Model
7. GLMM: Generalized Linear Mixed Model

INTRODUCTION

Background

Inclusion of local, expert, or Indigenous knowledge about wildlife populations and their habitats can inform wildlife research, while also increasing knowledge holder engagement and support for wildlife conservation decisions. However, experiential wildlife knowledge accumulated over time through the personal observations of knowledge holders differs from other data based on systematic observations collected through standardized methodology such as telemetry locations or field surveys. Differences in the form and the function of these two types of wildlife information makes combining them into a single comprehensive analysis more easily encouraged than accomplished. However, interweaving these two sources of knowledge can be highly beneficial in situations where wildlife or ecosystem management has impacts on Indigenous communities. The wellbeing, culture, and food security of Indigenous communities in Canada may be reliant on boreal ecosystems and the subsistence hunted species such as moose that dwell within them. Historically, Indigenous peoples impacted the boreal landscape through extensive hunting of wildlife species, plant harvesting for food and medicine, and forest management such as controlled burns. Fires have also been a major disturbance throughout the boreal zone, both naturally occurring and as controlled burns. However, these ecosystems are additionally highly disturbed by modern natural resource extraction including forestry as well as natural disturbances such as fire and disease. Forestry management should include Indigenous knowledge and involvement to reduce negative impacts on communities and increase local involvement and trust.

In Eeyou Istchee, the Cree traditional territory in the James Bay area of Quebec, Canada, forest logging is abundant in the southern extent of the area and concerns have been raised by communities about the impact these activities have on moose populations. Moose (*Alces alces*), which are vitally important to the diet, food security, health, and culture of Cree communities in Eeyou Istchee, may be negatively impacted by forestry. To reduce negative effects, in 2002 the Adapted Forestry Regime (AFR) was put into place. The AFR is a region covering much of Eeyou Istchee in which regulations were developed with the goal of conducting forestry in a manner that allows: a) adaptations to better take into account the Cree traditional way of life, b) greater integration of concerns relating to sustainable development, and c) participation, in the form of consultation, by the James Bay Crees in the various forest activities operations planning and management processes. In order to do this, key measures that changed the targets for mosaic cutting, rotation time, and residual stands were enforced. Within the AFR, Sites of Special Wildlife Interest to the Cree (25% areas) were also chosen based on pre-existing good wildlife habitat, with unique management requirements in these regions focusing on entirely mosaic cutting and longer intervals between harvesting stands that were designed to either preserve or produce high quality moose habitat, but the success of this has not been assessed.

The Moose Habitat Quality Steering Committee, composed of government, Cree community, and academic stakeholders and rightsholders, have emphasized the need for an analysis that interweaves Cree knowledge and Global Positioning System (GPS) collar data to answer whether these 25% areas have been effective, and improve understanding of which land and forest types are most used by moose in the area. While the impacts of forestry activities, logging, and clearcutting on moose have been extensively studied, we present a novel study in its focus on assessing the effectiveness of targeted protective measures designed for moose.

Objectives

In this thesis, I will contribute a systematic review of primary literature that has included experiential wildlife knowledge into quantitative habitat and population analyses, to document the methods, successes, and limitations defining these attempts at experiential wildlife knowledge inclusion. The first chapter of the thesis presents a systematic literature review designed to identify and present a toolbox of methods to interweave knowledges, employed by previous studies that have interwoven local, expert, and Indigenous knowledge into wildlife habitat and population science. In this chapter, I survey literature identified through systematic review and snowball collection, identify common themes in study characteristics, knowledge holders, elicitation method, and methodology, discuss six case studies that exemplify these themes, and conclude with a set of identified benefits, limitations, and recommended improvements for knowledge interweaving.

In the second part of the thesis, I develop a particular case study focused on a quantitative wildlife analysis that seeks to address a local knowledge priority while being inclusive of local knowledge. This case study considers how the Adapted Forestry Regime in the James Bay area of northern Quebec is impacting moose nearly 20 years after it was implemented. Using a knowledge co-production approach in which Cree expert knowledge, GPS collar data for a sample of female moose in the Eeyou Istchee population, and Geographic Information System (GIS) land covariate data are interwoven, I will assess: a) home range and movement patterns, b) how moose are using the 25% areas relative to areas outside these special interest sites, c) which land covers are most selected by moose, focusing on the importance of height class, presence of fir, types of mixedwood, and types of disturbances, and d) which combinations of land characteristics drive moose habitat selection and can be used for predictive mapping based on GLM models. These analyses will be done at both the second and third order of analyses to obtain a stronger understanding of moose behaviour. In the second chapter, I perform Minimum Convex Polygon (MCP) home range, Net Squared Displacement (NSD? movement, Resource Selection Functions (RSF) using Manly selection ratios, and Generalized Linear Model (GLM) analyses based on systematic data such as GPS collar locations and land covariate GIS data, as well as qualitative data collected from in-person interviews with Cree experts in which habitat relationship network maps were developed. These analyses answer key questions about moose habitat use and the success of special interest sites developed by the AFR in Eeyou Istchee.

LITERATURE REVIEW

Knowledge Co-Production in Wildlife Science

Knowledge co-production refers to research practices that "*co-produce knowledge with local decision-makers and stakeholders that is useful and usable, or 'actionable'*" (Latulippe and Klenk, 2020). Knowledge co-production is a philosophy of co-operative science based on the idea that ecological challenges and attempts to address them involve many groups of people with differing needs and interests and that science should be participatory and inclusive (Norström et al., 2020). For science to be effectively applied, it must be considered credible, salient, and legitimate by stakeholders (Cash et al., 2003). Cash et al. 2003 discuss that to be credible, science should be technically adequate and sound; to be salient, science should be relevant to the needs of stakeholders; and to be legitimate, science should be respectful of stakeholders beliefs and values, unbiased, and fair to opposing views or interests (Cash et al., 2003), however these requirements may be in conflict as both scientists and stakeholders inherently have bias, and being respectful of beliefs and values may be at odds with objectivity. By using knowledge co-production approaches, science can meet these requirements of credibility, saliency, and legitimacy, and avoid treating Indigenous knowledge extractively or simply assimilating it into science (Latulippe and Klenk, 2020), however each project must navigate bias and conflicting goals that arise and should communicate these challenges where applicable. Norstrom et al. (2020) lays out a framework of four principles of knowledge co-production, arguing that projects should be: 1) *context-based*, situated in the specific social, economic, or ecological context relevant to the stakeholders and issue at hand; 2) *pluralistic*, recognizing multiple ways of knowing; 3) *goal-oriented*, with collective, clearly defined goals and agreed measures of success; and 4) *interactive*, with frequent collaborative meetings throughout the process that include multiple stages including framing, conducting, using, and disseminating the research (Norström et al., 2020). Challenges and opportunities may emerge in systems with diverse stakeholders, goals, and perspectives on ecological challenges.

Case Study: Boreal Forest Logging and Moose in Eeyou Istchee

Importance of moose in Eeyou Istchee

One such situation in which diverse stakeholders, goals, and perspectives are cooperatively addressing a shared ecological challenge is occurring in Quebec, Canada with regards to the impact of boreal forest logging on subsistence harvested species and the Indigenous communities they are important to. In particular, Eeyou Istchee, also called the James Bay area, is currently struggling to balance the management of logging with the needs of Cree moose hunters. Eeyou Istchee is the traditional territory of the Cree in the James Bay region in Quebec. The region is large and contains 11 permanent communities, of which five are located in territories currently impacted by forestry activities (Jacqmain et al., 2008). Moose are considered the dominant concern regarding forestry impacts by those living in the area (Jacqmain et al., 2012). Moose are the primary source of subsistence harvested food, as well as being culturally and socially significant. This species is an essential part of the Eeyou Istchee Cree identity, through their connections to spirituality, as a food source, as materials for traditional crafts, and as a social glue through group or family hunting, time on the land, and crafting.

Eeyou Istchee Cree have a long history of self-regulating the number of moose hunted annually based on observations of moose populations and frequency of sightings during the year (Feit, 1987). Moose represent a critical food species in a northern region where traditional food security is a concern (Willows et al., 2005). Presently, moose and other traditional foods are vital for maintaining food security and reducing dependency on "ultra-processed products" in many remote communities in Eeyou Istchee or for individuals who reside in remote camps for large parts of the year (Noreen et al., 2018). These foods are additionally important for improving health, wellness, and protecting against chronic disease in communities which experience disproportionate rates of chronic diseases (Gaudin et al., 2014). Traditional food intake is also correlated with tendency to speak Cree at home (Noreen et al., 2018) and may be important for preserving culture and language.

Forestry activities and legislation in James Bay, Quebec

In Canada, the participation and consultation of First Nations people is a requirement for forestry management and activities that impacts their rights to traditional use (Canadian Council of Forest Ministers and Canadian Forest Service, 2006). First Nations groups have rights enshrined in law under Section 35 of the Constitution Act, which include rights over land and natural resources. There is often significant overlap between forestry activities and First Nations communities, necessitating cooperative management and formal agreements. To address rights-based conflicts, in 2002 the Cree people of the James Bay Area and the Government of Quebec signed the *Paix des braves* Agreement, which set out cooperative frameworks for hydro-electric, mining, and forestry development in Cree traditional territory (Le Gouvernement Du Québec, 2002). A major component of the *Paix des braves* agreement was the re-organization of forestry in southern James Bay to ensure more Cree influence in decision making (Chaplier, 2018).

The agreement laid the framework for an Adapted Forestry Regime (AFR) that contained new provisions which increased protection of areas of special interest, including zoning stipulations for 1% of land to be reserved as Sites of Special Interest to the Crees (1% areas) and 25% to be reserved for Sites of Special Wildlife Interest to the Crees (25% areas) (Gouvernement du Québec, 2002). The agreement specifies forestry management practices that must occur with the aim of reducing impacts on traditional use of the land. The *Paix des braves* was put into place in response to concerns that forestry companies were operating in ways that were detrimental to moose habitat. Prior to the implementation of the AFR, tallymen expressed specific needs to have “moose yards”, or highly productive forest stands for moose, protected from logging, however these concerns were not always heeded (Whiteman, 2004). Interviews of land users at the time indicated that moose yards were clear cut which had a negative impact on moose hunting (Whiteman, 2004). Historically, the Crees were excluded from much of the resource extraction decision making in the region (Desbiens, 2004).

Non-scientific media coverage at the time discussed crashing moose populations in the area and attributed this largely to forestry (Nicholls, 1999), however this claim was contested by scientific reports of moose status and management plans conducted for the Grand Council of Crees of Québec (Messier, 1993; Messier, 1996; Messier 1998). These reports corroborate the claims of moose decline and cite a 50% population decline in the region throughout the 1980s, but attribute the decline to an unsustainable harvest rate of 27% annually (Messier, 1993). These reports discuss the impact of forestry being primarily through increasing hunting access via road development (Messier, 1993; Messier, 1998). These reports suggested that forestry created more productive habitats for moose, but the expanded access has facilitated hunting-driven population decline (Messier, 1998).

Moose population concerns were supported in unpublished results of aerial population surveys conducted by the Gouvernement du Québec which indicated decreasing numbers of bull (male) moose relative to cow (female) moose and a substantial drop in calves (juveniles) per cow in the years preceding the implementation of the AFR. Furthermore, population estimates indicated major declines and population fluctuations throughout the 1990s. These concerns about forestry impacts on moose were not verified through peer reviewed study in the region at the time, with reports instead focusing on the overharvest hypothesis as the primary cause of moose declines, citing unsustainable harvest levels and high harvesting of female moose (Messier, 1993; Messier, 1996; Messier, 1998). However, the need was identified by Cree and government for substantial change in forestry practices that would strike a better balance between forestry industry needs and moose habitat conservation.

The Adapted Forestry Regime in the *Paix des braves* agreement was in part enacted to address moose population needs, however recent media focus has continued to emphasize ongoing concerns raised by the Cree in Eeyou Istchee about moose declines in the region (Bell, 2022; Bell, and Herodier, 2020). Further concerns continue to be raised about the negative impact of forestry activities, which contradicts claims from some forest managers that forestry activities benefit moose through rejuvenating stands and creating feeding areas (Jacqmain et al., 2012), as well as scientific literature that indicates that forestry may create moose forage and habitat similar to natural disturbance (Crête, 1988) and can increase moose site use and populations (Collins and Schwartz, n.d.; Potvin et al., 1999). Formal data on moose populations in the area is focused on aerial surveys conducted by the Gouvernement de Quebec, and further study is needed to explore these concerns of negative forestry impacts on moose. An increased understanding of how forestry practices are impacting moose habitat use is needed to address the situation in the James Bay area.

Key aspects of the AFR

The Adapted Forestry Regime is a large area of land in Eeyou Istchee, making up over 68,000 square kilometers of land. In the *Paix des braves* agreement, particular management stipulations were developed in order to accomplish the goals of better taking into account the Cree way of life, better integrating Cree concerns, and facilitating and increasing Cree participation in forestry planning and management (Gouvernement du Québec, 2002; Desbiens, 2004; Cyr et al. 2022), however the agreement is recognized as posing many challenges regarding implementation, complexification, and placing disproportionate burden on Indigenous parties (Cyr, 2022). A multi-pronged approach was used to achieve this, including the development of sites of special interest, new management practices aimed at preserving forest cover at a trapline scale, protecting forests adjacent to water bodies to leave riparian buffer zones unharvested, and developing a road access network to facilitate increased access to the land (Gouvernement du Québec, 2002). The AFR changed the scale of management to the trapline unit, with each trapline having individualized plans for harvest and management based on pre-existing disturbance rates (Gouvernement du Québec, 2002).This approach of trapline-based management and regulation was significant in that it legally recognized the importance of traditional family hunting territories and better aligned the scale of forestry management with the scale of traditional Cree hunting management (Tanner, 2018) In general, forestry practice stipulations include (Gouvernement du Québec, 2002):

a) Conservation of at least 30% of forest over 7 m tall,
b) Halting logging on traplines with over 40% of forest disturbed over the last 20 years,

c) Aim for 75% of logging to be done using mosaic cutting with an emphasis on protection of regeneration and soils,

d) Limiting cutblock size to 100 ha per single block, and mandating that 40% of logged areas must be made up of cuts under 50 ha,

e) Modulate the annual level of logging based on the previous level of disturbance,

f) Protect tall regeneration,

g) Use silvicultural practices that promote diversified habitats and avoid eliminating hardwood trees,

h) Develop special management approaches for mixedwood stands,

i) Leave 20 m buffer zones around permanent watercourses and waterbodies, and

j) Develop a road access network that limits the connections between traplines, forms closed circuits that does not facilitate easy movement between traplines, and limits construction of new access routes to water.

These stipulations apply generally to all land in the AFR, with notable exceptions for Sites of Special Interest to the Cree (1% areas), and Sites of Special Wildlife Interest to the Cree (25% areas) (Gouvernement du Québec, 2002)). These areas were chosen for each trapline by the tallyman managing the area. The 1% areas made up 1% of land per trapline, and contained important sites such as camps, cultural sites, bear dens, trails, and drinking water, and were to be left completely unlogged (Gouvernement du Québec, 2002)). The 25% areas made up 25% of land per trapline, and while forestry activities continued in these areas, separate management rules were set in place for them (Gouvernement du Québec, 2002)). These sites were generally chosen by tallymen based on pre-existing high-quality habitat for wildlife, especially moose. As such, the management guidelines had the aim of either preserving this pre-existing high-quality habitat or reducing negative impacts of forestry activities on moose. The specific management stipulations include (Gouvernement du Québec, 2002)):

a) Only mosaic cutting should occur (rest of land has 75% mosaic cutting target),
b) Conservation of at least 50% of forest over 7m tall (rest of land has 30% conservation target),
c) Selection of interconnected residual blocks decided by tallyman with a focus on minimizing gaps in connectivity, and
d) Slower rotation times with residual forest being left to regenerate to 7 m between harvests (rest of land may be harvested between 3 and 7 m tall).

The effectiveness of these 25% areas has not been assessed, but key insights may be provided by available moose GPS data and Cree knowledge.

Cree knowledge as a source of moose data

The Eeyou Istchee Cree have a long history of collecting knowledge on the moose in the area. In Eeyou Istchee, hunting management and monitoring is organized around a system of hunting territories called "traplines" or hereditary hunting territories. Management and monitoring is conducted by Tallymen, expert hunters who inherit the management responsibility, and monitor wildlife and hunt on the same trapline for long time periods (Feit, 1987; Whiteman, 2004; Chaplier, 2018), with some families spending weeks or months of the year on traplines (Berkes & Farkas, 1978). Tallymen are generally responsible for monitoring and approving hunting activities on their traplines, and are critical to preserving sustainable management in their regions (Whiteman & Cooper, 2000). Land-based activities were important for strengthening social networks, values, beliefs, and culture (Rodon, 2014; Chaplier 2018), and people outside a tallymans family may also hunt or do land-based activities on a tallymans trapline with permission (Whiteman, 2004). Historically, families were highly dependent on moose meat to survive because it is a substantially more efficient hunting choice than alternative animals, being 3 times more efficient than beaver hunting, 6 times more efficient than fishing, and 15 times more efficient than small game hunting (Feit, 1987). However, in the past moose hunting had much lower reliability and success than other hunting activities, resulting in detailed knowledge of where to find moose needing to be collected to increase success (Feit, 1987). This detailed collection of knowledge is exacerbated by the management system, in which hunting tends to occur repeatedly on specific traplines, resulting in a tendency to hunt frequently over a limited area (Feit, 1987). This dynamic resulted in substantial long-term collection of detailed information about moose populations and habitat for specific areas. Tallymen build up complex and detailed knowledge of moose, other wildlife, and ecosystem dynamics in these highly localized areas (Feit, 1987; Whiteman, 2004). This knowledge is built up through an interweaving of personal observations and experiences, inter-generational transfer of knowledge through families and communities, and Cree worldviews and values (Scott, 1989). This provides an ideal source of information to explore moose habitat dynamics and changes to ecosystems through knowledge co-production approaches.

Boreal forest ecology

General boreal forest ecology

Boreal forests are one of the world's largest ecosystems, covering large portions of North America and Eurasia (Brandt, 2009), and serve as critical habitat for species (Racey and Arsenault, 2007), provide ecosystem services such as climate regulation and carbon sequestration (Brandt, 2009), support communities dependent on subsistence hunting and trapping (Johnson and Miyanishi, 2012), and have cultural and spiritual significance to many Indigenous peoples (Brandt, 2009). Boreal ecosystems are typically made up of a mosaic of different forest types, wetlands, and water bodies (Brandt, 2009). The abundant natural resources such as natural gas, minerals, and lumber within boreal forests make this ecosystem one of the most intensely disturbed ecosystems on the planet (Brandt, 2009; Johnson and Miyanishi, 2012), with approximately two-thirds of the global boreal forest being utilized in some way (Gauthier et al., 2015). This intensity of anthropogenic disturbance has altered the natural dynamics of ecosystems and may affect broad scale conditions such as dominant vegetation and wildlife assemblage compositions (Bichet et al., 2016), as well as fine scale conditions such as microsite abiotic dynamics including sunlight and wind in both open clearings and linear disturbances (Stern et al., 2018).

The boreal forest is a complex ecosystem from a management perspective, serving as a critically important ecozone for species conservation and subsistence food harvest and subsistence food harvest, while also being one of the most heavily resource extracted ecosystems on the planet. Oil exploration and extraction, mining, and forestry operations are frequent in the boreal forest. The balance between species conservation and resource extraction is challenging to strike. The needs of species and communities that depend on these species for subsistence food may diverge from resource extraction goals. Resource extraction such as forestry may negatively impact subsistence species through loss or change in food plants and habitat, and through disturbance or disruption to normal or important wildlife behaviours. However, in some cases, the potential exists for forestry to occur in ways that support or promote wildlife habitat, by providing disturbance to old forest that kickstarts new growth, creating new habitat types such as clearings or open meadows, or creating heterogeneity across areas. Particular forestry practices such as mosaic harvesting and retention forestry may more closely mimic natural disturbance, preserve enough intact areas for habitat, increase heterogeneity, and retain old growth areas and species. These effects may reduce negative impacts on wildlife and communities.

Boreal forest disturbances

Disturbances are not by default harmful to boreal ecosystems, which are adapted to intense and frequent disturbance regimes. Large natural disturbances such as fire can drive habitat diversity and biodiversity in boreal forests (Burton et al., 2008). Boreal forests are located at northern latitudes that receive weather conditions conducive to frequent fires (Krawchuk et al., 2006), tend to be in areas prone to ice and wind disturbance (Romeiro et al., 2022), and are prone to periodic insect-related diseases (Sánchez-Pinillos et al., 2019). This history has caused boreal forest ecosystems and the species within them to be well adapted to disturbances and are resistant to high burn rates (Héon et al., 2014). Additionally, in Canada's boreal forest, human-driven successional changes have been occurring for millennia, through controlled burns (Hoffman et al. 2022). Dominant species in boreal forest ecosystems often display disturbance-adapted traits such as serotiny, in which seeds are released only after exposure to fire (Lamont et al., 2020), and rapid post-disturbance establishment and growth through tactics like root suckering (Jean et al., 2020). Post-disturbance dynamics in boreal forests are a complex sequence of succession events in which understory and canopy vegetation communities shift over time based on changing nutrients, sunlight, and competition in the decades following a major disturbance (Angelstam and Kuuluvainen, 2004; Bergeron and Fenton, 2012) . Successional stages generally include bare, stand initiation, young, middle-aged, mature, aging, and old growth (Angelstam and Kuuluvainen, 2004). After a natural disturbance, forest dynamics are driven by the remaining dominant species, and structure and composition of the stand post-disturbance (Sánchez-Pinillos et al., 2019). Different boreal wildlife species may thrive at different successional stages, with species such as some songbird species and caribou finding suitable habitat in old growth boreal forests, and other species such as deer, moose, and beavers relying on increased food plant productivity of young forests relatively soon after disturbance (Telfer, 1974).

Boreal forestry

The impacts of forestry on boreal succession

Successional change may be driven by anthropogenic disturbances from resource extraction along with natural disturbances. The proportion of old forest has declined with intensifying forest utilization, and a major driver of forest successional change in boreal forests is short-rotational harvesting (Kuuluvainen and Gauthier, 2018). Resource extraction pressure is heavy in boreal forests. In particular, forestry operations, in which trees are removed to be turned into lumber, are abundant. Forestry activities are very diverse, and there is a wide range of harvesting methods and tactics that may be employed to change the potential impacts on the affected ecosystems. In the 20th century, most boreal forestry logging was clear-cutting (Telfer, 1974), and the legacies of this forestry practice are still evident today (Bouchard and Pothier, 2011; Lundmark et al., 2013). A large body of work in the 20th century documented the significant effects that clear-cutting had on every level of forest ecology, including nutrient retention of ecosystems (Bormann et al., 1968), soil and organic matter (Covington, 1981; Johnson et al., 1991), hydrology (Brown and Krygier, 1970; Jones and Grant, 1996), vegetation (Hix and Barnes, 1984), and wildlife (Potvin et al., 1999; Schelker et al., 2013).

Adaptive forestry techniques

In response to the growing understanding of how clearcutting and the associated site preparation and management negatively impacted the boreal ecosystem and the species within it (Keenan and Kimmins, 1993; Potvin et al., 1999; Schelker et al., 2013), alternate methods have developed. One such method is mosaic harvesting, in which cutblocks tend to be small, irregularly shaped, and are designed to increase heterogeneity across large forested areas by balancing their proportions with retained or unlogged forest. Mosaic harvesting is claimed to more closely resemble the asymmetric and patchy patterns left by natural disturbances such as fire and wind throw. Another method is retention harvesting, in which particular types or stands of trees may be left behind in cutblocks or areas based on age, height, or species in efforts to reduce the structural and functional differences between logged and natural forests (Fedrowitz et al., 2014). Retention harvesting may reduce negative impacts on species by retaining critical food or habitat plants, maintaining some canopy closure and protective cover, or maintaining coarse woody debris (Gustafsson et al., 2012; Kuuluvainen et al., 2019). However, species response to retention practices is highly situational and dependent on the level and type of retention and benefits may be reduced if retention practices are monotonous across landscapes (Kuuluvainen et al., 2019). Other practices may include: the use of buffer zones around critical habitat such as water bodies; use of rotation schedules, where forests are allowed a certain timeline with undisturbed regrowth (Egnell and Björheden, 2013; Roberge et al., 2016); harvest area or proportion limits; natural-disturbance forestry, where harvest is done in a way that mimics fire or ice storm disturbance in timing, scale, or post-disturbance forest remnants (Bolton and D'Amato, 2011; Harvey et al., 2002); and selective harvesting (Asner et al., 2004; Piponiot et al., 2016).

While tree height recovery time is generally similar between harvest and wildlife disturbances (Bartels et al., 2016), regrowth from forestry may not mimic natural successional change that follows natural disturbance such as forest fires or ice storms (Lindenmayer and McCarthy, 2002; Schmiegelow et al., 2006). These post-logging successional stage changes can impact wildlife community assemblages (Hobson and Schieck, 1999; Schlossberg and King, 2009), and drastically change forest composition and wildlife populations for long time frames (Eyre et al., 2015; Phoonjampa et al., 2011). Measures to reduce negative impacts on some species such as mosaic harvesting, retention harvesting, and natural disturbance harvesting have been shown effective in some cases (Ketzler et al., 2018); however, other studies suggest that these tactics are not sufficient at protecting some species at risk because of legacies not accounted for by these methods, such as logging roads which can increase wildlife mortality (Nielsen et al., 2008). These long term remnants of forestry on the landscape like roads can counter positive impacts such as new growth and browse creation by increasing vehicle collisions and hunting access (Nielsen et al., 2008).

These long-term legacies can be altered by post-harvest techniques, which include measures to replant logged areas. Because the successional path of a stand is highly driven by the remaining dominant species and composition of vegetation after any disturbance (Sánchez-Pinillos et al., 2019), post-harvest decisions can have major impacts on harvested landscapes (Thompson et al., 2003). In boreal forests the composition of stands post-disturbance generally differs between natural and logged stands: in cases where forests were originally upland or lowland black spruce, following natural disturbance black spruce tends to re-establish, while following logging and post-harvest treatment mixedwood and alder most commonly re-establishes (Thompson et al., 2003).

Forestry impacts on wildlife habitat and populations

In the boreal forest, forestry impacts on wildlife generally involve increasing fragmentation, changes to connectivity, cover from predators, changing behaviour of predators, increased access for hunters, changing habitat types from old growth to new growth forest, and changing food availability. The response that wildlife have to forestry operations varies between taxa and is scale dependent (Kellner et al., 2019). For species adapted to intact, undisturbed forest such as caribou, forest cutblocks can fragment landscapes and impair connectivity (Yemshanov et al., 2021). This fragmentation can be partially mitigated with residual blocks, but the configuration of them is important to be effective (Boucher et al., 2011). Fragmentation may also increase forest edges (Boucher et al., 2011) that may reduce cover from predators and increase predation (Thompson et al., 2008). Disturbances such as clearings and linear features can also facilitate predator movement (Houle et al., 2010) that can increase predation pressure on prey species (Courbin et al., 2014). Hunting access may also increase due to increased roads and provide hunters access to regions which were previous refuges for wildlife (Rempel et al., 1997). Forestry changes successional stage of stands from tall, mature forest to young, regrowing forest which can either harm species that are dependent on older stands like caribou and some birds, or may benefit species that rely on the high productivity of early-stage stands such as grouse, beaver, and deer (Telfer, 1974). Forestry logging may remove some food species if they are sourced from old growth species such as lichen used by caribou (Metsaranta, 2007) or it may increase food sources for species that forage on new growth species (Telfer, 1974). These impacts can accumulate to significantly effect wildlife habitat directly or indirectly, with positive, negative, and mixed effects. The largest changes to populations and abundance are usually seen in bird species but effects are present in mammals and reptiles as well, with different taxa showing either positive or negative responses to forestry (Kellner et al., 2019).

Forestry impacts on Local Communities and Indigenous Peoples

The impacts of forestry activities in forests extend beyond wildlife, with local communities and Indigenous Peoples being acutely impacted by natural resource extraction worldwide. If forestry activities negatively impact populations or behavior of wildlife species important to indigenous subsistence food, communities may be affected by the changes in ecosystem services and potential land use (Fuentes et al., 2020; Stevenson and Webb, 2003). The culture, health, and wellbeing of Indigenous communities is closely tied to the land and wildlife that inhabit it because of the importance of hunting, fishing, and trapping (Bélisle et al., 2021; Bélisle and Asselin, 2021). Changes to the abundance and quality of, access to, or experience on the land is a pressing concern for many Indigenous communities and people (Bélisle and Asselin, 2021).

Conversely, forestry practices can also support the livelihoods of Indigenous communities (Nath and Inoue, 2010), and when forest resource extraction is community driven, it has the potential to reduce inequality and poverty (Nhem et al., 2018). Furthermore, it is not always true that Indigenous communities goals are always in concert with forest preservation and at odds with resource extraction. While intact forests are tremendously valuable (Watson et al., 2018) and can preserve species populations and habitat, blanket habitat protection goals may be at odds with

Indigenous and local food security by restricting access to traditionally hunted or gathered plant and animal species (Sylvester et al., 2016).

For these reasons, the involvement of Indigenous communities in forestry management presents a path towards conducting forestry in ways that supports Indigenous communities while reducing negative impacts on land value and traditional land uses (Lawler and Bullock, 2017). This can have positive impacts to local economic, sociocultural, and environmental wellbeing (Lawler and Bullock, 2017). Indigenous communities have significant ecological knowledge and long histories of ecosystem maintenance, of which forest and wildlife use may be a part (Berkes, 2018). For these reasons, forestry and logging management is increasingly utilizing Indigenous knowledge and participation to direct or inform management in Canada (Abu et al., 2020; Armitage et al., 2011; Jacqmain et al., 2012, 2008, p.; Latulippe and Klenk, 2020).

Moose ecology

General moose habitat use

Moose are a cervid (deer) species that is present in many boreal regions of the world and are widely distributed in Eurasia and North America (Timmermann and McNicol, 1988). In most regions that moose live, they are a commonly hunted species for both sport and subsistence purposes and have significant intrinsic, spiritual, recreational, and economic value (Condon and Adamowicz, 1995; Grima et al., 2019; Timmermann and Rodgers, 2005). Because of the close relationships between moose and humans, moose habitat use has been extensively studied and is fairly well understood. In general, moose inhabit boreal forests and thrive in diverse habitats (Telfer, 1974). Because of the highly varying seasonal differences in the far northern latitudes moose inhabit, moose display strong seasonal habitat preferences based on food and shelter availability. In winter, moose tend to concentrate habitat use in small, focused areas with high value resources and home range sizes tend to be relatively small and moose commonly reside in a mosaic of patches only several hectares in size (Telfer, 1974). In these winter habitats, moose forage can include willow, red osier dogwood, saskatoon, and other shrubs (Poole and Stuart-Smith, 2006).

Moose seasonal behaviour

Moose may use different habitats in different seasons. In winter, when the ground is covered by snow and deciduous plant species have lost much of their foliage, moose may still use deciduous browse such as hazel and aspen stems (Courtois et al., 1998). Other known sources of winter food for moose are willow and white birch (Newbury et al., 2007).Moose are known to select for winter home ranges with pine, spruce, and older (>10 years) logged forest (Poole and Stuart-Smith, 2006). Fir can be important habitat for moose in winter, as well as areas with dense cover and abundant vegetation (Pierce and Peek, 1984). Moose are known to avoid open habitats in winter and use closed hardwood stands (Jung et al., 2009).

In summer, habitat may differ. In this season when vegetation is growing and more abundant than winter, moose eat a higher quantity of food than in winter (Renecker & Hudson 1985). The primary diet in summer is leaves from deciduous trees and shrubs (Timmermann and McNicol, 1988). Other important sources of summer food include aquatic plants with both emergent and submergent species being eaten as well as some algae species (Drucker et al., 2010; Timmermann and McNicol, 1988), making wetlands and water potential important habitat types. In summer, moist lowland habitats near surface water are known to be preferred (Timmermann and McNicol, 1988). In summer habitat use can be highly varied, with moose using cutblocks, aquatic areas, and forests within a single day based on temperature, food availability, and time of day (Leptich and Gilbert, 1989).

Disturbance impacts on moose

The impact of fire on moose is inconclusive, with different studies indicating that moose may prefer previous sites of low-severity burns (Brown et al., 2018) and other studies indicating moose prefer previous sites of high-severity burns (Lord and Kielland, 2015). Moose may use the abundant regrowth after a fire as a source of food (MacCracken et al., n.d.). Abundance of moose may increase in burned stands, with stand value peaking between 17-26 years after a fire, after which value may decrease (Loranger et al., 1991). Post-fire regrowth may be especially good food for yearlings and cause associated increases in density of moose after a burn (Peek, 1974).

The impacts of forestry activities on moose populations and behaviour have been extensively studied in many boreal areas, including Canada and Fennoscandia. Logging may improve moose habitat by creating browse (Collins and Schwartz, 1998). Through careful utilization of techniques such as clearing, scarification, and seedling establishment, logging may replicate the benefits that fire disturbance has for moose food production, and these techniques may increase carrying capacity based on forage supply up to 20 to 45 fold compared to unlogged mature forest (Collins and Schwartz, 1998). However, these benefits may not be seen in logged areas that do not utilize moose-benefitting techniques (Collins and Schwartz, 1998). These benefits are time-sensitive; post-logging moose browse production has been found to peak between 5 and 20 years after logging disturbances occur (Crête, 1988). These benefits may be limited by species dynamics, as historically logging regrowth has tended towards coniferous species (Crete 1988), which may not align with the most desirable forage for moose which is usually known to be deciduous trees and shrubs such as aspen, willow, and birch (Hörnberg, 2001; Månsson et al., 2007). Logged mixedwood stands in particular have been identified as good moose habitat (Potvin et al., 1999). However, the potential exists for early succession plant assemblages to provide valuable moose food for particular windows of time after logging, while mature stands many years after logging may not provide valuable moose habitat. Moose have been found to avoid very recent cuts within 1-8 years of disturbance, select regenerating cuts between 9-24 years, and avoid cuts over 25 years since disturbance (Mumma et al., 2021). Past evidence suggests that while small logged areas may be used by moose, large clearcuts (over 1.3 km^2) are avoided, and increasing amounts of small area clearcutting has also been associated with increasing moose populations in some regions (Telfer, 1974). Moose are more likely to be found in areas of high logging in some parts of Canada (Shura and Roth, 2013), and moose densities have increased up to 87% in logged areas when paired with strict hunting regulations (Potvin et al., 2005).

Logging may have additional impacts beyond the removal of trees. Forestry may impact moose through the increase in road or cutline densities on landscapes. Forestry activities leave an abundance of roads behind long after logging has ceased. This impacts moose by affecting predator-prey relationships critical to moose population dynamics through altering habitat quality and connectivity (Courbin et al., 2014). This may facilitate increased contact between moose and wolves or hunters. Increasing road density has a negative impact on the presence of moose

(Beazley et al., 2004). Clearings near forestry roads can provide high quantity and quality food plants for moose, however these roads are also used by predators such as wolves and humans which can increase pressure on moose populations (Loosen et al., 2021). Moose are also susceptible to vehicular collision mortality, and special roadside vegetation management practices may be needed to reduce this once roads are installed (Rea, 2003).

Moose habitat use in Eeyou Istchee and Québec

Previous work has been done in Eeyou Istchee to assess moose habitat selection at the start of the implementation of the AFR, providing the ability to compare results from the present day analysis to moose behaviour and habitat use in the same area in prior years. Notably, Jacqmain et al. conducted a 2008 study employing data from the initial years of the AFR, performing a multi-season analysis (Jacqmain et al., 2008). They found that in mid-winter, moose selected elevated terrains, mid-aged and mature mixedwood forests, and avoided wetlands, water, alder, and black spruce. In summer, they found that moose selected coniferous stands with fir, alder, and mature mixedwood, while avoiding water bodies and black spruce without fir. Jacqmain et al. (2008) also assessed the effect of distance to water on site selection, and found that in summer, moose prefer to be within 0 - 50 m from water bodies and watercourses. Their home range analysis found high variation between individuals, with largest home ranges occurring in summer, with a mean area of 125.5 km^2, and smallest home ranges occurring in winter, with a mean area of 3.6 km^2 (Jacqmain et al., 2008). This study provides valuable historical context of how moose used habitat in Eeyou Istchee before key practices of the AFR came into effect.

Other literature has documented moose habitat use in adjacent areas to Eeyou Istchee. A study in eastern Québec and central Labrador identified river valleys and adjacent hillsides, riparian areas, hardwood stands, closed canopy coniferous forest, and burned forests as areas inhabited by moose, while highlighting bogs, open areas, and barren hilltops as less frequently used by moose (Jung et al., 2009). A study in the Laurentides Wildlife Reserve of Québec found that the best supported model to explain moose habitat use was determined by both presence of roads and ecosystem conditions (presence of shelter and food plants), finding that in summer moose used shelter proportionate to availability, and in winter moose used shelter more than availability (Laurian et al., 2008). They also found that moose selected high elevations in summer and autumn, which differ from the findings of Jacqmain et al. (2008) that moose preferred high elevations in winter. These differences may be attributed to difference in location and ecosystem. A study in north-west Québec that assessed moose use of habitat categorized as food and cover (deciduous, mixedwood, and spruce stands with budworm outbreaks), only cover (coniferous stands), or cuts found that moose selected stands that acted as both food and cover, as well as only cover, and had no specific relationship with cuts (Courtois et al., 2002).

Moose use of clearcut areas in a similar region to Eeyou Istchee have been studied in the past. Courtois et al. (2002) studied moose use of clearcuts near Rouyn-Noranda, an area with a similar ecosystem several hundred kilometers south of Eeyou Istchee (Courtois et al., 2002). They assessed habitat use at two scales, and found that at a coarse scale, moose preferred cuts to other forest types. At a fine scale, cuts were less preferred in summer but were preferred in winter. However, an increasing proportion of cuts in a moose home range increased the home range size for female moose. They found that habitat preference was more pronounced at a fine scale than a course scale of analysis.

The effects of road networks on moose has been assessed in similar areas to Eeyou Istchee. A study in the Laurentides Wildlife Reserve of Québec studied the interactions between moose and road networks and found that moose avoided highways and paved roads up to a distance of 750 m, depending on the season (Laurian et al., 2008). Females tended to have narrower areas of avoidance close to paved roads compared to male moose, and moose avoided up to 500 m away from drivable roads, however this study focused on paved provincial roads and not post-logging forestry roads (Laurian et al., 2008).

CHAPTER 1: INTERWEAVING LOCAL, EXPERT, AND INDIGENOUS KNOWLEDGE INTO QUANTITATIVE WILDLIFE ANALYSES: A SYSTEMATIC REVIEW

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Highlights

- Review articles were distributed around the globe and across terrestrial vertebrate species, but most frequently were situated in Australia, Canada, and United States and focused on large, harvested or charismatic mammals
- The most common knowledge holders were hunters and trappers, community members, and academic experts
- The most common form of experiential wildlife knowledge was as point observations or habitat covariate selection or valuation and the most common models were GLMM, GLM, and other regressions to build habitat models
- Benefits of experiential wildlife knowledge interweaving include increased trust in science and management, improving equity between knowledge holders and scientists, providing additional or rare data, and improving temporal transferability of models
- Improvements needed are multi-model studies and comparisons, standardized methods of accounting for variation and bias, increased discussion of power disparity and intellectual property rights, and more involvement of knowledge holders in multiple study stages

Abstract

Inclusion of local, expert, or Indigenous knowledge about wildlife populations and their habitats can inform wildlife research, while also increasing knowledge holder engagement and support for wildlife conservation decisions. However, experiential wildlife knowledge accumulated over time through the personal observations of knowledge holders differs from other data based on systematic observations collected through standardized methodology such as telemetry locations

or field surveys. Differences in the form and the function of these two types of wildlife information makes combining them into a single comprehensive analysis more easily encouraged than accomplished. Here, we systematically review primary literature that interweaves the experiential wildlife knowledge of diverse knowledge holders into quantitative, mixed methods analysis of terrestrial vertebrate populations and their habitats. Forty-nine studies that met our selection criteria were distributed around the globe and across terrestrial vertebrate species, but most frequently were situated in Australia, Canada, and United States and focused on large, harvested mammals including ungulates, carnivores, primates, and elephants. The most common descriptor of knowledge holders was hunters/trappers, with academic experts and community members also common. The most common analyses interweaved experiential wildlife knowledge as point observations in habitat models or as habitat covariates in habitat selection analyses. Local knowledge was also included, less frequently, in species distribution models, population models, and occupancy models. Most articles accounted for bias and uncertainty either in the knowledge elicitation stage through study design or knowledge holder selection, or in the analysis stage through regression methods. Most articles that assessed model success did so through comparison to independently collected telemetry locations or field survey data. There was wide variation in self-reported success, with the majority of authors offering neutral or positive assessments and many discussing study-specific factors contributing to model performance. Our overall assessment of these 49 studies, including 6 examples described in more detail, highlight several key challenges and solutions related to the inclusion of local, expert, and Indigenous knowledge into quantitative wildlife habitat and population analyses related to i) the incorporation of uncertainty, bias, reliability, and variation in experiential wildlife knowledge, ii) matching the scale of experiential wildlife knowledge to scale of study objectives, and iii) the appropriate use, communication, and application of experiential wildlife knowledge, including issues of consent, member checking, and knowledge co-production. We conclude with several recommendations intended to better standardize and communicate uncertainty, increase the involvement of knowledge holders in multiple stages of the research, improve validity assessment through multiple model comparisons and triangulation, and encourage more careful consideration of intellectual property protection and research ethics.

Keywords

Wildlife, Local knowledge, Indigenous knowledge, Expert knowledge, Modeling, Wildlife populations, Wildlife habitat

1. Introduction

The distribution, abundance, and habitat requirements of wildlife species is a knowledge priority shared by many people, communities, and organizations around the world. Biodiversity observations are used in species distribution modeling to assess impacts of global climate change (Austin and Van Niel, 2011; Bond et al., 2011) and species' overlap with localized anthropogenic impacts (Silva et al., 2017; Leu et al., 2008). Changes in population abundance over time are used to assess patterns and potential drivers of species population growth and decline (Franks et al., 2017; Busch et al., 2020) while changes in abundance across spatial gradients are used to delineate species abundance distributions (Acevedo et al., 2014), identify barriers to dispersal (Parker et al., 2016), infer habitat quality (Johnson, 2007; Holt et al., 2013), and help to prioritize habitat protection (Morris, 2003; Sebastián-González et al., 2010; Fulbright et al., 2013).

Wildlife science has become more quantitative over the last several decades (Michener and Jones, 2012; Brennan and Marcot, 2019) at a time when the importance of experiential wildlife knowledge beyond the quantitative domain has also become better recognized (Brook and McLachlan, 2008; Thornton and Scheer, 2012). The emerging emphasis on quantification, modeling, and big data within ecological, biodiversity, and wildlife sciences (Guthery, 2008; Blanco et al., 2012; Peters et al., 2014) has been referred to as "*datafication*" and interpreted as "*a shift in priorities in the ecological sciences - from concerns about localities and interaction milieu - to a focus on the emerging concept of global biodiversity… viewed as something that can be monitored, as an object of governance"* (Devictor and Bensaude-Vincent, 2016). At the same time, there is growing recognition of the need to democratize conservation science by "*broaden[ing] the definition of science to include multiple knowledge systems (e.g., traditional and local knowledge) and expand[ing] the practice of conservation science to include the participation and objectives of all those who wish to act collectively to support the stewardship of the biosphere*" (Salomon et al., 2018). The compatibility or incompatibility of these two trajectories - towards quantification (or datafication) and/or towards interweaving (Crabtree and

Klain, 2021; Hessami et al., 2021; Younging, 2018) of local knowledge and priorities - is an important and under-examined transdisciplinary challenge in wildlife and conservation science.

The experiential wildlife knowledge held by local people, communities, and Indigenous Peoples can improve understanding of wildlife populations and their habitat requirements (Shokirov and Backhaus, 2020; Su et al., 2020; Wilson et al., 2010; Low et al., 2009; Mavhura and Mushure, 2019; Popp et al., 2019). Local experiential wildlife knowledge had been shown to fill gaps in scientific understanding that may be difficult or impossible to obtain through other means (Brook and McLachlan, 2008; Popp et al., 2019), offer "multiple lines of evidence" (Service et al., 2014), identify and address seasonal, experience, and scale biases (Martinez-Levasseur et al., 2017), improve temporal transferability (Tuanmu et al., 2011), provide context for interpreting results (Abu et al., 2020), and enhance community support for and involvement in wildlife science (Salomon et al., 2018; Lute and Gore, 2014; Holsman et al., 2010) by remedying the sterile dichotomy between science and knowledge (Agrawal, 1995a; Agrawal, 1995b). Despite these many advantages, local knowledge inclusion and community partner involvement in wildlife science remains limited (Brook and McLachlan, 2008; Popp et al., 2019). Challenges to interweaving experiential wildlife knowledge in wildlife science may include skepticism in the scientific community (Gilchrist et al., 2005), the difficulty of identifying suitable knowledge holders (Davis and Wagner, 2003), the potential for local knowledge to be appropriated, marginalized, misunderstood, and misused (Nadasdy, 2021), how to assess the validity, reliability, bias and uncertainty of experiential wildlife knowledge (Gilchrist et al., 2005; Kadykalo et al., 2021), and determining how knowledge may be interwoven into science while maintaining the integrity of both knowledge approaches (Nadasdy, 2021).

Local, expert, and Indigenous knowledge can be characterized as "experience-based knowledge" (Brook and McLachlan, 2005) or "place-based knowledge" (Pascua et al., 2017; Reed et al., 2021; Zurba et al., 2019) intrinsically linked to place, sourced from personal experience, and held for the benefit of place and community (Berkes, 2017). Throughout this review, we will use the phrase *experiential wildlife knowledge* to refer to experienced-based or place-based knowledge, whether possessed by local people, Indigenous Peoples, landowners, land users, citizens, and/or experts, *knowledge holder* to refer to the people who have experiential wildlife knowledge, and *wildlife science* to refer to the acquisition and application of wildlife knowledge, whether that knowledge is experiential or more methodological, systematic, and quantitative in nature.

Although the distinctions and potential complementarity of experiential wildlife knowledge and other kinds of wildlife data, such as telemetry locations or population surveys, have been discussed frequently (Temple et al., 2020), the how-to challenge of interweaving experiential wildlife knowledge and quantitative habitat and population analyses has yet to be systematically reviewed. As wildlife and conservation science seeks to become both more quantitative and democratic this how-to challenge becomes more difficult and more important. Here we conduct a systematic review of primary literature that has included experiential wildlife knowledge into quantitative habitat and population analyses, to document the methods, successes, and limitations defining these attempts at experiential wildlife knowledge inclusion. We characterize study locations and study species, categories of knowledge holders involved and the stages of their involvement, the methods used to elicit and model their experiential wildlife knowledge, and the techniques used to accommodate potential bias and uncertainty. We also describe, in more detail, six case studies from the set of identified articles that exemplify common emerging themes and some of the successes, benefits, and limitations of experiential wildlife knowledge inclusion in quantitative wildlife science.

2. Methods

2.1. Systematic review methodology

Our search term string was defined and guided by six central themes (Table S1) that included: *Knowledge Holders*, focused on different types or groups of knowledge holders; *Inclusion*, with search string synonyms targeting articles that included or integrated knowledge; *Knowledge Area*; targeting wildlife focused articles; *Study Topic*, with search strings considered relevant to habitat use or population analyses; and *Modeling*, with search strings requiring articles to address some form of quantitative analysis. Boolean operators were used to include spelling variations, plural terms, and other string variations. A scoping phase was performed to guide a final decision on search terms. Several rounds of test-searches were done using different combinations of several search themes. Total numbers of results found were assessed, as well as quality and number of the most relevant papers from these searches. We did not include more specific wildlife names (e.g., moose, marsupial) or cultural names (e.g., Inuit, Sami) to avoid introducing regional, taxonomic, and cultural bias into our search terms. To avoid overlap with the recent review of local knowledge inclusion in aquatic and marine research (Dam Lam et al., 2019), we included a search theme to exclude the terms marine, fish, and aquatic. We focused on peer reviewed primary literature articles for this review and excluded grey literature.

We searched for English language articles published in any year in the Web of Science Core Collection. The search was conducted on September 13, 2020 and yielded an initial total of 2945 articles. Over 100 Web of Science Categories were represented by these articles, many of which categories were irrelevant to our search, which we attribute to the inclusion of the term "animal", which broadly applied to many unrelated fields such as biomedical research and laboratory science. The results were initially refined using Web of Science Categories by selecting articles which were classified within relevant categories, including wildlife, zoology, statistics, and social sciences, among others. For the 1607 articles remaining, titles and abstracts were read and an article was retained for further analysis if the following five requirements were met:

1. Primary literature, journal articles (reviews, books, and reports were excluded)
2. Studied terrestrial vertebrate wildlife at a species level (if multiple species were studied, habitat or population analyses had to be reported at a species-specific level)
3. Focused on habitat and population analyses,
4. Included local, Indigenous, or expert knowledge, and
5. Interweaved knowledge into a quantitative analysis during a pre-modeling, modeling, and/or post-modeling stage.

This screening reduced the article set from 1607 to 25 articles identified as meeting the five requirements and being suitable for the review. These 25 articles were then used as the source material for snowball collection to retrieve more articles. Articles that cited or were cited by the 25 articles retrieved from the systematic review were screened using the same sequence and protocol as described above and were retained if they satisfied the five criteria and had not already been identified. This snowball sampling added 24 new articles to the original set of 25, leading to 49 total articles identified by the systematic review and snowball sampling (Table S2) (Service et al., 2014; Abram et al., 2015; Alkhairy et al., 2020; Austin et al., 2009; Aycrigg et al., 2015; Aylward et al., 2018; Brittain et al., 2020; Brook and McLachlan, 2009; Clevenger et al., 2002; Crawford et al., 2020; Di Febbraro et al., 2018; Doswald et al., 2007; Evangelista et al., 2012; Evangelista et al., 2018; Froese et al., 2017; Gros, 1998; Irvine et al., 2009; Jordt et al., 2016; Kangas et al., 1993; Kellner et al., 2020; Kowalchuk and Kuhn, 2012; Leblond et al., 2014; Linde et al., 2012; Logan et al., 2015; Lopes et al., 2019; Lunney et al., 2009; Murray et al., 2009; O'Leary et al., 2009; Parry and Peres, 2015; Pearce et al., 2001; Pearman-Gillman et al., 2020; Pédarros et al., 2020; Phommachanh et al., 2017; Pillay et al., 2011; Polfus et al., 2014; Reza et al., 2013; Seoane et al., 2005; Skroblin et al., 2019; Smith et al., 2007; Taubmann et al., 2016; Tendeng et al., 2016; Turvey et al., 2015; van der Hoeven et al., 2004; Warren et al., 2016; Webb et al., 2019; Wilkinson and Van Duc, 2017; Yamada et al., 2003; Zeller et al., 2011; Ziembicki et al., 2013).

2.2. Summary & analysis

For each article, we identified and coded the following eight topics: i) general characteristics, ii) knowledge holder information, iii) experiential wildlife knowledge elicitation method, iv) form of experiential wildlife knowledge collected, v) quantitative analyses, vi) inclusion stage, vii) bias correction, and viii) model assessment. After reading through all articles and recording specific methods used, we determined suitable method sub-categories for each topic (i.e., within knowledge elicitation method category, subcategories included: interview, survey/questionnaire, participatory mapping, etc.). We recorded which sub-categories were employed by articles in spreadsheet tables for each topic and tabulated this predominantly categorical information for the analysis. To reduce the potential for author bias, sub-categories were identified using the same words as the authors, trying to avoid imposing our own interpretations as much as possible.

General characteristics included study area/site, year of publication, and species studied. Knowledge holder information categories reflected what the authors communicated about their location (whether they were local or non-local to the area where the wildlife were studied) and the nature of the experiential wildlife knowledge they held (Table S3). Descriptors/sub-categories for consulted knowledge holders included hunters or trappers, university-affiliated academic scientists, community members, wildlife managers, Indigenous Peoples, landowners, etc. Nine articles included knowledge holders we classified as “other”, because they involved a type of knowledge holder not consulted in any other article (e.g., industry stakeholders, local enthusiasts/naturalists, tour guides, local NGO, etc.) and which did not fit any other category well. Descriptors of knowledge holders could be intersectional, with one knowledge holder representative of multiple categories if they were described this way (e.g., Indigenous AND hunter/trapper AND local). An exception to this is the descriptors “Indigenous” and “Community Member”, which are mutually exclusive in our classification; knowledge holders were classified as “Indigenous” if the author described the knowledge holder to be Indigenous or Aboriginal (or belonging to a specific cultural group that self-identifies as Indigenous or Aboriginal) and as “Community Member” if the authors indicated they lived locally but made no mention of Indigenous-identity. Three articles did not provide enough information to discern which type of knowledge holders were involved and were therefore classified as “Unknown”.

Wildlife elicitation method described how knowledge was collected from knowledge holders, and we divided categories based on method (e.g., interview, participatory mapping, etc.) and location (e.g., in-person, online, mail, etc.) (Table S4). To assess at which stages of a study knowledge holders were involved, we defined five separate stages: *Consultation/Study Design*, in which knowledge holders were involved in planning study methodology including appropriate elicitation methods, what knowledge should be collected, how to include it, etc.; *Pre-Modeling/Analytical Approach*, in which knowledge holders were involved in developing model parameters such as study area, scale, or time frame, selecting model covariates or GIS layers to use, or directly developing statistical models; *Modeling/Data*, in which experiential wildlife knowledge in the form of observations, or quantitative information was directly included as model inputs; *Post-Modeling/Validation*, in which knowledge holders were involved in model validation, refinement, and re-parameterization; and *Follow-Up/Member Checking*, in which knowledge holders were engaged after analyses were completed to assess whether results appropriately interpreted and reflected their knowledge. We then determined whether the article described knowledge holder involvement in any or all of these stages (our results here indicate only what was discussed in the published article; it is possible that articles did not report all stages of inclusion). Different stages of knowledge holder inclusion were not necessarily conducted with the same knowledge holder, as different sets of individuals or types of knowledge holders were sometimes used in different phases of the study. To summarize bias correction and assessment, where methodologies are very context-specific and not easy to categorize without losing important nuance, we report the methodologies used in articles and discuss where trends or similarities occur.

Meta-analyses were not conducted for several reasons: i) not all papers conducted a quantitative assessment of their models; ii) assessing model success was not the focus of this review, and iii) the practice of quantitatively assessing experiential wildlife knowledge models, particularly through comparison to models based on independent data, has been criticized (Brook and McLachlan, 2005). As such, we focused on a qualitative synthesis of articles, summary tables and figures, and exploration of case studies. Six case study articles, which we describe in more detail, were chosen from the complete set of 49 articles based on the relevance and importance of each to the focus of our review as well as their collective diversity in helping to communicate the variety of approaches, opportunities, and challenges involved.

3. Results

3.1. Study location, date, and taxonomic coverage

The articles were globally distributed, including every continent except Antarctica (Fig. 1A). Countries with the most studies included Australia (10), Canada (8), and the United States (8). The articles spanned a publication timeline of nearly 30 years with the earliest article published in 1993 and more articles appearing in the last 10 years than in the 20 years prior to that (Fig. 1B). The majority of the articles (32 of 49) were single-species focused, with an additional 11 articles focused on 2–10 species, 3 articles focused on 11–20 species, and one article that focused on 50 species. The highest number of species studied was described by Aycrigg et al. (2015) as “over 6000 taxa”. This article, along with another that did not specify species to the genus level, are not included in taxonomic coverage summaries. A total of 128 genera were considered by the 47 studies that specified to genus level, including 2 amphibian genera, 4 reptilian genera, 16 avian genera, and 106 mammalian genera. Commonly studied mammal groups included even-toed ungulates (e.g., bovids, suids, and cervids), carnivores (e.g., canids, felids), primates, and elephants, with the most frequently studied mammalian genera including leopards or panthers (*Panthera*, 7 articles), red deer or elk (*Cervus*, 5 articles), and wolves or relatives (*Canis*, 5 articles) (Fig. 1C).

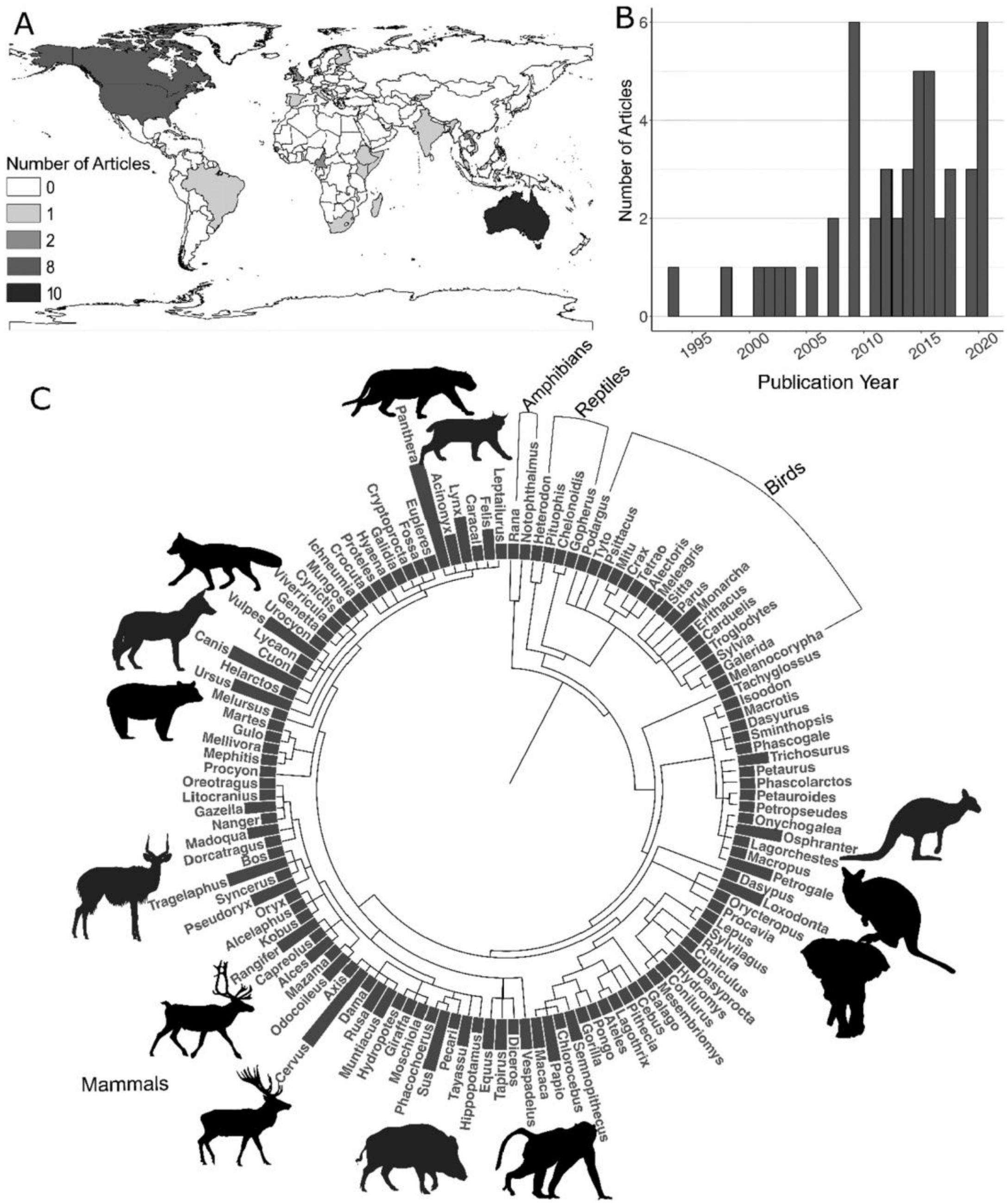

Fig. 1. Global distribution (A, articles per country), year-of-publication (B, articles per year), and taxonomic coverage (C, articles per genus arranged in a circle phylogeny based on National Center for Biotechnology Information (NCBI) Taxonomy (Schoch et al., 2020)) of journal articles describing inclusion of local experiential wildlife knowledge in quantitative analysis identified through a systematic review (date unlimited – 2020).

3.2. Knowledge holders

The number of knowledge holders that experiential wildlife knowledge was elicited from ranged from 1 to 16,526 with a median of 32. Most articles exclusively involved local knowledge holders (40 of 49 articles), but five articles involved a combination of local and non-local knowledge holders, and the remaining four articles relied solely on non-local experts (Fig. 2B). The most frequent descriptors for consulted knowledge holders were hunters or trappers (17 articles), university-affiliated academics (15 articles), and community members (14 articles; Fig. 2C). Less frequently included knowledge holder types included wildlife managers (9 articles), Indigenous People (6 articles) and landowners (5 articles).

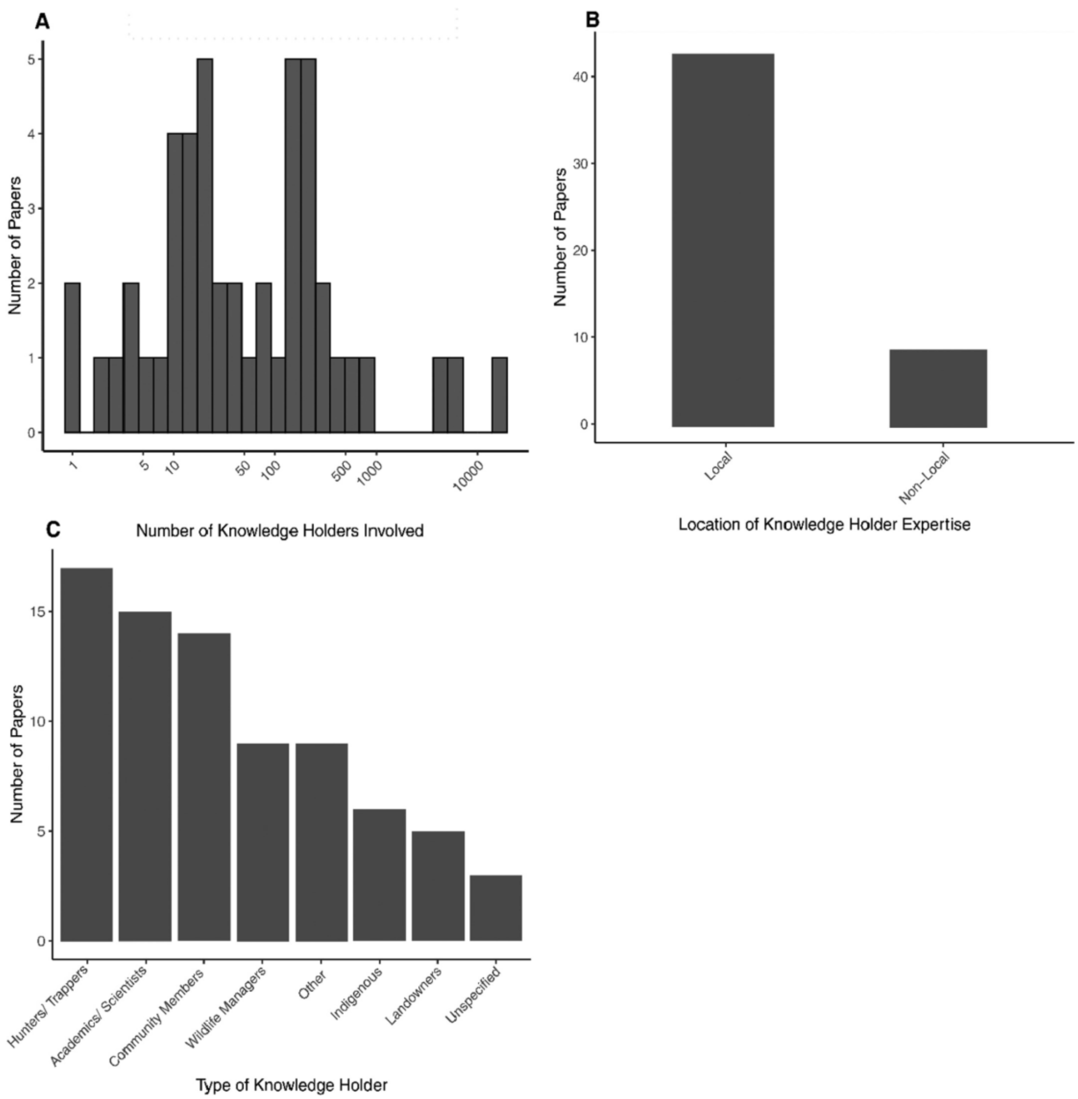

Fig. 2. Number of articles in systematic review including A) different numbers of knowledge holders, B) local and non-local knowledge holders, and C) different knowledge holder types.

3.3. Knowledge elicitation method

The most common methods of knowledge elicitation were interviews (26 articles) followed by surveys or questionnaires (18 articles; Fig. 3). Less frequent elicitation methods included participatory mapping sessions, workshops, and collecting pre-existing datasets, typically in the form of hunting records or observation records (Fig. 3). Most experiential wildlife knowledge was elicited in-person, usually in interviews, participatory mapping sessions, or in-person questionnaires or surveys, but some studies elicited knowledge online or through mail (Fig. 3.). Mail delivery was used with both large groups, small groups, or individual knowledge holders. For example, Lunney et al. (2009) sent over 100,000 participatory mapping forms across koala (*Phascolarctos cinereus*) range in Australia. Other articles sent mails surveys to several thousand landowners (e.g., Jordt et al., 2016), smaller numbers of registered hunters or trappers (e.g., Linde et al., 2012), and some others sent mail surveys to several specific individual experts (e.g., Gros, 1998). Two articles elicited experiential wildlife knowledge online using a web-based survey interface (Aylward et al., 2018; Pearman-Gillman et al., 2020).

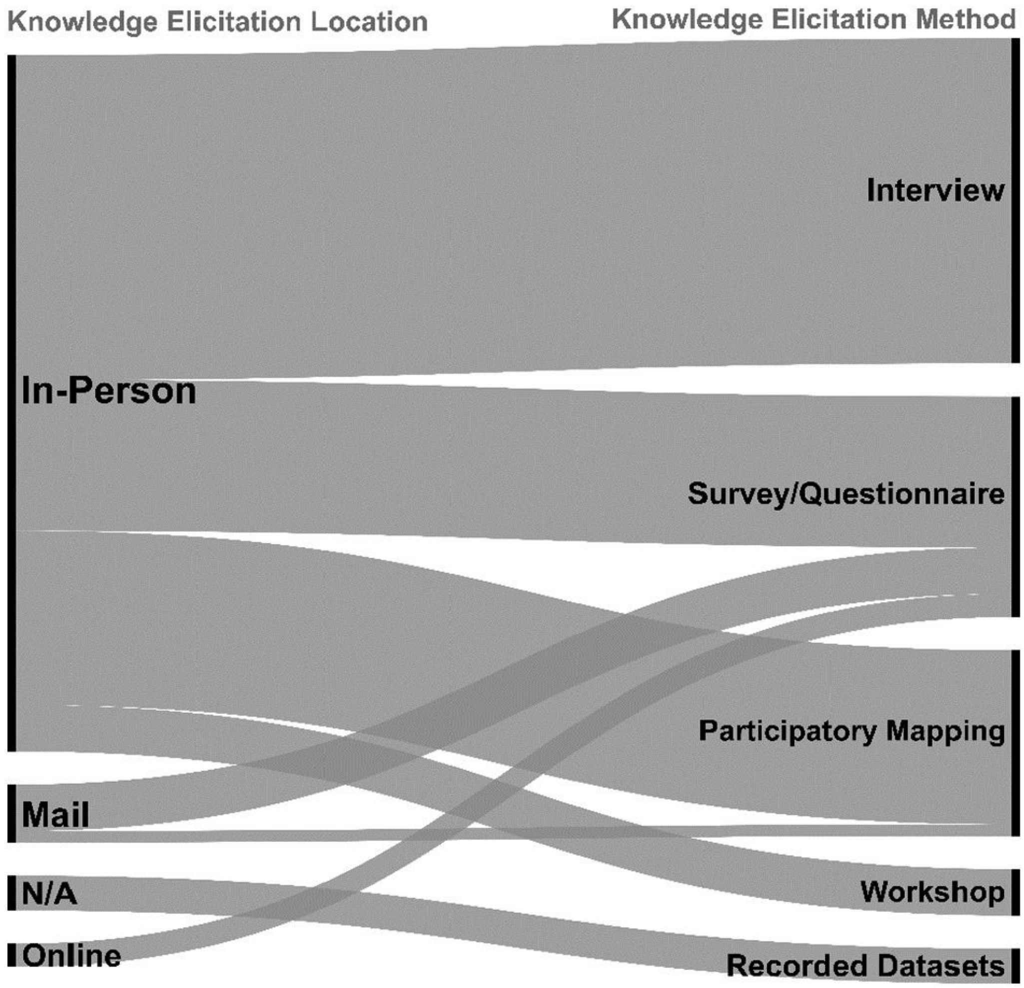

Fig. 3. Knowledge elicitation approaches presented as an association between location (left) and method (right). Width of black side bars indicate total frequency of locations and methods across all studies. Width of connecting bands indicates the frequency of each location and method combination. Four papers classified as “N/A" for location were due to the knowledge being collected through existing datasets, typically hunter records.

3.4. Stage of knowledge holder involvement

Most articles involved knowledge holders in only one stage (29 of 39 articles), less than half in two stages (16 articles), relatively few in three stages (5 articles) and none in more than three stages (Fig. 4). Knowledge holders were most frequently included in the modeling/data stage (44 articles), followed by pre-modeling/analytical approach (15 articles), consultation/study-design (6 articles), post-modeling/validation (4 articles), and follow-up/member-checking (3 articles). If knowledge holders were involved in two stages, it was most often in the pre-modeling/analytical-approach and modeling/data stage (8 articles) and only one article included knowledge holders in both the consultation/study design and follow-up/member-checking stage.

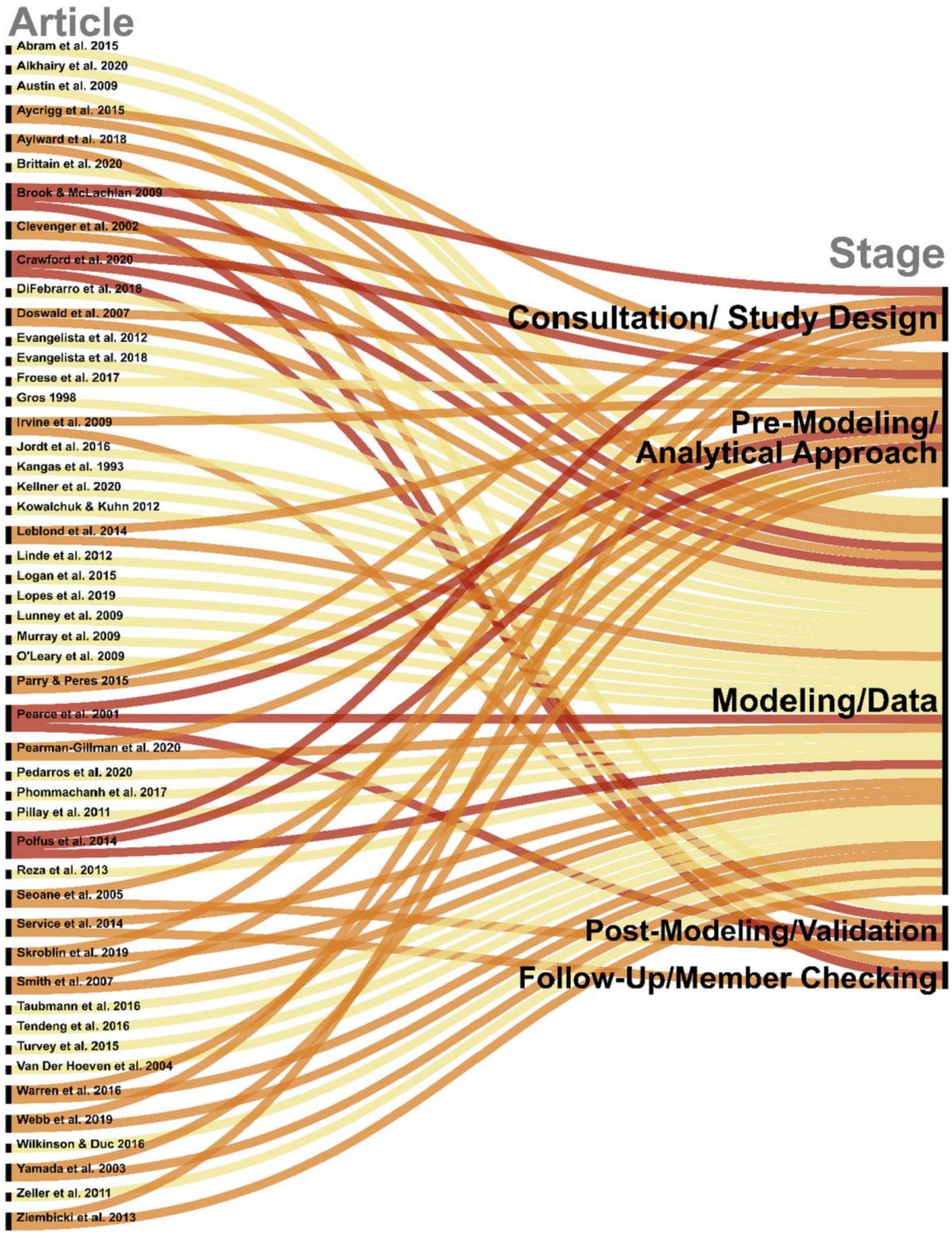

Fig. 4. Alluvial chart depicting knowledge holder inclusion across five study stages. Colors represent the number of stages knowledge holders were included (yellow = 1 stage, orange = 2 stages, red = 3 stages). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

3.5. Experiential wildlife knowledge into quantitative models

Multiple combinations of knowledge form, statistical models, and general model topics occurred across the review articles (Table S5, Fig. 5). The most frequent combination was habitat models produced by "other regression" (neither Bayesian nor mixed model regressions) statistical methods using knowledge in the form of point observations. The next most frequent combination was habitat models produced by Generalized Linear Models (GLMs) or Generalized Linear Mixed Models (GLMMs) using knowledge in the form of either model covariate ranking, values, or coefficients, or in the form of habitat relationship networks.

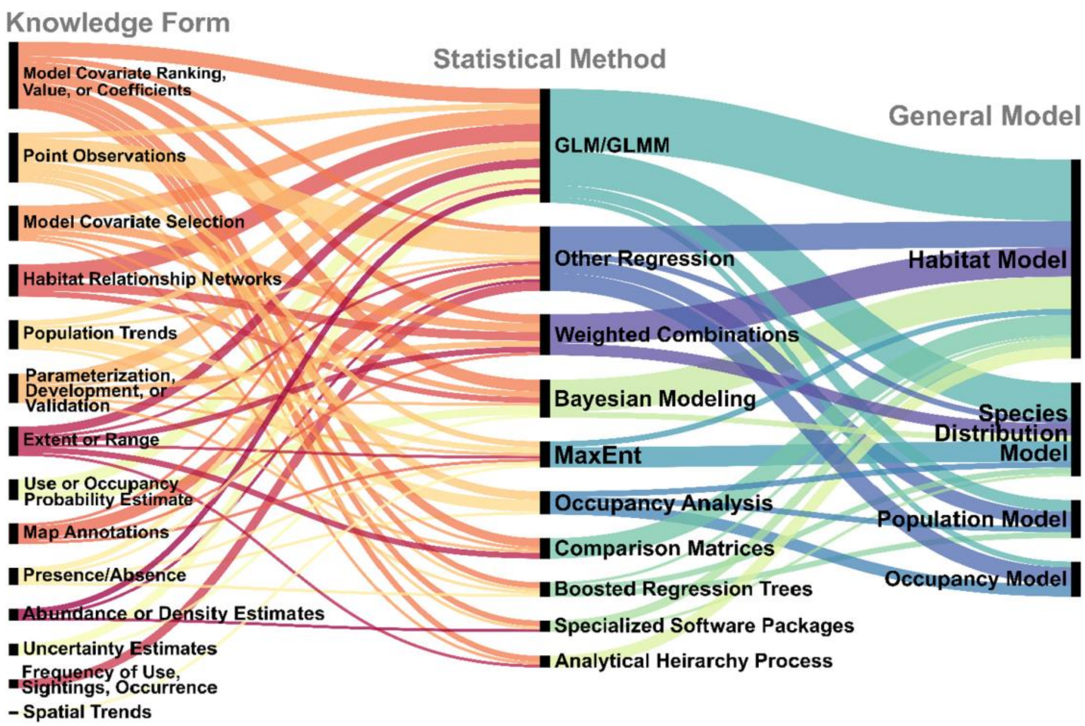

Fig. 5. Alluvial chart depicting relationships between knowledge form (left), statistical methods (middle), and general models (right). Width of the bars indicates frequency of occurrence across all articles, and width of bands between bars represents occurrence frequency for combinations

of knowledge form and statistical method or statistical method and general model. Colors highlight connections but do not indicate magnitude or other distinctions.

Habitat models, focused on predicting or modeling habitat use or quality by species, were most prevalent across articles. Many statistical methods were used to perform these models, with the most frequent being GLMs or GLMMs, Bayesian Models, other regressions, and weighted combinations. Articles focused on habitat models typically collected knowledge in the form of i) observations or occurrence, ii) extent of distribution or presence/absence in certain areas, iii) selecting or informing key habitat covariates in the models, or iv) estimating habitat covariate ranking, importance, weighting, or coefficients. Other forms of knowledge contributing to habitat models included annotated maps, information on spatial and temporal trends, building habitat relationship networks, habitat covariate use estimates, and parameterizing or developing models. Species distribution models were similar to habitat models and employed similar statistical methods and knowledge forms, with the addition of Maximum Entropy (MaxEnt) methods being frequently used. Articles that focused on population modeling typically employed regression methods and collected knowledge in the form of observations or occurrence, information on population trends over time, and estimates of abundance or frequency. Most spatial or mapping methods were performed in ArcGIS (ESRI, 2011) or Maxent (Steven et al., 2017). Statistical analyses were generally performed in R (Core Team, 2020), STATA (StataCorp, 2021), SAS software (SAS Institute Inc, 2013), or PRESENCE (MacKenzie and USGS, 2021). Additional studies developed models with more specialized software packages including InVEST (Di Febbraro et al., 2018; Sharp et al., 2018), FunConn (Evangelista et al., 2012; Theobald, 2006), SwiColBM (Jordt et al., 2016; Lange et al., 2012), and PageRank simulations (Wilkinson and Van Duc, 2017).

3.6. Bias correction

Articles describing methods used to compensate for potential bias or error in knowledge-based information applied these methods during knowledge holder selection, knowledge elicitation, or in the modeling stage. Methods to reduce bias that focused on knowledge holder selection included identifying reliable experts using focus groups (Brittain et al., 2020), selecting respondents based on ability to identify species and/or species presence (Abram et al., 2015;

Gros, 1998; Parry and Peres, 2015), selecting respondents based on hunting experience with the focal species (Linde et al., 2012), deliberately selecting respondents who did not specifically see the species (Turvey et al., 2015), or other methods of establishing reliability or “vetting” respondents (Phommachanh et al., 2017; Zeller et al., 2011; Ziembicki et al., 2013). Methods that focused on reducing bias through interview questions included interviewing knowledge holders individually to prevent audience-effect bias (Brittain et al., 2020), phrasing questions neutrally (Brittain et al., 2020), concealing the focal species during interviews (Brittain et al., 2020; Turvey et al., 2015), using the “interview funnel approach” (Brittain et al., 2020), and openly discussing the research objectives and species prior to interviews (Parry and Peres, 2015). Methods to reduce bias focused on the knowledge collection stage included validating reliability by asking respondents to repeat their reports of species detections at the end of the interview and removing unsure or inconsistent responses (Brittain et al., 2020), providing model feedback to experts during knowledge elicitation sessions to reduce cognitive bias (O'Leary et al., 2009), screening the dataset for reliability by cross validating interviewee responses and removing knowledge holders who gave incompatible answers as well as removing brief responses (Abram et al., 2015), and cross examining key knowledge holders (Pillay et al., 2011). After knowledge collection, statistical methods used to reduce bias in data included: using triangulation techniques to verify quality and respondent reliability (Abram et al., 2015), evaluating consistency and reliability matrices based on consistency ratio (Doswald et al., 2007; Kangas et al., 1993), assigning knowledge holders a reliability score and using that to filter or remove potentially biased data (Gros, 1998; Logan et al., 2015). Modeling methods of bias correction included using mixed-model approaches (Pearman-Gillman et al., 2020), highly weighting responses or sites where confidence was strong (Pearman-Gillman et al., 2020), entering respondents into models as a random effect or intercept (Aylward et al., 2018; Lunney et al., 2009), incorporating reliability scores into models (Gros, 1998; Logan et al., 2015), incorporating road, distance, or site-accessibility as a covariate to account for accessibility bias (Pédarros et al., 2020; Skroblin et al., 2019), or standardizing observations (Service et al., 2014).

3.7. Uncertainty

Experiential wildlife knowledge uncertainty was most frequently interweaved using Bayesian modeling (Alkhairy et al., 2020; Froese et al., 2017; O'Leary et al., 2009) or Bayesian Belief

Networks (Smith et al., 2007). Some articles directly elicited uncertainty estimates from knowledge holders by having experts define their own overall uncertainty in their judgements (Froese et al., 2017; Pearman-Gillman et al., 2020) or estimate uncertainty in the form of standard deviation (Aylward et al., 2018). In other articles, the authors estimated knowledge holder uncertainty by estimating "reliability codes" for data representing confidence in the information correctness (Ziembicki et al., 2013) or by estimating expert impact and confidence for each variable and converting them to Bayesian priors (O'Leary et al., 2009). Other methods included averaging model parameters to incorporate model selection uncertainty (Logan et al., 2015), including variation or bias in knowledge holders responses by incorporating individual knowledge holder as random effects in mixed-effects models (Aylward et al., 2018; Lunney et al., 2009; Pearman-Gillman et al., 2020).

3.8. Model assessment

Around half of the articles (24 of 49) described attempts to validate or compare experiential wildlife knowledge to other sources of information. Most commonly, model outputs or predictions were compared to either an independent dataset or the dataset used in the authors model that was derived from telemetry locations or other population surveys (11 articles). Specific statistical methods included area under curve (AUC) (3 articles), Kappa Index (3 articles), Boyce Index (2 articles), sensitivity analysis (2 articles), and spearman rank coefficients (2 articles). Numerous other methods were represented in individual articles, including contrast validation index, correct classification rate, deviance information criterion (DIC), field validation, goodness of fit, jack-knifing techniques, Mann-Whitney statistics, marginal and conditional R (Bond et al., 2011), specificity analysis, true skill statistic, and z-test.

Authors generally described the outcomes of these assessments in terms of consistency with independent data or similarity to models developed from other data such as GPS, VSH, or field-survey data. While this metric of success is flawed (see introduction and discussion), here we report the authors self-described assessment outcomes. Of the articles who assessed performance, many reported that models interweaving wildlife knowledge moderately to very accurately matched independent or original data, or performed satisfactorily to very well (Abram et al., 2015; Di Febbraro et al., 2018; Doswald et al., 2007; Froese et al., 2017; Jordt et al., 2016;

Leblond et al., 2014; Pearman-Gillman et al., 2020; Polfus et al., 2014; Wilkinson and Van Duc, 2017). Some authors reported that models interweaving wildlife knowledge performed differently under different circumstances (Evangelista et al., 2018; Irvine et al., 2009; Murray et al., 2009; Pearce et al., 2001; Smith et al., 2007), with some reporting differences in the predictive accuracy of high and low suitability sites (Smith et al., 2007), or between two different kinds of models (e.g., Maxent and Boosted Regression Tree (Evangelista et al., 2018)), or based on the scale of comparison (Irvine et al., 2009). Some authors found that experiential wildlife knowledge lowered accuracy of models (Evangelista et al., 2012) or reported that the accuracy of pure experiential wildlife knowledge models was less than independent data alone (Pearce et al., 2001) and that increasing experiential wildlife knowledge input resulted in decreasing predictive ability (Seoane et al., 2005). Articles that developed and compared both experiential wildlife knowledge and independent data models report mixed results, with some reporting significant agreement between datasets (Brook and McLachlan, 2009; Clevenger et al., 2002; Pédarros et al., 2020; Tendeng et al., 2016) and others finding few similarities between wildlife knowledge and independent data based models (Kowalchuk and Kuhn, 2012).

4. Case studies

4.1. Murray et al. (2009): an example of distribution analysis and bayesian logistic regression using expert knowledge

Murray et al. (2009) aimed to assess uncertainties in expert opinion using Bayesian logistic regression, using three Bayesian statistical models: 1) expert-only model, 2) field-data only model, and 3) combined expert and field data model. Nine experts with backgrounds in conservation management and knowledge of rock-wallaby (*Petrogale* spp.) ecology and habitat were included, some of which was local experience. A GIS software tool was used to facilitate experiential wildlife knowledge inclusion into modeling. Regression coefficients were obtained by asking questions to assess the probability of presence at sites with known habitat characteristics. Coefficients were then used as prior distributions suitable for Bayesian analysis. Bayesian logistic regression, performed in WINBUGS 1.4 (Lunn et al., n.d.), was used to fit models via Markov chain Monte Carlo simulations. The authors found differences in posteriors formed from expert- informed informative priors and non-expert-informed non-informative priors, indicating that experts were contributing information that extended beyond the field-collected data. The deviance information criteria (DIC) showed that a combination of field data and expert-informed priors provided improved goodness-of-fit relative to non-expert informed models. Predictive performance checks indicated that models built on expert knowledge and field data performed consistently well, however all three models performed well overall. They concluded that expert knowledge, in the form of informative priors in Bayesian modeling, enhanced estimates.

4.2. Polfus et al. (2014): an example of HSIs, RSFs, and indigenous experiential wildlife knowledge

Polfus et al. (2014) developed a Traditional Ecological Knowledge (TEK) model and a “western science” model to study caribou habitat. These models were then compared using k-fold cross validation and spatial Kappa statistics to assess differences between models. TEK was collected through in-person, individual, semi-directed (i.e. interviewer had loosely structured questions and allowed topic to follow natural course) interviews with 8 experts, including hunters, gatherers, and community elders. TEK collected information including seasonal use and food resources, drawings of important areas, animal locations, and habitat requirements such as land cover types, habitat associations, seasonal foraging, and other seasonal resources. Habitat descriptions in TEK were linked to spatial resource covariates in the habitat model. From these variables, rule-based Habitat Suitability Index (HSI) models were developed for summer and winter. Models were built by applying these values to combinations for ecological conditions in ArcGIS by overlaying raster layers into one final map layer. The authors then used independent VHF collar data to assess the predictions made by the knowledge-based HSI model and the collar data-based Resource Selection Function (RSF) model. Spearman's rank correlation was used to assess the strength of association between knowledge and data. Further visual examination was used to assess discrepancies between the RSF and HSI models, and a weighted Kappa statistic was further used to statistically compare the two. The authors found high correlation between the RSF model and caribou locations, as well as between the HSI model and caribou locations. Visual comparison found that most discrepancies arose from the RSF predicting more high value habitat than the HSI. The Kappa statistic indicated generally strong spatial agreement between the knowledge based HSI and the data-based RSF.

4.3. Abram et al. (2015): an example of population trend analysis, boosted regression trees, and community expert knowledge

Abram et al. (2015) aimed to reconstruct recent population trends of orangutans (*Pongo pygmaeus*) in Indonesia. Experiential wildlife knowledge was collected in questionnaire-based interviews with villagers, and included the number of individuals seen in the previous year and their locations as well as perceptions of population change over times. Reportings of the number and location of recent observations produced georeferenced presence-only occurrence data derived from sightings. Reporting of perceived population change generated population trend responses that were categorical in nature, with respondents asked to indicate whether the contemporary population compared to 10 years ago was 'more than now' (1), 'same as now' (2), 'fewer than now' (3), 'locally extinct' (4), or 'never seen orangutans here' (5), with the associated numbers representing the values the answers were coded as in the dataset. Questions were converted to continuous response variables and condensed to village averages. Predictive modeling was performed on the data, with response variables including frequency of sightings and perceptions of orangutan populations. ArcGIS was used to average survey responses and allocate 39 spatial predictor variables to each village coordinate. R-based Boosted Regression Trees (BRT) were used to develop predictive models from the summarized ArcGIS output. Model outputs were mapped using ArcGIS, and the correlation between observed and predicted values was used to assess predictive performance of the models. Abram found that the BRT model for orangutan sightings performed "well", and the BRT model on perceived orangutan population change performed "excellently".

4.4. Smith et al. (2007): an example of Bayesian Belief Networks to produce habitat suitability models using scientist expert knowledge

Smith et al. (2007) aimed to demonstrate how Bayesian Belief Networks (BBNs) developed by a small group of experts could be used to study habitat suitability of Julia Creek dunnart (*Sminthopsis douglasi*) in Queensland, Australia at a region scale. The habitat suitability model was developed in two stages: i) conceptual model development and ii) creation of predictive models from conceptual models. Conceptual model development was conducted to build an influence diagram that depicted important environmental variables for habitat. The process was broken into four main steps: 1) literature review, based on published literature, theses, and research reports, 2) a meeting with a region and species expert used to build a draft influence diagram from the review findings and expert information (with the influence diagram being composed of habitat variables that influence suitability, GIS variables that could represent those key habitat variables, and environmental variables that influence key habitat variables), 3) surveys sent to 10 ecologists with specific expertise in the region and species so that their opinions on the draft could be collected and the diagram revised based on the feedback, and 4) the revised draft was then sent to two other experts for final review, and was continually revised until both of these experts were satisfied with the influence diagram. Predictive model development was done in four steps: 1) converting the revised influence diagram from the first stage into a BBN using Netica™ software (Norsys Software Corporation, 1998), 2) scenarios were constructed from the different node combinations, with associated predictions and probabilities, 3) sensitivity analysis was performed to assess relative influence between variables using Netica's entropy reduction to measure sensitivity, using expert consultation, and 4) the BBN was developed into a habitat suitability model using ArcGIS. The authors assessed accuracy of the BBN derived habitat suitability model through comparison to field survey data using the error matrix method and the Kappa statistic. They found high accuracy of model predictions with overall accuracy being 89%, however this varied based on site quality, with low quality sites predicted better than high quality sites.

4.5. Pearce et al. (2001): an example of multi-stage inclusion of experts in distribution analysis using logistic regression

Pearce et al. (2001) investigated several approaches of incorporating expert opinion into species distribution models at different stages for 16 species including reptiles, birds, marsupials, and bats in Australia. A panel of three experts were included in the pre-modeling, model-fitting, and post-modeling stages. Expert knowledge was included by: i) modifying or refining existing statistical models by specifying additional rules in order to refine predictions to better reflect their knowledge, ii) deriving vegetation index maps and developing a new GIS layer for each species and defining the habitat-value indices, iii) selecting predictor variables for each species, specifying the form of the relationship between species and variable, and refitting the GAM model to reflect their choice, and iv) creating models based purely on expert opinion by developing models by combining available GIS layers and creating new vegetation variables as needed. The authors assessed predictive accuracy as varying significantly between models. Multiple Pairwise Comparisons suggested that models which were developed using only expert-defined rules performed significantly worse than models which included less expert-opinion. They determined that models derived from knowledge-based vegetation indices were not significantly more accurate and concluded that "expert modification of fitted statistical models should be confined to species for which models are grossly in error, or for which insufficient data exist to construct solely statistical models".

4.6. Skroblin et al. (2019): an example of Indigenous Knowledge based Distribution models using Maxent

Skroblin et al. (2019) aimed to assess whether Species Distribution Models (SDMs) including Indigenous Knowledge (IK) or field survey only data produced similar predictions on greater bilby (*Macrotis lagotis*) in Western Australia. Collected experiential wildlife knowledge included spatial information on where species may be present, perceptions on whether species distribution has changed, and where suitable habitats were. Data were collected through interviews and participatory mapping, whereby respondents would provide spatial information by annotating maps. Maps were then digitized in ArcGIS to create spatial polygons of occurrence. IK maps were converted to point data by sampling random points in the IK polygons to adjust IK into a format that could be used by Maxent to produce SDMs. Two Maxent models were run using IK and field-survey data, and were then evaluated using the area under the receiver operating curve (AUC) to assess model performance. They found that the AUC of the field-survey model was higher than that of the IK model and joint models, indicating higher performance, however it was indicated that the field-survey model may have overfitted data compared to the IK model. The predictive maps of habitat suitability differed among data types.

5. Discussion

5.1. Search results

This systematic literature search focused on key words related to knowledge holders, inclusion, wildlife populations/habitat use, and quantitative analysis to identify nearly 3000 candidate articles. Of these systematically identified articles, only 25 satisfied all eligibility criteria. Snowball sampling of articles that were cited by or cited these 25 identified an additional 24 eligible articles. Thus, the current systematic review is based on 49 primary literature articles incorporating local or expert knowledge into a quantitative analysis of terrestrial vertebrate populations or habitats. A potential limitation of our search string is the absence of taxa-specific terms (e.g., bird, waterfowl, goose, mammal, cat, leopard) or culture-specific terms (e.g., Maori, Maasai, Inuit, Cree) that may have been used by authors in place of more generic terms like "wildlife" or "Indigenous". However, we did not include these to minimize the regional and species biases associated with their inclusion as it would be infeasible to include all appropriate terms for all taxa, regions, and cultures. Although snowball sampling of citing and cited articles identified additional articles missed in the systematic search, it remains likely we missed some eligible articles that omitted targeted search terms such as wildlife and Indigenous. A further limitation is our focus on peer-reviewed journal articles, which excludes reports, reviews, and project summaries published in other domains that may be more conducive to diverse knowledge inclusion in wildlife sciences.

The study locations, focal species, and knowledge holders identified through our review were inter-related, with the most common study species having high relevance and accessibility to the most common knowledge holders, and the most common knowledge holder type reflecting the most studied species and study areas. Although we identified that experiential wildlife knowledge has been included in quantitative analysis of wildlife populations and habitats across many parts of the world and across a wide diversity of terrestrial wildlife, we found that most articles were from the United States, Canada, and Australia, involved harvested birds and mammals, and most often solicited the knowledge of hunters and trappers. North America and Australia are regions from which there is generally high biodiversity research output (Trimble and van Aarde, 2012), meaning that these areas do not necessarily focus disproportionately on inclusion of experiential wildlife knowledge, and the high representation of academic experts in the review may be related to this. Indigenous Knowledge Holders were concentrated in Canada and Australia, and the inclusion of Indigenous or Aboriginal knowledge is consistent with the legal and social impetus in these countries for increased consultation and involvement in wildlife and natural resource management and research (Gilchrist et al., 2005; Lawrence and Macklem, 2000). Beyond these key regions, articles were globally distributed, including all continents except Antarctica. Thus, inclusion of experiential wildlife knowledge in wildlife science is a global phenomenon and has applicability in numerous regions worldwide (Brook and McLachlan, 2008). While a wide variety of taxa were studied, the most frequently studied species were game species or large charismatic carnivores. This could be attributed to the cultural and dietary importance of many genera in these taxa, particularly in the United States, Canada, and Australia (Titus et al., 2009; Hewitt, 2015; Krause and Robinson, 2021) where hunting is common and survey data is systematic and available (Arnett and Southwick, 2015; White et al., 2015; Sharp and Wollscheid, 2009). It could also be attributed to the high scientific and public interest in these species; of the most studied genera in the review which were represented more than three times across our articles, all are on a list of the 20 most charismatic taxa developed by Albert et al. (2018). Biases towards collecting more observational, person-based data on charismatic species dates back for centuries (Monsarrat and Kerley, 2018). There is also a high overlap between scientific and social interests regarding charismatic species (Jarić et al., 2019). Furthermore, familiarity with many of these large and charismatic species is high (Ulicsni et al., 2019) which could facilitate high levels of experiential wildlife knowledge in local and community knowledge holders.

5.2. Benefits

Benefits of including experiential wildlife knowledge in wildlife science that emerged across the review include experiential wildlife knowledge providing information that may be challenging or impossible to obtain through other data, increasing the rigor of models build on GPS collar, VHF, or field survey data, through guiding model development, providing context and improving interpretation of findings, decreasing costs of data collection, and increasing transferability of findings across long periods of collection (Table 1). We focused our analysis on the benefits that experiential wildlife knowledge provided to wildlife science and did not address benefits this process had to knowledge holders, because there was not enough information in the articles to extract and categorize those benefits. In some cases, knowledge holders provided information that would have been challenging or impossible to obtain through other means (Abram et al., 2015; Brook and McLachlan, 2009; Pearman-Gillman et al., 2020), particularly for rare or poorly documented species (Pearman-Gillman et al., 2020). Experiential wildlife knowledge was frequently discussed as a means to improve data-based models, with knowledge holder models providing the "backbone" for other models and making them more robust (Aycrigg et al., 2015; Brittain et al., 2020), improving or expanding upon other data (Brook and McLachlan, 2009), and identifying areas of weakness and guiding improvements in other models (Pearce et al., 2001). Experiential wildlife knowledge frequently identified new issues or areas of study, provided context, and improved understanding of model results beyond what would have been achieved without knowledge holder involvement (Brittain et al., 2020; Brook and McLachlan, 2009; Phommachanh et al., 2017). The comparatively low cost of experiential wildlife knowledge solicitation relative to other data collection was also discussed (Pearman-Gillman et al., 2020). Finally, the long time scales across which experiential wildlife knowledge is accumulated may offer additional advantages, including improved temporally transferability relative to models collected using single year data (Tuanmu et al., 2011) and the development of models more suited to future projection.

Table 1. Summarized benefits, limitations, and recommended improvements for interweaving local, expert and Indigenous wildlife knowledge into wildlife science.

Benefits

- Inherently recognizes the validity of diverse knowledge in science
- Increases knowledge holder diversity and trust in science and management
- Improves equity between scientists and other knowledge holders
- Provides additional information that improves or expands other data
- Useful for rare or under-studied species, with relatively low cost of acquisition
- Identifies points of consensus and disagreement as well as knowledge gaps
- Improves temporal transferability of models

Limitations

- May be exclusively local in scale
- May not be systematic in coverage
- Possible scale mismatches between different knowledge forms
- Interviewer and/or respondent reliability difficult to assess
- Potential biases introduced by interviewers and/or respondents

- Many wildlife scientists lack social sciences/qualitative methods training
- Communication/collaboration challenges due to differing knowledge priorities, language, worldview

Improvements

- Avoid assuming that quantitative data can be used to assess the reliability of other knowledge forms
- Use multiple statistical methods to assess congruence or disagreement between data sources
- Develop standardized methods to accommodate uncertainty and observer reliability
- Meaningfully include knowledge holders in more study phases, including study design and member checking
- Discuss intellectual property rights, knowledge ownership, and knowledge protection
- Acknowledge and discuss power differences between researchers and knowledge holders
- Assess and communicate knowledge holder benefits or negative outcomes in addition to science outcomes

5.3. Limitations

Common limitations to inclusion of experiential wildlife knowledge that emerged across the review were related to scale, reliability, bias, subjectivity, uncertainty, and author unfamiliarity with local knowledges (Table 1). Scale limitations include concerns including: experiential wildlife knowledge may be highly local in nature, it may be collected and applicable at a smaller scale than is desired for some projects (Doswald et al., 2007), and large regions of focus proved challenging to some experiential wildlife knowledge models in this review (Pearce et al., 2001). Other literature has discussed that experiential wildlife knowledge may not be systematic in coverage (Moller et al., 2004), and that Indigenous knowledge in particular does not resonate with the short temporal and large spatial scales at which most research or management projects are conducted (Wohling, 2009). Other authors contradicted this by arguing that experiential wildlife knowledge was well suited to large, particularly remote areas (Brittain et al., 2020; Pearman-Gillman et al., 2020). Many of the challenges of scale attributed to experiential wildlife knowledge may not reflect its own inherent limitations, but rather the mismatch of the scale of experiential wildlife knowledge with the scale of other data. For example, when habitat covariate data such as GIS layers are produced at a much broader scale than very local, experiential wildlife knowledge, utility is limited and using them both is a challenge (Bauder et al., 2021). Both interviewer and respondent reliability were identified as potentially problematic (Abram et al., 2015), and personal bias was described as impacting results and interpretation (Pearman-Gillman et al., 2020). The reliability of experiential wildlife knowledge depends critically on the knowledge elicitation method (Pearman-Gillman et al., 2020) and knowledge holder access to and familiarity with different parts of the study area. Another reliability challenge is that most ecologists interested in collecting and analysing experiential wildlife knowledge lack formal social science training, meaning they have limited experience interpreting and analysing qualitive information (Brook and McLachlan, 2005) and limited understanding of how to effectively account for identity, bias, and positionality in study design (Shank, 2002). Some authors believed that statistical methods could sufficiently accommodate the inherent subjectivity of experiential wildlife knowledge (Leblond et al., 2014), but others raised concerns that this might not be the case (Aylward et al., 2018). A final limitation we discuss here is a requirement of relevance and relatability between wildlife science objectives and available experiential wildlife knowledge, where wildlife science and experiential wildlife knowledge may lack common priorities or have major differences in worldviews. For example, knowledge holders may have different interpretations of taxonomy or levels at which species are distinguished (Berkes and Mackenzie, 1978; Newmaster et al., 2007; Phaka et al., 2019). Ziembicki et al. (2013) discussed that many species they intended to study had to be collapsed into larger groups or eliminated because knowledge holders did not differentiate some animals to the species level, or did not collect specific knowledge on some species where they had no cultural or dietary relevance. Experiential wildlife knowledge is likely to be strongest and most robust for species that are recognized and culturally important to knowledge holders (Brook and McLachlan, 2009; Ziembicki et al., 2013; Monsarrat and Kerley, 2018).

5.4. Comparison and assessment

Articles identified in this review frequently assessed the experiential wildlife knowledge models through comparison to independent, often quantitative, data sets, but this validation approach has been criticized and alternate methods to assess experiential wildlife knowledge should be considered. In their review of local knowledge inclusion in ecological literature, Brook and McLachlan (2005) discuss the tendency to use "*ecological data as a test to determine the reliability of Local Ecological Knowledge*". They argue that articles frequently fail to appropriately discuss the "assumptions, limitations, or constraints of the ecological articles that they use". Data such as telemetry locations or field-surveys introduces its own error, bias, and incomplete representation (Brook and McLachlan, 2005; Rykiel, 2001). For example, data that are limited in spatial and temporal scope may provide a de-contextualized snapshot of animal ecology with poor population-level inference (Hebblewhite and Haydon, 2010). Thus, it is becoming better recognized that independent data do not offer an opportunity to validate experiential wildlife knowledge, but that there is an opportunity to combine both forms of experiential wildlife knowledge in a manner that advances understanding of the ecological system and each source of information (Polfus et al., 2014). Indigenous experiential wildlife knowledge in particular arises from distinct worldviews and ways of understanding ecosystems and wildlife relative to those involved in the design and collection of data (Bohensky and Maru, 2011). Difficulties with Indigenous knowledge inclusion in particular can be addressed through "*reframing integration as a process in which the originality and core identity of each individual knowledge system remains valuable in itself, and is not diluted through its combination with other types of knowledge*" (Bohensky and Maru, 2011). Methods such as "two-eyed seeing" (Reid et al., 2020) are increasing in use and may help to achieve this goal.

5.5. Improvements

Experiential wildlife knowledge inclusion in quantitative analyses can be further improved in future works through multi-model inference, better accounting of variation, bias, and uncertainty of experiential wildlife knowledge, engagement of knowledge holders in multiple study phases, and further consideration of intellectual property rights and power dynamics of experiential wildlife knowledge and policy (Table 1). Comparing the success of different models is challenging when each article applies a single statistical method to a unique study area, focal species, knowledge holder category, and experiential wildlife knowledge type. Future works may benefit from performing several statistical models on the same set of observations to assess how model selection impacts success of experiential wildlife knowledge integration (Polfus et al., 2014). Furthermore, additional work to develop standardized techniques for accommodating expert uncertainty may be beneficial. Many authors addressed uncertainty, and challenges with quantifying bias and variation in responses as a primary concern. Another area that future studies can improve upon is knowledge holder engagement in earlier and later phases of the research process and, if this occurs, clearer presentation of the form and outcomes of these engagements in research articles. Few articles discussed whether knowledge holders were involved in the co-design of study approaches and knowledge elicitation methods and few described member-checking or results dissemination after local knowledge was collected. While many authors communicated satisfaction with results of the knowledge integration process, the lack of member checking or follow-up interviews makes it challenging to verify if the knowledge holders shared their assessment. Brook and McLachlan (2005) state that "*if local knowledge is to be used in a respectful way that recognises its inherent and use-value, community members should be meaningfully involved in most, if not all, aspects of a study*". Finally, more discussion of the intellectual property rights, knowledge ownership, and control over the resulting data is warranted (Brook and McLachlan, 2005). For experiential wildlife knowledge inclusion to be effective, co-management and research requires equitable partnerships and sharing of information and power (Popp et al., 2019).

6. Conclusion and implications

Through this review, we aim to provide a resource for future research teams from which to begin designing projects that meaningfully include experiential wildlife knowledge into analysis. By reporting all categories and sub-categories of methodologies found in articles, we provide a portfolio from which researchers may observe a plethora of options and select the most suitable methods or strategies. Our case studies may also provide a brief framework from which to begin planning or brainstorming a future study design, while also informing readers of the authors self-reported assessment of success using such methods. By doing this, we hope to assist teams already incorporating experiential wildlife knowledge by exploring and presenting other options, and to assist teams just beginning to consider this valuable area of science or work with knowledge holders by providing a toolbox of resources and references for consideration. The growing publication output of experiential wildlife knowledge in wildlife science may benefit both scientists and knowledge holders by increasing communication, engagement, and trust. It may also benefit scientific rigor and application by increasing contextual understanding of data and results and increasing or improving data with which to conduct analyses. By meaningfully including experiential wildlife knowledge, there is the potential to develop methods of more inclusive science that benefits scientists, knowledge holders, and wildlife.

Declaration of competing interest

We declare no conflicts of interest.

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CONNECTING STATEMENT

This thesis employs a knowledge co-production approach to assess moose habitat use in the AFR. In Chapter 1, I explored methods of quantitatively interweaving local, expert, and Indigenous knowledge into other wildlife analyses. In Chapter 2, I aim to use knowledge co-production methods to answer a local knowledge priority about moose habitat use in an Adapted Forestry Regime (AFR) in Eeyou Istchee, Québec. The toolbox of methodologies identified in Chapter 1 were used to determine an appropriate and achievable methodology for our study of moose in Eeyou Istchee. Using Cree knowledge in the model development stage combined with systematic data such as GPS moose locations and GIS land covariate layers in the modeling and analysis stage, we were able to interweave qualitative Cree knowledge into an analysis using knowledge co-production methods that provided quantitative answers to pressing questions about the effectiveness of the AFR.

CHAPTER 2: MOOSE HABITAT SELECTION IN AN ADAPTED FORESTRY REGIME IN EEYOU ISTCHEE, NORTHERN QUÉBEC: TELEMETRY, MODELING, AND KNOWLEDGE CO-PRODUCTION

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Abstract

Forestry activities in boreal forests may impact wildlife species or ecosystems that Indigenous communities rely on for food and culture. Forestry impacts wildlife through the removal of food or shelter plants, changing the successional stage of forests, and increasing road density on the landscape. In Eeyou Istchee, the Cree traditional territory in Northern Québec, forestry is a major disturbance and, in 2002, the Adapted Forestry Regime (AFR) was implemented with the intention of improving the complementarity of forestry practice with the traditional Cree way of life and increasing Cree participation in forest management. Included within the AFR was the selection of "Special Sites of Wildlife Interest to the Cree" (hereafter '25% areas') that would be managed in ways to promote and protect wildlife habitat, including habitats used by moose. Moose habitat use in the AFR area, in particular of the 25% areas, regenerating stands, and thinned and brushed cut forests, has not been assessed since the implementation of the AFR. Using a knowledge co-production approach guided by a steering committee that included Cree community representatives, we analysed the winter and summer habitat selection of 38 GPS collared female moose in relation to land cover, 25% areas, elevation, road density, and distance to water using Minimum Convex Polygon (MCP) home range, Generalized Linear Model (GLM), and Resource Selection Functions (RSF) using Manly selection ratios. Moose had much larger home ranges in summer than winter and selected for 25% areas and against 75% areas in both seasons. Habitats that were used across both seasons and scales included forests over 7 m tall that were mixedwood-coniferous, mixedwood-deciduous, deciduous, and coniferous with fir. In summer, moose additionally used alder, flood zones, and thinned and brush cut forests, which were avoided in winter. Moose selected for regenerating post-forestry stands in winter at the home range scale and summer at the study area scale, but did not select them as strongly as preferred undisturbed forests. Moose avoided forests regenerating after natural disturbances at all

seasons and scales except summer at the home range scale. Through these analyses, we provide the first assessment of moose use of the 25% areas and quantify use of logged stands and thinned and brush cut forests. Consistent with previous literature and Cree Knowledge, we identify the moose habitat importance of mixedwood and deciduous stands, as well as coniferous stands that contain fir, and based on key differences identified between summer and winter habitat, the importance of a multi-season approach to assess moose habitat selection.

Introduction

Forestry can have significant direct and indirect impacts on ecosystems, wildlife, and Indigenous communities that live in and rely on boreal forests. Forestry impacts are widespread across the global extent of boreal forests (Gauthier et al., 2015) and are often related to the successional changes to forest ecosystems caused by the removal of tall, old trees and the early-succession regrowth of natural or replanted trees and shrubs that colonize rapidly after disturbance (Kuuluvainen and Gauthier, 2018). Boreal forests are generally well-adapted to frequent disturbance and continuous successional change, with intense fire, weather, and disease regimes being a normal part of the boreal ecosystem cycle (Krawchuk et al., 2006; Machado Nunes Romeiro et al., 2022; Sánchez-Pinillos et al., 2019).

Forestry adds to or replaces these natural disturbance regimes, and has come to represent the primary driver of successional change in many parts of the boreal forest (Kuuluvainen and Gauthier, 2018). These successional changes can impact different wildlife species differently, with species adapted to mature forests, such as birds and caribou, losing habitat quality after disturbances, and species such as deer, moose, and beavers capitalizing on the increase in forage available in early successional stands following disturbance (Telfer, 1974). Species that are positively impacted by forestry may be well-adapted to open habitats (Meijaard and Sheil, 2008) or may benefit from the early succession plant growth that increases browse and productivity (Collins and Schwartz, 1998; Telfer, 1974). Species that are negatively impacted by forestry may be impacted by habitat loss (St-Laurent et al., 2009), reduced habitat connectivity (Bergsten et al., 2013; Mikoláš et al., 2017), altered predation impacts (Gardner et al., 2019; Pillay et al., 2019), or loss of old-growth food species (Mcshea et al., 2007; Morgan et al., 2018). Forestry strategies to reduce negative impacts on species and people reliant on them include mosaic harvesting, retention harvesting (Gustafsson et al., 2012; Kuuluvainen et al., 2019), rotation schedules (Egnell and Björheden, 2013; Roberge et al., 2016), and selective harvesting (Asner et al., 2004; Piponiot et al., 2016). However, the success of these measures may be negated by other impacts of forestry not accounted for such as the increase in road networks that can increase wildlife mortality through collisions (Nielsen et al., 2008), changing predator movement and access (Houle et al., 2010), increased predation (Bastille-Rousseau et al, 2011) and increased hunting (Rempel et al., 1997; Courtois & Beaumont, 1999).

Changes to wildlife habitat, abundance, and behaviour resulting from forestry may also impact human communities dependent on wildlife for subsistence food. In particular, Indigenous communities situated in boreal forests and reliant on boreal wildlife can be impacted by forestry by the changes to ecosystems and the services they provide (Fuentes et al., 2020; Stevenson and Webb, 2003). The wellbeing of Indigenous people and communities living in boreal forests is integrally linked to connection to the land and the ecosystem services the land provides (Bélisle et al., 2021; Bélisle and Asselin, 2021). In particular, land value for hunting, fishing, and trapping can be derived from the abundance, quality, access, and experience (Bélisle et al., 2021). Changes to landscapes, especially hunting grounds, due to natural resource extraction is a major concern for many of Canada's First Nations peoples (Bélisle and Asselin, 2021). These changes to landscapes can impact habitat quality for wildlife, which may impact the abundance, distribution, and behaviour of frequently harvested species. Reducing these impacts should include Indigenous involvement in forestry and wildlife management, which can have significant local economical, sociocultural, and environmental benefits (Lawler and Bullock, 2017).

Methods to further improve outcomes for wildlife include knowledge co-production approaches, in which local communities are involved in wildlife and forestry management. Knowledge co-production can be defined as: "*Iterative and collaborative processes involving diverse types of expertise, knowledge and actors to produce context-specific knowledge and pathways towards a sustainable future*." (Norström et al., 2020). It can also be defined as: "*involv[ing] stakeholders from diverse knowledge systems working iteratively toward common vision and action*" (Nel et al., 2016). Four key principles of knowledge co-production are that it is 1) context based, 2) pluralistic, 3) goal-oriented, and 4) interactive (Norström et al., 2020). The principle of plurality, in which co-produced knowledge must recognize multiple ways of knowing (Norström et al., 2020) by bringing together multiple knowledge sources and types to address a defined problem (Nel et al., 2016), is especially important because when people are closely involved in the production of knowledge, trust in and likelihood to act on results is increased (Cash et al., 2003). For scientific information to be effective, it must be considered credible, salient, and legitimate by stakeholders (Cash et al., 2003). By using knowledge co-production methods, studies can treat Indigenous knowledge more respectfully, increase trust and invite participation (Latulippe and Klenk, 2020). Knowledge co-production has been used in Canada's north to a high degree of success (Armitage et al., 2011; Dale and Armitage, 2011; Johnson et al., 2020).

Eeyou Istchee is the traditional territory of Cree Indigenous peoples, located on the eastern side of James Bay, in northern Québec, Canada. Forestry activity in Eeyou Istchee has and is impacting moose, a hunted wildlife species that is critical for subsistence food stability and cultural activities. Concern about the impact of forestry practices on moose and other wildlife populations led to the development of the Adapted Forestry Regime (AFR) in 2002 (Le Gouvernement Du Québec, 2002). The objectives of the AFR were to conduct forestry in a manner that allows: a) adaptations to better take into account the Cree traditional way of life, b) greater integration of concerns relating to sustainable development, and c) participation, in the form of consultation, by the James Bay Crees in the various forest activities operations planning and management processes (Le Gouvernement Du Québec, 2002).

In this agreement, stipulations were made that involved identifying 25% of land area in each "trapline" (family hunting territories managed by a tallyman, or experienced hunters that manage and/or monitor hunting and activities on traplines) to be specially managed as Sites of Special Wildlife Interest to the Cree (25% areas) to preserve or generate high quality wildlife habitat. Areas were chosen based on importance to wildlife collectively, but moose habitat was a strong driver of 25% area selection (Jacqmain et al., 2012). The land within these 25% areas had special management stipulations that aimed to better preserve pre-existing high-quality habitat or conduct logging in ways that reduced negative impacts. These measures include mandating that only mosaic cutting (a harvesting method in which the area of logged stands is balanced with comparable areas of retained stands, and cuts are made to increase heterogeneity across landscapes) would be used in these areas, preserving at least 50% of stands over 7 m tall, selection and use of interconnected residual forest blocks, and slower rotation times with residual forest being allowed to regenerate to a minimum of 7 m tall. This differs from areas outside the 25% areas (75% areas), in which mosaic harvesting is done at a 75% target level, 30% of stands over 7 m tall are preserved, and rotation times are quicker with harvesting being allowed for forests between 4 and 7 m tall. Twenty years since the AFR was implemented, moose habitat use inside the AFR can now be assessed.

The current study is part of a multi-stage, collaborative, knowledge co-production research project advised by a steering committee composed of representatives from Cree communities, Cree regional government and trapper associations, wildlife and forest scientists from the provincial and a forestry co-management organization. The overall project focuses on Cree knowledge and moose GPS collar locations to assess the evolution of moose habitat use and quality in the AFR. Here we focus on moose GPS collar locations from within AFR to assess how moose habitat selection is related to forest type and stand height, elevation, distance to water, road density, and 25% areas. In particular, we explore how moose are using 25% areas, regenerating stands, and thinned and brush-cut forests. We analyzed habitat selection at a second and third order scale of analysis using a Net Squared Displacement (NSD) movement analyses, Minimum Convex Polygon (MCP) home range analyses, Generalized Linear Models (GLM), and Resource Selection Functions using Manly selection ratios. While GPS collar locations and GIS ecosystem data were used in the models, the time periods and the variables we included in our habitat selection analysis are informed by Cree knowledge of moose seasonality and habitat requirements.

Methods

Study area

We conducted the study in Eeyou Istchee James Bay, the Cree traditional territory of the James Bay region of Québec, Canada, focused specifically in the area of the AFR (Figure 1). Eeyou Istchee James Bay is predominantly a black spruce feathermoss ecosystem, with patches of mixedwood and deciduous forest, and is extensively forested in the southern portion of the territory (Jacqmain et al., 2012). Prior to widescale commercial logging, the region was made up of 70% mature forest stands in large, connected areas (Jacqmain et al., 2012), with forest fires representing an important agent of natural disturbance with a mean fire interval of about 85 years (Bergeron et al., 2002). Timber logging, which expanded rapidly at a large, commercial scale in the 1960s, has left a legacy of major clearcutting across a large southern, inland portion of Eeyou Istchee James Bay (Jacqmain et al., 2012) and logging continues throughout the southern and central portion of Eeyou Istchee James Bay in the AFR. The AFR encompasses 68,812 km^2 of area in total and is made up of three contiguous regions throughout the Eeyou Istchee James Bay area (Figure 1). The importance of moose habitat in the AFR has led to long term moose monitoring programs that include deployment of GPS collars on female moose. For this study, we selected an area of interest that encompassed all available moose GPS points from 2018 - 2021, with a 10 km buffer (Figure 1). This study area was just under 40,000 km^2 in size.

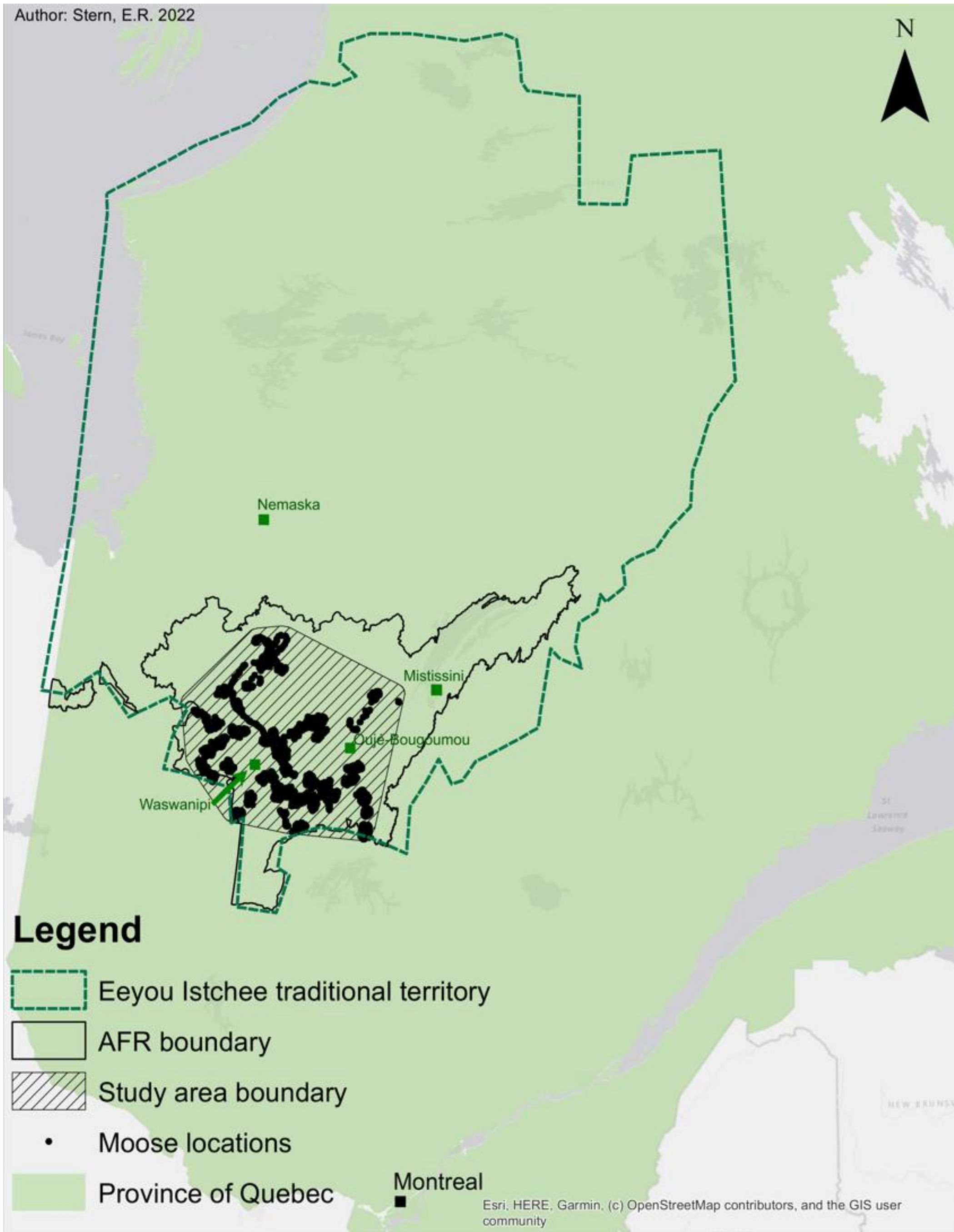

Fig 1. Map of Eeyou Istchee, the Cree traditional territory in Québec, Canada, including the four communities where Cree knowledge interviews about moose habitat quality were conducted and the moose GPS collar locations used in the habitat selection analysis. Also indicated is the administrative boundary of the Adapted Forestry Regime and the study area boundary used in the habitat selection analysis.

Knowledge Co-production

We involved local knowledge in the consultation/study design, pre-modeling/analytical approach, and follow-up/member checking stages (Stern and Humphries, 2022) as part of a knowledge co-production process. This study is one component of a larger, comprehensive project on moose habitat quality in Eeyou Istchee led by researchers at McGill University. The project is advised by a multi-stakeholder steering committee composed of Indigenous, academic, and government representatives. Project goals, objectives, deliverables, and methodologies are co-developed with this steering committee. In the consultation/study design phase, scoping interviews were conducted to situate the analysis in local context, identify concerns and priorities, and identify appropriate sources of knowledge.

The methodologies included in the broad moose habitat quality project were iteratively developed with feedback from the steering committee, and regular meetings occurred over the course of the study to discuss and develop the project. This quantitative study will act as one component of a future mixed-methods analysis that interweaves Cree expert knowledge and quantitative analyses. In the pre-modeling/analytical approach stage, we conducted in-person semi-structured interviews with individuals and family groups who held extensive knowledge of moose behaviour and populations to retrieve information to guide model variable selection. During these interviews, moose habitat quality was discussed with tallymen and landusers for each trapline. Common topics addressed in these interviews were used to develop a list of recurring, modellable variables to be explored in the quantitative analyses. As part of the follow-up/member checking stage, workshops with interview participants were organized approximately 8 months after the pre-modeling/analytical approach stage interviews to validate the researcher's interpretations of results.

GPS Collar and Land Covariate Data

GPS collars were deployed between mid-January and mid-February of three consecutive years (2018, 2019, and 2020) totalling 38 adult female moose. Immobilization procedures are described in detail by Lamglait et al., (2021), but briefly, animals were located from a helicopter and were pursued until they could be darted using aluminum 3-4 ml projectile darts (Slo-Inject RDD Device type U, 1.5-inch 14-ga needle with a gelatin collar, Pneu-Dart, Williamsport, PA

17701, USA) and a CO2-powered dart rifle (CO2 injection rifle model J.M.SP.25, Dan-Inject ApS, 6000 Kolding, Denmark). Fixed doses (3–4 ml, depending on dart supply and estimated weights) of a premix BAM combination (butorphanol tartrate 27.3 mg/ml, azaperone tartrate 9.1 mg/ml, and medetomidine hydrochlorate 10.9 mg/ml, BAM-II, Chiron Compounding Pharmacy Inc, Guelph, ON N1H 6T9, Canada) were used. After drug administration, the helicopter pulled back and increased its altitude to approximately 200 m above the ground to limit disturbance while maintaining observability of the animal during the induction phase. Once recumbent, the helicopter landed within approximately 200 m, and the moose were approached quietly, placed in sternal recumbency, and blindfolded. Supplemental oxygen was provided and heart rate and body temperature were monitored during handling. Moose were fitted with Vertex Lite GPS collars (Vectronic Aerospace GmbH, Carl-Scheele-Str.12, 12489 Berlin, Germany) equipped with an Iridium or Globalstar satellite communication, as well as a VHF beacon, mortality sensor, temperature sensor, 3-axis activity sensor, and a timer-controlled drop-off mechanism. Collars were programmed to record location at 2-hr intervals throughout the year and to drop-off between 18 and 22 months after deployment, however the actual amount of time collars were on moose varied between 3 and 30 months

Collar deployments were initiated and conducted by biologists working with the Ministère des Forets, de la Faune et des Parcs du Québec (MFFP), and accordingly procedures and immobilization of moose were approved by institutional animal care committees of the MFFP. Researchers from McGill University accessed the collar locations via a data sharing agreement with MFFP. This study prioritized the collaring of only mature female given the broad implication of females in fitness components of large herbivore populations (Clark and Tait, 1982; Gaillard et al. 2020). For moose in particular, populations may be productive with biased sex-ratios and sex-ratio may not be a strong determinant of productivity compared to factors such as female body size and health (Solberg et al., 2002).

GIS-based land covariate data were obtained from MFFP, the Cree Nation Government (CNG), AQrseau+, and NASA (Farr et al., 2007). MFFP sourced forest inventory data included information on disturbance history, coarse forest type (i.e. mixedwood, coniferous, deciduous, unclassified regeneration), dominant species, and stand height. Forest inventory data are stored for each stand in polygon shapefile format, with median stand size being 0.04 km^2 (4.1 hectares). Road network data were retrieved from AQréseau+ data from Données Québec, and permanent water data were collected from MFFP. Boundaries of Cree-designated 25% areas were provided by the CNG. Elevation data were derived from NASA STRM 30m Digital Elevation, sourced from Google Earth Engine (Farr et al., 2007).

<u>*Analysis*</u>

Data: Handling

We identified mid-winter and mid-summer as highly contrasting seasons in which we would expect significantly different selection due to differing forage availability, climate and snow depth, and different hunting and predation pressures. Based on the local climate and discussions of seasonality in interviews, we defined our winter window of analysis to be January 1 - February 28 and our summer window of analysis to be July 1 - August 31). We divided GPS collar locations into summer and winter datasets based on these dates, using data from all years available for each moose. These time windows generally align with the Cree seasons of *Niipin*, or 'Time for gatherings' (July - August), and *Pipun*, or 'Best time to trap' (January - February). Cree knowledge holders also identified to us the importance and distinctiveness of spring calving and autumn habitat selection, which we do not consider here but will be the subject of subsequent analyses and articles.

Variable selection

We related variables identified as important to moose habitat in interviews, to land covariate variables that could be quantitatively modeled in our analysis. We identified land cover including stand height of forests, elevation, distance to water, road density, and 25% areas as explanatory variables for the GLM analysis (Table A1). *Land cover* was derived from forest inventory data and contained specific information on forest or ecosystem type as well as stand height. "Productive forest", or harvestable and accessible forest stands were classified as mixedwood (divided into deciduous dominated, coniferous dominated, and unknown dominated), deciduous, or coniferous (divided into fir present, fir absent, and fir unknown). Stand height was derived from the same forest inventory data and was grouped into the following categories: non-forest (alder, water, dryland, roads, flood zones, etc.), under 4 m, 4 - 7 m, under 7 m, and 7 + m; these height categories were chosen to be able to explore the effectiveness of particular stand height regulations applied to the 25% areas (within 25% areas reharvest is delayed until regenerating stands exceed a > 7m threshold, whereas outside of the 25% areas reharvest can proceed as soon as stands exceed a > 4 m height threshold). Within the land cover category, we defined: regenerating post-forestry stands as sites that were recorded as having a forestry-related disturbance occur on them and had not regenerated to sufficient canopy heights to assess vegetation composition; regenerating post-natural stands as sites that had experienced natural disturbance such as fire, wind, or disease and had not regenerated to sufficient canopy heights to assess vegetation composition; and thinned forest stands as sites that had experienced pre-commercial thinning or brush-cutting between 2013 – 2021. *Distance to water* was calculated at 25 m cell resolution as a continuous Euclidean distance variable and binned into the following categories 0m, 0-25 m, 25 - 50 m, 50 - 100 m, 100 - 250 m, 250 - 500 m, 500 - 1000 m, and 1000 + m from water features, which included permanent streams, rivers, ponds, and lakes. Distance to standing water, distance to permanent to still water, and distance to all water were explored separately, but only distance to all water was retained in the analysis as it was the most supported variable and these three variables were highly collinear. *Elevation* was expressed as lowland, midland, and upland categories determined by creating a hill map in raster form using a DEM to identify hills relative to the surrounding terrain. Cell values were divided into quartiles and classified into lowland, midland, or upland (Table A.1). *Road density* was calculated by summarizing length of drivable roads (including paved address network roads and graveled forestry roads) per square kilometer, and using quantiles to classify this continuous variable into low, medium, and high density categories (Table A.1.). Covariate layer preparation was performed in ArcGIS version 10.7.1 (ESRI, 2011). Because logging continued to occur in the area throughout the study period, we developed updated land cover layers for each annual timestep in the model from 2018-2021, based on available data of the location and timing of cutblocks.

AFR habitat composition

We developed complex land type maps based on forest inventory data for the years 2018 to 2021 within the study area boundary, and used summary statistics to assess the status and change in habitat type proportions over the study period. Calculations were made for all habitats within the study area boundary (Figure Table A1). Because of the ongoing logging in the study area throughout the study period, maps were updated for each administrative year to include reported forestry and natural disturbances. Disturbance data were available for the 2017-2018 through 2020-2021 administrative years. A total of four timestep layers were made for land type and height class accordingly. Differences in composition of land cover, road density, and elevation were compared between the 25% areas and the 75% areas.

Net Squared Displacement analysis

Net Squared Displacement (NSD) (Bunnefeld et al., 2011) was performed on the entire set of available moose GPS points to determine whether migration patterns were present in the population sample, and identify abnormal relocations to be removed from the analysis data. We calculated and graphed NSD over time and visually interpreted the figures to identify major relocations. NSD analysis was done using the package “adehabitatLT” (Callenge, 2006) in R version 4.1.1 (R Core Team, 2020). NSD analysis indicated that the majority of moose in the population sample (36 of 38) did not display major relocations or seasonal migration patterns. Most movement occurred in summer months, and very little movement tended to occur in winter months. A limited number of moose (2 of 38) underwent major relocations in which long distance movement was documented. The longest relocation occurred over approximately 125 km from the starting position. We removed relocation events from the data before further narrowing the dataset to winter (January 1 - February 28) and summer (July 1 - August 30) locations for home range, GLM, and RSFs.

Home range analysis

We performed a home range analysis to help guide the scale of the habitat selection analysis and provide home range polygons within which to generate random points for RSF Manly selection ratios and GLMs. After removal of relocation events based on NSD analysis and separation into summer and winter datasets, home range polygons were created for summer and winter in each year for each moose. These were created using the Minimum Convex Polygon (MCP) (Mohr, 1947) method set at the 95% threshold (Figure 3). Analysis was done using the “adehabitatHR” package (Callenge, 2006) in R version 4.1.1 (R Core Team, 2020). Correlation between sample size and home range size was assessed, and based on this analysis moose where seasonal home range sample size was under 100 were removed from habitat selection, GLM, and RSF analysis.

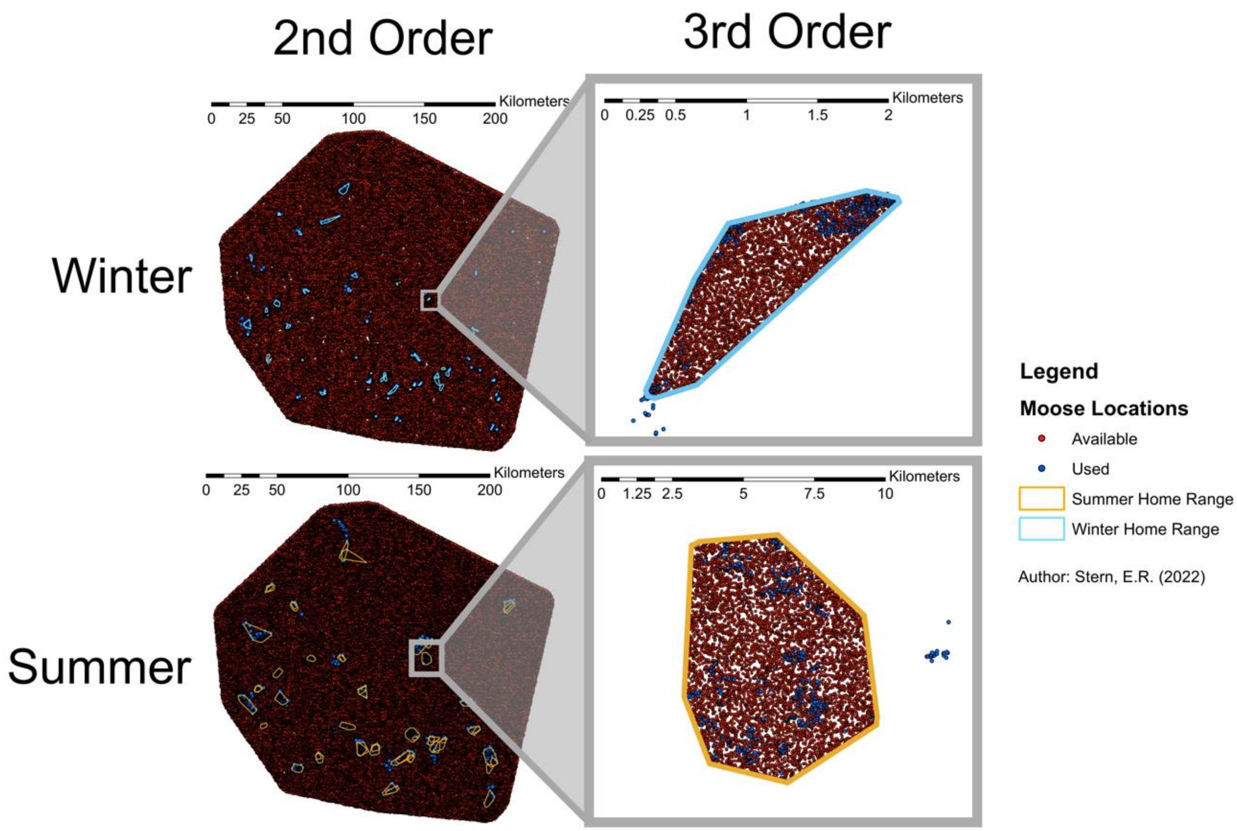

Fig 2. Locations used (blue) and available (red) for 28 collared female moose in the Adapted Forestry Regime within Eeyou Istchee in two seasons (summer and winter) and at two scales of analysis (second order and third order). The second order analysis (study area scale) compares used locations within home ranges to available locations from across the study area, whereas the third order analysis (home range scale) compares used locations to available locations within home ranges, which are shown as 95% minimum convex polygons outlined in yellow for summer and blue for winter. The third order analysis compares used locations within home ranges to available locations within home ranges, which are shown as 95% minimum convex polygons outlined in yellow for summer and blue for winter.

GLMs

GLM analyses were conducted for second order habitat selection (study area scale) and third order habitat selection (home range scale) (Johnson, 1980). GLM modeling was conducted for moose location data generated to assess habitat selection at the second and third order. A GLM approach was selected because of the unbalanced sample size of GPS points for each moose, area, and season because of the varying periods of time collars were on each moose. Second order analyses, used to assess where moose position their seasonal home ranges across the broad study area, were conducted on a dataset composed of "used" summer and winter locations within seasonal home ranges and "available" points randomly generated across the entire study area (Figure 2). Third order analyses, which assess habitat selection within seasonal home ranges, was conducted on a dataset composed of "used" summer and winter locations within seasonal home ranges and "available" locations randomly generated within the same seasonal home ranges as the "used" locations (Figure 2). For both datasets, 10 random points were generated for each "used" location using the package "amt" (Signer et al., 2019) in R.

GLM models were performed separately for winter 3rd order selection data, summer 3rd order selection data, winter 2nd order selection data, and summer 2nd order selection data. Models were built iteratively by exploring single variable models, using Akaike information criterion (AIC) to assess variable strength and importance, and gradually increasing model complexity in order of most supported to least supported variables. The variables land cover, distance to water, road density, elevation, and 25% area status were explored. Collinearity of variables was assessed using Generalized Variance Inflation Factor ratios in the package "car" (Fox and Weisberg, 2019), with all models showing all variables with gvif < 1.1. In total, 10 models were assessed for each order of analysis and season (40 total models). McFaddens R-squared value was calculated for each model to further explore model fit and strength. We selected the best models from which to assess predictive accuracy and develop habitat maps based on AIC and R-squared value. GLM analyses were performed in R using the package "glm2" (Marschner, 2011, p. 2). Models were assessed using AIC and Nagelkerke's pseudo R-squared values, which are suitable for logistic regression and were calculated using the package "pscl" (Jackman, 2020).

Resource Selection Functions

Resourse Selection Functions were conducted for second order habitat selection (study area scale) and third order habitat selection (home range scale) (Johnson, 1980) using Manly selection ratios. Manly selection ratios were calculated for land cover, distance to water, elevation, road density, and 25% area status for second and third order selection in summer and winter. Habitat types where the number of available points was under 10 were removed from summary figures because of low estimate accuracy. Both study area and home range scale analyses used a Design III structure, in which availability of habitat types differed for individuals, producing individual selection estimates for each habitat category for each individual using 95% confidence intervals. Based on this confidence interval, Manly selection ratio under 0.95 were considered avoidance behaviour, ratios between 0.95 and 1.05 were considered use proportional to availability, and ratios over 1.05 were considered selection. Manly selection analyses were performed using the package “adehabitatLT” (Callenge, 2006) in R. Individual variation in selection was assessed in the Resource Selection Functions using Coefficient of Variation (CV) for each categories mean selection estimate.

Results

AFR habitat composition

The study area habitat composition is primarily forest, with all productive forest area totaling 70.1% of all land at the start of the study, in the 2017-2018 administrative year. The other 29.9% of land was made up of alder (1.64%), drylands (0.69%), flood zones (0.64%), roads (0.03%), small islands (0.03%), water (11.58%), wetlands (14.28%), and other habitats (0.12%). Within the 70.1% of land counted as forest, the most common forest type was mature (7+ m canopy height) coniferous forest without fir, totaling 50.34% of forest area. The next most common type of forest was regeneration post-forestry disturbance (Table A.1), which made up 13.67% of forest. This doubled the amount of forest that was regenerating post-natural disturbance (Table A.1), which made up 6.45% of forest. Mid-height (4 to 7 m) coniferous forests with fir absent (9.4%), mature mixed forest with coniferous dominant species (4.25%), mature mixed forest with deciduous dominant species (3.04%), and mature coniferous forest with fir present (2.65%) were also relatively common forest types. All other forest types made up under 2.5% of forest area each. The majority of forest was 7+ m in height (63.8%), followed by 0 to 4 m in height

(23.2%), and the least common height class was 4 to 7 m forest (13.0%). Over the course of the study period, proportion of undisturbed forest declined, and proportion of forest regenerating post-forestry increased from 13.54% to 16.84% of forest area. Most of these new regenerating stands were originally mature coniferous forest with no fir, which declined from 51.60% to 48.41% of forest area.

The 25% areas displayed some key similarities and differences from 75% areas in terms of land cover, height class, elevation, and road density. Within the land cover variable, 25% areas and 75% areas shared the same most common five land covers with differences in proportions being generally less than 0.05. These five most common land cover categories were: 1) coniferous forest without fir over 7 m, 2) wetland/bog, 3) regenerating post-forestry, 4) coniferous forest without fir between 4 and 7 m, and 5) regenerating post-natural disturbance. Within the land cover variable, the primary differences were in less abundant categories, wherein 25% areas had greater proportions of tall deciduous, mixedwood, and coniferous forest with fir. Elevation within 25% areas differed with 25% areas having more upland, less lowland, and equal amounts of midland to 75% areas. Road density also differed, with 25% areas having much lower proportion of land with low road density, and greater area with medium and high road density than the 75% areas.

Home range analysis

Strong differences occurred between the sizes of summer and winter home ranges across the population sample of female moose (Figure 3). Summer home ranges tended to be very large and have wide variation in size, with median area of 29.40 km^2, mean area of km^2, minimum area of 6.70 km^2, and maximum area of 133.33 km^2. Winter home ranges were much smaller and more consistent, with median area of 0.97 km^2, mean area of 3.00 km^2, minimum area of 0.007 km^2, and a maximum area of 25.12 km^2.

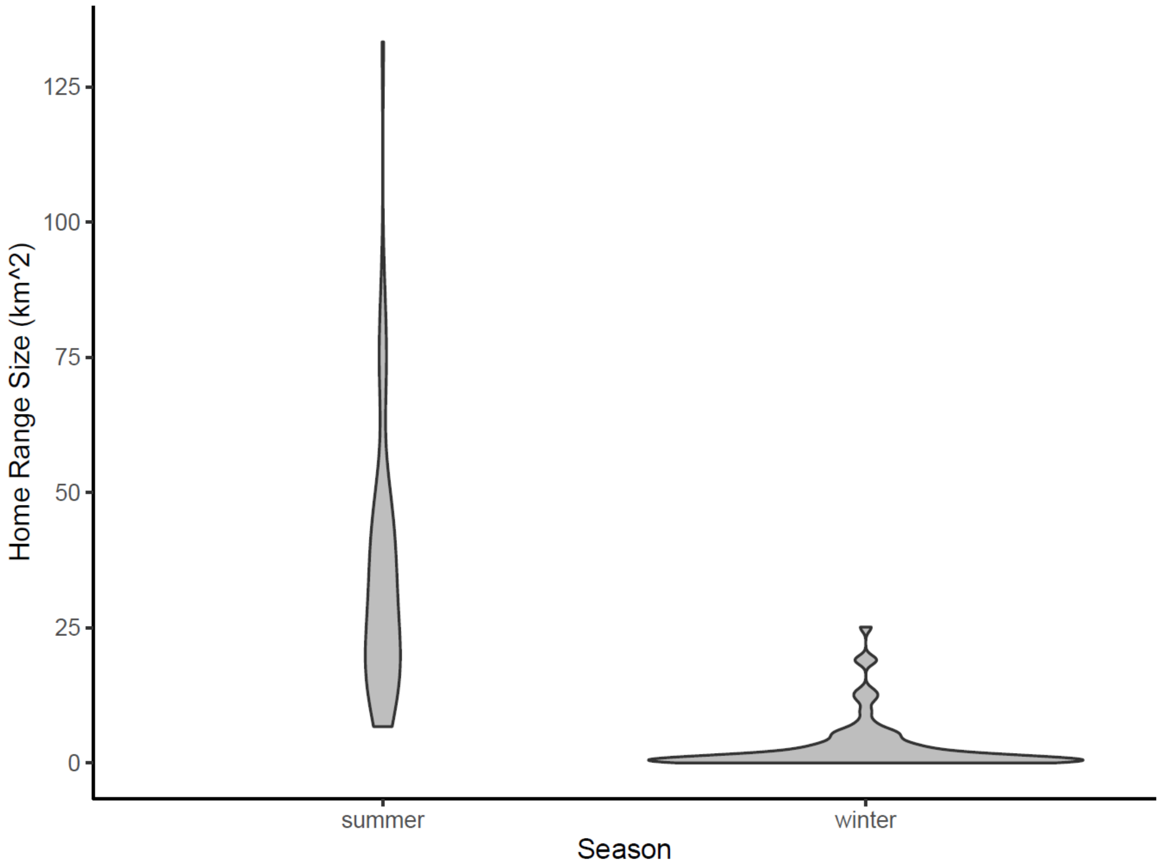

Fig 3. Home range size distribution of 38 female moose in Eeyou Istchee, Québec, with unique values calculated for each individual in each mid-summer (July 1 - August 31) and mid-winter (January 1 - February 28) of each available study year from 2018 - 2021.

GLM analysis

In both seasons and scales of analysis, the most supported models included all of five explored variables, and in all cases the null model was least supported (Table 1., Table A2., Table A3.). Land cover was consistently the most important variable for both seasons and scales of analysis, whereas the relative importance of elevation, distance to water, road density, and 25% areas varied among seasons and scales. In winter at both scales of analysis, land cover, elevation, and distance to water tended to be the most important variables, with road density and 25% area status having less importance. In summer at both scales of analysis, land cover, distance to water, and 25% area status had higher importance and road density and elevation had lesser importance. Most support was generally higher in study area scale analyses than home range analyses, however pseudo-R squared cannot be directly compared across datasets and the precise relative performance of the best supported models is challenging to assess.

Table 1. Best supported GLM models for each season and order of selection based on AIC, pseudo-R-squared (Nagelkerke, 1991) for all seasons and orders of analyses.

Season	Scale	Best Supported Model	Pseudo R^2
Winter	study area	presence ~ land cover + elevation + distance to water + 25% areas + road density	0.214
Summer	study area	presence ~ land cover + 25% areas + distance to water + elevation + road density	0.098
Summer	home range	presence ~ land cover + distance to water + 25% areas + elevation + road density	0.054
Winter	home range	presence ~ land cover + distance to water + elevation + 25% areas + road density	0.044

RSF analysis

In winter at the study area scale, moose situated home ranges in habitats offering more land cover that was mixedwood-deciduous, mixedwood-coniferous, deciduous, and coniferous forest with fir over 7 m tall, as well as mixedwood-coniferous forest under 7 m. They also situated home ranges in areas with more 25% areas and sites of upland and midland elevation, that were between 250 – 1000 m from water, as compared to the whole study area (Table 2). Moose somewhat avoided land covers including coniferous forest without fir over 7 m, regenerating post-forestry stands, alder, coniferous forest under 4 m, thinned and brush cut forest, and regenerating post-natural disturbance stands. Other habitats that were somewhat avoided included 75% areas, lowland areas, areas with medium or high road density, and areas close (0 – 250 m) from water. Strongly avoided land covers included deciduous forest under 7 m, flood zones, coniferous forest with or without fir between 4 – 7 m, drylands, wetlands/bogs, powerlines, open water, islands, or other areas (Table 2). There was generally very high individual variation in winter selection at the study area scale, with thinned and brush cut forest, deciduous forest under 7 m, flood zones, coniferous forest with fir between 4 – 7 m, dryland, and water, islands, and other having high individual variation in selection (Table 2).

In winter at the home range scale, analysis indicated that moose used locations within their home ranges that contained more land cover that mixedwood-deciduous, mixedwood-coniferous, deciduous, and coniferous forest without fir over 7 m, mixedwood-coniferous forest under 7 m, and regenerating post-forestry stands (Table 2). Within winter home ranges, moose also selected for 25% areas, midland elevation, low and medium road densities, and areas 100 – 1000 m from water (Table 2). Within the land cover variable, moose were indifferent to alder, coniferous forest under 4 m, and mixedwood-deciduous forest under 7 m in home ranges in winter. Moose were also indifferent to 75% areas and sites that were 1000 + m and between 50 – 100 m from water (Table 2). Moose somewhat avoided land covers including coniferous forest without fir over 7 m, regenerating post-natural disturbance stands, flood zones, and coniferous forest with or without fir between 4 – 7 m tall, as well as areas that were lowland elevation, of high road density, and between 0 – 50 m from water (Table 2). In winter home ranges, moose strongly avoided land covers including thinned and brush cut forests, deciduous forest under 7 m, dryland, wetland/bogs, powerlines, open water, island, and other habitats (Table 2). The strongest individual variation in selection in winter at the home range scale was found in flood zones, water, islands, and other, regenerating post-forestry stands, alder, thinned and brush cut forests, coniferous forest without fir between 4 – 7 m, drylands, and wetlands/bogs (Table 2).

In summer at the study area scale, moose situated home ranges in habitats offering more land cover that was deciduous, mixedwood-deciduous, mixedwood-coniferous, and coniferous forest with fir over 7 m, mixedwood-coniferous and mixedwood-deciduous forest under 7 m, coniferous forest with fir between 4 and 7 m, thinned and brush cut forests, regenerating post-forestry stands, alder, and flood zones, as well as sites that were 25% areas, areas of upland and midland elevation, and areas that were very close (0 – 25 m) or mid-distance (100 – 500 m) from water (Table 2). The most preferred habitat in summer at the study area scale was flood zones (Table 2). In summer at the study area scale, moose were indifferent to areas with low or medium road density (Table 2). Moose somewhat avoided land covers including coniferous forest without fir over 7 m, coniferous forest under 4 m, regenerating post-natural disturbance sites, deciduous forest under 7 m, dryland, and powerlines, and strongly avoided wetland/bogs, mixedwood-deciduous forest under 7 m, water, islands, and other habitats (Table 2). The strongest individual variation in selection in summer at the study area scale occurred in land covers mixedwood-deciduous forest under 7 m and powerlines (Table 2).

In summer at the home range scale moose used locations that had more land cover that was flood zone, mixedwood-deciduous, mixedwood-coniferous, deciduous, and coniferous forest with fir over 7 m, mixedwood-deciduous, and mixedwood-coniferous forest under 7 m, alder, thinned and brush cut forest, regenerating post-natural disturbance stands, and coniferous forest with and without fir between 4 – 7 m, as well as 25% areas, upland and lowland areas, areas with low road density, and areas either very close (0 – 25 m) or mid-distance (250 – 1000 m) from water (Table 2). In summer within home ranges, moose somewhat avoided land covers including regenerating post-forestry stands, coniferous forest under 4 m, deciduous forest under 7 m, dryland, wetland/bogs, powerlines, water, islands, and other habitats, as well as 75% areas, lowland areas, areas with medium and high road density, and areas between 25 – 100 m from water and over 1000 m from water (Table 2). The highest individual variation in selection occurred in land cover categories deciduous forest under 7 m, water, islands, and other, powerlines, and coniferous forest with and without fir between 4 – 7 m (Table 2).

Table 2. Manly selection ratios in summer and winter at the second order (study area scale) and third order (within home range scale) of selection for all variables by 38 female moose in Eeyou Istchee, Québec, with strong avoidance shown in red, minor avoidance in orange, selection proportional to availability in yellow, minor selection in light green, and strong selection in dark green. Selection estimates were calculated at the individual level and summarized as mean, with associated Coefficient of Variation (CV) shown for each category.

		Winter (January - February)				*Summer (July - August)*			
Variable	**Category**	**Second Order (Study Area)**	**CV**	**Third Order (Home Range)**	**CV**	**Second Order (Study Area)**	**CV**	**Third Order (Home Range)**	**CV**
Land Cover	Mixedwood-deciduous > 7 m	3.538	0.760	2.001	0.712	1.076	1.373	1.240	1.029
	Mixedwood-coniferous > 7 m	2.463	0.950	1.624	0.598	1.442	1.163	1.189	0.542
	Deciduous > 7 m	2.288	1.080	2.305	0.693	1.952	1.010	1.842	0.821
	Coniferous w/ fir > 7 m	1.487	1.473	1.151	0.717	1.711	1.034	1.349	0.955
	Mixedwood-coniferous < 7 m	1.267	1.407	1.270	0.499	1.716	0.795	1.376	0.561
	Coniferous w/o fir > 7 m	0.798	0.712	0.846	0.454	0.860	0.473	0.958	0.368
	Regenerating post-forestry < 4 m	0.781	1.329	1.209	1.684	1.194	0.582	0.903	0.445
	Alder	0.722	2.172	0.967	1.221	1.673	0.639	1.573	0.724
	Coniferous <4 m	0.708	2.143	0.954	0.898	0.941	1.603	0.918	0.953
	Mixedwood-deciduous < 7 m	0.666	2.672	0.972	0.766	0.316	3.351	1.735	0.365
	Thinned & Brush Cut < 4m	0.535	4.262	0.158	1.236	1.418	1.894	1.807	0.905
	Regenerating post-natural < 4 m	0.528	2.471	0.850	0.960	0.768	1.592	1.182	1.279
	Deciduous < 7 m	0.527	3.223	0.393	1.098	0.797	2.125	0.903	1.401
	Flood Zone	0.403	3.527	0.766	2.143	2.568	0.839	2.105	0.691
	Coniferous w/o fir 4-7 m	0.403	1.664	0.641	0.811	0.847	1.082	1.104	0.701
	Coniferous w/ fir 4-7 m	0.347	4.114	0.865	1.385	1.429	1.984	1.616	1.248
	Dryland	0.083	3.768	0.104	1.552	0.580	2.112	0.614	1.117
	Wetland/Bog	0.062	2.810	0.217	1.876	0.438	0.910	0.545	0.592
	Water, Islands, and Other	0.055	3.544	0.172	2.178	0.158	1.276	0.677	1.779
	Mixedwood-unknown < 4 m	0.000	0.000	0.000	0.000	1.963	0.710	1.527	0.496
	Powerline	0.000	0.000	0.000	0.000	0.505	3.350	0.548	1.435
Special Interest Sites	25% areas	1.558	0.876	1.529	1.259	1.497	0.706	1.173	0.403
	75% areas	0.803	0.478	0.951	0.253	0.848	0.330	0.899	0.217
Elevation	Upland	1.515	0.895	1.032	0.421	1.108	0.987	1.218	0.766
	Midland	1.093	0.544	1.181	0.349	1.152	0.393	1.159	0.252
	Lowland	0.536	1.111	0.884	0.720	0.713	0.572	0.828	0.396
Road Density	Low	0.985	0.388	1.118	0.434	1.034	0.219	1.103	0.150
	Medium	0.847	1.027	1.187	0.846	1.037	0.620	0.846	0.343
	High	0.844	1.818	0.826	0.415	0.477	1.201	0.662	0.798
Distance to Water	250 - 500 m	1.268	0.552	1.079	0.384	1.234	0.317	1.170	0.233
	500 - 1000 m	1.053	0.788	1.097	0.603	0.962	0.451	1.083	0.515
	100 - 250 m	0.963	0.738	1.469	1.340	1.099	0.299	1.017	0.248
	1000 m +	0.860	1.844	1.022	0.655	0.623	1.799	0.751	0.762
	50 - 100 m	0.738	0.981	1.023	0.577	0.974	0.582	0.948	0.584
	25 - 50 m	0.673	1.399	0.853	0.656	0.908	0.662	0.893	0.595
	0 - 25 m	0.622	1.711	0.830	0.874	1.226	0.532	1.371	0.684
	0 m (open water)	0.106	2.383	0.468	1.172	0.306	0.684	0.754	0.938

Discussion

General findings

Here we assess moose habitat use in an Adapted Forestry Regime 20 years after its implementation, informed by Cree knowledge of important habitat variables such as land cover, elevation, road density, and distance to water. Female moose in Eeyou Istchee had a median mid-summer home range area that was 30 times larger than their median mid-winter home range. GLM analyses indicate that land cover was the most important variable driving habitat selection across all seasons and scales, but the most supported models included all variables that were explored. GLM analyses indicate that in winter, land cover, distance to water, and elevation most strongly drive habitat selection, while road density and 25% areas contribute to selection to a lesser degree. In summer, GLM analyses indicate that land cover, distance to water, and 25% areas most strongly drive habitat selection, while road density and elevation contribute to a lesser degree. In the RSF analysis, five core land covers were used by moose in both seasons and at both scales: mixedwood-deciduous forest over 7 m, mixedwood-coniferous forest over 7 m, deciduous forest over 7 m, coniferous forest with fir over 7 m, and mixedwood-coniferous under 7 m. Furthermore, 25% areas were used across both seasons and scales of analysis.

Performance of GLM models

In general, the GLM models varied in their ability to explain variation in moose habitat selection based on order of analysis and season. The second order analyses tended to be better performing and explain more of the variation than the third order analyses. The results indicated strong selection was occurring at the second order of habitat selection. Pseudo R-squared values cannot be directly compared across datasets making precise determination of the strongest and weakest models challenging. In general, study area scale models performed better than home range scale models. This difference is especially notable in the winter analyses, where the winter study area scale model had relatively good fit while the winter home range scale model had very poor fit. Given the very small winter home ranges, used and available points were either overlapping or very close to each other, which limited the strength of third order selection. Fourth order selection of specific resources for food and shelter within these restricted area winter moose yards would be an additional scale of selection worthy of evaluation, which could be accomplished through the analysis of moose video collars footage or by conducting browse and bedding surveys within winter home ranges.

For the second order analysis, the distribution of used and available points represented better opportunity to reveal strong selection, yet the model fit of selection models remained relatively low (9.8% - 21.4%). We attribute this low explanatory power to individual variability, inherent randomness in ecological systems, and potentially missed variables. Part of the process of working from qualitative interview data to inform quantitative analyses includes prioritizing themes according to their ease of quantitative modeling and their data accessibility. As such, a lot of very important topics in interviews, such as noise disturbance, different moose population management practices by different tallymen, traditional and changing hunting methods, and predation could not be included in the analysis. These themes may be key to developing stronger models of moose habitat use. Moreover, our habitat type analysis focused on forest stand classifications that prioritized the composition of canopy vegetation, whereas understory vegetation is more relevant to moose foraging and perhaps shelter (Boan et al., 2011; Kolstad et al., 2018; McInnes et al., 1992; Proulx and Kariz, 2005). Given there may be a diversity of understory vegetation available within a given canopy composition class and, conversely, similar understory vegetation may be available across multiple canopy types (Brosofske et al., 2001; Légaré et al., 2001), vegetation categories that emphasize understory vegetation composition (e.g., Lone et al., 2014; Melin et al., 2013) may yield stronger patterns of selection and help to highlight why particular canopy types are most or least selected.

Use of deciduous, mixedwood, and fir forest

Moose selected deciduous forest over 7 m in both seasons and at both scales of analysis, while avoiding deciduous forest under 7 m in both seasons and at both scales of analysis. Moose selection of deciduous forests has been well established in other literature. Deciduous species are often preferred moose forage, with plants like birch being highly valuable food sources (Hörnberg, 2001), and deciduous twigs being important winter browse, especially in Québec (Crête, 1988). Deciduous species such as birch, ash, and willow are understood to be highly digestible browse plants (Hjeljord et al., 1982) and studies comparing varying assortments of shrubs, deciduous trees, and coniferous trees have indicated that mountain ash, aspen, and willow are strongly preferred to alternatives such as pine, spruce, and juniper (Månsson et al., 2007).

Moose selected mixedwood-deciduous and mixedwood-coniferous forests over 7 m as well as mixedwood-coniferous forest under 7 m in both seasons and at both scales. Mixedwood-deciduous forest under 7 m was additionally selected in summer analyses. Moose selection of mixedwood forests has been documented as important in other literature (Jacqmain et al., 2008; Jung et al., 2009), with particular selection for mature mixedwood forests over 7 m tall being seen in mid-winter in seasonal habitat selection analyses (Jacqmain et al., 2008). In some cases, Traditional Ecological Knowledge (TEK) has indicated that mixedwood and deciduous forests have high food value to moose and have high habitat suitability (Tendeng et al., 2016). In Québec, mixedwood forests have been shown to be selected by moose in all seasons (Courtois et al., 2002), which is consistent with the results of our analysis.

In general, fir was identified as an important factor to moose use of coniferous stands in Eeyou Istchee. Moose selected for coniferous forests with fir over 7 m in both seasons and scales of analysis, and selected for coniferous forest with fir between 4 and 7 m tall in summer at the home range and study area scale. In comparison, coniferous forest without fir over 7 m tall was avoided and coniferous forest without fir between 4 and 7 m was avoided excpt in summer at the home range scale. This finding is reflected in other literature that has identified old growth fir as being critical moose habitat (Pierce and Peek, 1984), with tall fir stands being especially important and highly selected in midwinter (Jacqmain et al., 2008). Fir species may be preferred in winter because of its year round availability and high nutrient content compared to other coniferous species (Belovsky, 1981). In some areas, moose have been found to select fir species strongly and graze them with enough intensity to inhibit regeneration of fir species (Brandner et al., 1990) or cause ecological damage to fir stands (Bergerud and Manuel, 1968).

Use of regenerating stands and thinned or brush cut forests

Moose selected for regenerating stands post-forestry disturbance by moose in winter at the home range scale and summer at the study area scale, while avoiding these areas in winter at the study area scale and summer at the home range scale. Regenerating stands after a natural disturbance (fire, windthrow, epidemics) were slightly avoided in winter at both scales and summer at the study area scale, while being selected only in summer at the home range scale. Thinned or brush cut forests, in which we included regenerating stands in which brush-cutting or pre-commercial thinning was performed, were strongly selected in summer at the study area and home range scale, while being avoided in winter at both scales, albeit with very high individual variation in selection in winter at the study area scale. A limitation of our assessment of moose selection of disturbed stands was that our analysis considered stand height but not time since disturbance. As such, regenerating stands under 4 m tall were binned together, which could miss key differences within early successional stands. Moose habitat selection literature indicates that time since disturbance is very important to moose selection of disturbed stands. Mumma et al. (2021) found moose specifically selected 9 - 24 year old cuts while avoiding recent or older cuts, while (Crête, 1988) found that moose use of cutblocks peaked between 5 and 20 years post-disturbance.

Our findings that moose select stands regenerating post-forestry disturbance at some scales and seasons but not others is consistent with other studies in Québec (Courtois et al., 2002). The inconsistency in selection across orders of selection and seasons may reflect that moose selection of cutblocks only occurs at particular scales, which is reinforced by findings that moose selection of clearcut landscapes differs between within home range and study area scales (Courtois et al., 2002). Moose selection of cutblocks can be explained by the change in successional stage caused by the disturbance, which increases browse and forage supply (Collins and Schwartz, 1998). Extensive study has been done on the impact of wildfires on moose habitat use in areas outside Québec such as Alaska, where the effect of burn severity on moose habitat selection has been assessed. Some studies indicate that moose prefer to use low-severity burned sites in winter (Brown et al., 2018), while others found that high severity burns are selected (Lord and Kielland, 2015). Our disturbance history data lacked burn severity, which could be an important component to capturing habitat selection dynamics. Other studies have shown that moose avoid burns under 25 years old overall, which is consistent with our findings (DeMars et al., 2019).

Use of 25% areas

Moose selected 25% areas in both winter and summer, at both the study area and home range scale. Moose avoided 75% areas except in winter at the home range scale, where they were indifferent to 75% areas. When interpreting moose selection of 25% areas, it is important to note that the 25% areas were chosen by Cree tallyman because they represented high quality wildlife habitat, and in most cases high quality moose habitat during their preferred hunting season in the spring (Jacqmain 2008). Logging has occurred within these 25% areas since the implementation of the AFR, using specific management thresholds that reduce the intensity of disturbance by forestry. While the current analysis indicates that these 25% areas are selected by moose twenty years after implementation of the AFR, our analysis does not disentangle the relative importance of pre-existing and permanent habitat characteristics of these 25% areas relative to the effectiveness of special management methods that are applied within them. For example, 25% areas include more upland terrain and less lowland terrain relative to the study area as a whole, and we found moose selected upland terrain more than lowland terrain in both seasons. However, our GLM analysis indicated support for a model that included both 25% and elevation, suggesting that moose selection of 25% areas is not only due to their elevation profiles.

Similarities existed in the most common land cover categories contained in 25% areas and 75% areas, while differences existed in the less common land cover categories. Both 25% and 75% had the same five most common land covers, those being: 1) coniferous forest without fir over 7 m, 2) wetland/bog, 3) regenerating post-forestry, 4) coniferous forest without fir between 4 and 7 m, and 5) regenerating post-natural disturbance. However, 25% areas had higher proportions of the less common habitat types which were highly selected by moose such as tall mixedwood, deciduous, and coniferous forest with fir than 75% areas did. These habitats tended to be selected by moose in both seasons and at both scales. The 25% areas also had less wetlands/bogs, which were avoided by moose at all scales in all seasons. Other studies conducted in Labrador and Québec have emphasized the importance of hillsides (Jung et al., 2009) and elevated terrain (Jacqmain et al., 2008) as winter habitat for moose. The avoidance of bogs by moose was also seen in aerial surveys by Jung & Chubbs (Jung et al., 2009), and the selection of forest over 7 m high was also seen in fir forests in summer and mixedwood forests in mid-winter in Jacqmain et al. (2008). Thus, it seems likely that moose selection of 25% areas arises from both their intrinsic upland habitat quality in combination with 25% area prioritization of conserving 50% of 7 + m tall stands, as well as slower rotation periods that allow stands to regenerate to 7 m. Similar proportions of area in 25% areas and 75% areas were disturbed by forestry activities, and the higher selection of 25% areas could indicate favourable disturbance practices. Other literature has found that logging can strongly benefit moose if the logging techniques used are compatible with the protection and creation of moose preferred habitats (Collins and Schwartz, 1998).

Future directions

The analysis presented here has provided valuable insights and quantitative answers to key project partner questions about moose use of the 25% areas, regenerating stands, and different habitat types. A revised quantitative analysis could include individual moose and study year as random effects in a Generalized Linear Mixed Model analysis. Further exploration of moose habitat selection in the AFR could involve application of the general habitat selection approaches presented here to more targeted comparisons able to differentiate moose habitat preference of stands logged with and without 25% area measures, and in relation to time since disturbance, understory composition, patch size and proximity, and landscape configuration. More broadly, a mixed methods approach could be employed to interweave the quantitative habitat selection approaches presented here with a qualitative analysis of Cree understanding of moose habitat quality shared in the semi-structured interviews. These mixed methods approaches would enable the inclusion of themes brought up in interviews that are difficult to quantify and were, accordingly, not considered in the present analysis. Excluded variables known to be important to Cree land users include noise disturbance, wolf predation, and Cree and non-Cree hunting dynamics, changing traditional management practices, respect, and Cree ways of life. More inclusionary consideration of these drivers is invaluable to fully understanding how moose are using habitat in the AFR. A mixed methods analysis such as a Bayesian methodology, which has been employed by other studies interweaving knowledge and quantitative analyses (Alkhairy et al., 2020; Froese et al., 2017; O'Leary et al., 2009; Smith et al., 2007), could be an interesting future step, but would be dependent on expanded interview or relationship network data to develop model priors. This kind of analysis that interweaves Indigenous knowledge and quantitative analyses has key benefits that include increasing trust in study results, improving equity between scientists and community experts, and expanding on missing or unavailable data (Stern and Humphries, 2022).

Conclusion

Sustainable, conservation-compatible, and community-supported forestry in Eeyou Istchee, Québec, requires knowledge, cooperation, and participation that effectively balances logging activity with the protection of moose and their habitats. A multi-stakeholder steering committee, guiding a moose habitat study in the AFR, identified key questions about how moose are using different land covers, forest heights, and 25% areas. We performed home range, GLM, and RSF analyses for mid-summer and mid-winter habitat use by moose, informed by Cree expert knowledge and developed using GPS collar data from 38 female moose from 2018-2021. We identified important winter habitat as tall forest, deciduous forest, mixedwood forest, and coniferous with fir, and identified important summer habitat as short forest, flood zones, pre-commercially thinned forest, deciduous forest, mixedwood forest, and coniferous forest. Forests that were regenerating from forestry was mildly selected for at a study area scale but avoided within home range scale, and forests regenerating from natural disturbance was avoided at most scales of analysis. We determined that the 25% areas in the AFR have differing composition to the 75% areas, and moose strongly select home ranges within these 25% areas. Future research should expand upon this analysis by studying moose habitat selection in spring and autumn, using GLMM to assess the random effects of year and individual, and expanding on the inclusion of Cree knowledge by using a mixed-methods Bayesian analysis.

Appendix

Table A1. Definitions of habitat type categories used in GLM and RSF analysis.

Variable	Category	Definition
Land Cover	Alder	Non forested stands dominated by alder shrubs (i.e., alder grove along watercourses)
	Coniferous w/ fir 4 - 7 m	Coniferous forest with fir, canopy height between 4 and 7 m
	Coniferous w/o fir 4 - 7 m	Coniferous forest without fir, canopy height between 4 and 7 m
	Coniferous < 4 m	Coniferous forest of any kind, canopy height between 0 and 4 m
	Coniferous w/ fir > 7 m	Coniferous forest with fir, canopy height over 7 m
	Coniferous w/o fir > 7 m	Coniferous forest without fir, canopy height over 7 m
	Deciduous < 7 m	Deciduous forest, canopy height under 7 m
	Deciduous > 7 m	Deciduous forest, canopy height over 7 m
	Dryland	Mosaic of barren or semi-barren rock outcrop
	Flood Zone	Unregenerated previously flooded zones
	Mixedwood-coniferous < 7 m	Mixedwood forest, with coniferous dominated canopy species, canopy height under 7 m
	Mixedwood-coniferous > 7 m	Mixedwood forest, with coniferous dominated canopy species, canopy height over 7 m
	Mixedwood-deciduous < 7 m	Mixedwood forest, with deciduous dominated canopy species, canopy height under 7 m
	Mixedwood-deciduous > 7 m	Mixedwood forest, with deciduous dominated canopy species, canopy height over 7 m
	Mixedwood-unknown < 4 m	Mixedwood forest, with unknown dominated canopy species, canopy height between 0 and 4 m
	Powerline	Cleared areas under powerlines
	Regenerating post-forestry < 4 m	Forest stands regenerating after forestry-related disturbances, canopy height between 0 and 4 m
	Regenerating post-natural < 4 m	Forest stands regenerating after natural disturbances including fire, wind, and disease, canopy height between 0 and 4 m
	Thinned & Brush Cut	Forest stands that have been recently thinned (between 2013-2021) using brush-cutting or pre-commercial thinning, canopy height between 0 and 4 m
	Water, Islands, and Other	Major water bodies, both standing and moving water, small islands with productive forest under 1 ha, and all other habitats
	Wetland/bog	Non-treed bog habitat
Special Interest Status	25 % area	area inside Sites of Special Wildlife Interest to the Cree
	75 % area	area outside Sites of Special Wildlife Interest to the Cree
Elevation	lowland	bottom 25% quartile, 0 m hills
	midland	middle 50% quartiles, 0 - 22 m hills
	upland	top 25% quartiles, 22 - 321 m hills
Road Density	low	0 km/km^2 – 0.6 km/km^2
	medium	0.6 km/km^2 - 1.1 - 1.5 km/km^2
	high	1.5 km/km^2 - 10.9 km/km^2
Distance to Water	0 m	Locations directly on permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	0 - 25 m	Locations between 0 and 25 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	25 - 50 m	Locations between 25 and 50 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	50 - 100 m	Locations between 50 and 100 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	100 - 250 m	Locations between 100 and 250 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	250 - 500 m	Locations between 250 and 500 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	500 - 1000 m	Locations between 500 and 1000 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds
	1000 + m	Locations more than 1000 m away from permanent waterbodies including major and minor permanent rivers, lakes, and ponds

Table A2. Ranking of all assessed GLM models based on AIC, pseudo R-squared (Nagelkerke, 1991) for all seasons and orders of analyses.

Season	**Scale**	**Model**	**AIC**	**Pseudo R^2**
Winter	Study area	presence ~ land cover + elevation + distance to water + 25% areas + road density	169594	0.214
Winter	Study area	presence ~ land cover + elevation + distance to water + 25% areas	169680	0.213
Winter	Study area	presence ~ land cover + elevation + distance to water	170412	0.209
Winter	Study area	presence ~ land cover + elevation	170794	0.207
Winter	Study area	presence ~ land cover	172352	0.197
Winter	Study area	presence ~ elevation	197104	0.044
Winter	Study area	presence ~ distance to water	198116	0.037
Winter	Study area	presence ~ 25% areas	200034	0.025
Winter	Study area	presence ~ road density	203759	0.001
Winter	Study area	presence ~ 1	203856	0.000
Winter	Home range	presence ~ land cover + distance to water + elevation + 25% areas + road density	197239	0.044
Winter	Home range	presence ~ land cover + distance to water + elevation + 25% areas	197251	0.043
Winter	Home range	presence ~ land cover + distance to water + elevation	197336	0.043
Winter	Home range	presence ~ land cover + distance to water	197797	0.040
Winter	Home range	presence ~ land cover	197828	0.040
Winter	Home range	presence ~ distance to water	202924	0.006
Winter	Home range	presence ~ elevation	203561	0.002
Winter	Home range	presence ~ 25% areas	203809	0.001
Winter	Home range	presence ~ road density	203820	0.000
Winter	Home range	presence ~ 1	203890	0.000
Summer	Study area	presence ~ land cover + 25% areas + distance to water + elevation + road density	186809	0.098
Summer	Study area	presence ~ land cover + 25% areas + distance to water + elevation	188015	0.091
Summer	Study area	presence ~ land cover + 25% areas + distance to water	188274	0.089
Summer	Study area	presence ~ land cover + 25% areas	188830	0.085
Summer	Study area	presence ~ land cover	190804	0.073
Summer	Study area	presence ~ 25% areas	198450	0.023
Summer	Study area	presence ~ distance to water	198903	0.020
Summer	Study area	presence ~ elevation	200744	0.008
Summer	Study area	presence ~ road density	201288	0.005
Summer	Study area	presence ~ 1	202002	0.000
Summer	Home range	presence ~ land cover + distance to water + 25% areas + elevation + road density	201478	0.054
Summer	Home range	presence ~ land cover + distance to water + 25% areas + elevation	202638	0.047
Summer	Home range	presence ~ land cover + distance to water + 25% areas	202703	0.047
Summer	Home range	presence ~ land cover + distance to water	202910	0.045
Summer	Home range	presence ~ land cover	203281	0.043
Summer	Home range	presence ~ distance to water	207592	0.016
Summer	Home range	presence ~ 25% areas	209095	0.006
Summer	Home range	presence ~ elevation	209435	0.004
Summer	Home range	presence ~ road density	209443	0.004
Summer	Home range	presence ~ 1	210062	0.000

Table A.3. GLM regression coefficients for the most supported models for summer and winter analyses at the home range and study area scale.

	Winter (January-February)						Summer (July-August)					
	Study Area Scale			*Home Range Scale*			*Study Area Scale*			*Home Range Scale*		
Category	**Estimate**	**Std.Error**	**Pr(>|z|)**	**Estimate**	**Std.Error**	**Pr(>|z|)**	**Estimate**	**Std.Error**	**Pr(>|z|)**	**Estimate**	**Std.Error**	**Pr(>|z|)**
(Intercept)	-5.463	0.081	2.00E-16	-4.583	0.083	2.00E-16	-4.094	0.042	2.00E-16	-4.03	0.05	2.00E-16
landcover_alder	2.811	0.110	2.00E-16	1.815	0.112	2.00E-16	2.567	0.067	2.00E-16	2.33	0.07	2.00E-16
landcover_dryland	-0.801	0.334	0.016612	-1.044	0.335	0.001808	1.241	0.118	2.00E-16	1.34	0.12	2.00E-16
landcover_flood_zone	2.711	0.126	2.00E-16	2.281	0.127	2.00E-16	3.199	0.072	2.00E-16	2.84	0.07	2.00E-16
landcover_forest_coniferous_0to4m	2.377	0.117	2.00E-16	1.814	0.117	2.00E-16	2.433	0.075	2.00E-16	2.09	0.07	2.00E-16
landcover_forest_coniferous_firabsent_4to7m	1.700	0.111	2.00E-16	1.677	0.112	2.00E-16	1.692	0.069	2.00E-16	1.87	0.07	2.00E-16
landcover_forest_coniferous_firabsent_over7m	2.116	0.106	2.00E-16	1.690	0.107	2.00E-16	1.679	0.065	2.00E-16	1.80	0.06	2.00E-16
landcover_forest_coniferous_firpresent_4to7m	2.404	0.154	2.00E-16	2.659	0.157	2.00E-16	2.575	0.105	2.00E-16	2.13	0.10	2.00E-16
landcover_forest_coniferous_firpresent_over7m	3.052	0.110	2.00E-16	1.989	0.111	2.00E-16	2.600	0.071	2.00E-16	2.46	0.07	2.00E-16
landcover_forest_deciduous_over7m	3.885	0.109	2.00E-16	2.460	0.109	2.00E-16	2.826	0.071	2.00E-16	2.79	0.07	2.00E-16
landcover_forest_deciduous_under7m	2.479	0.142	2.00E-16	1.658	0.141	2.00E-16	2.222	0.109	2.00E-16	2.10	0.11	2.00E-16
landcover_forest_mixed_coniferousdominant_over7m	3.770	0.107	2.00E-16	2.292	0.108	2.00E-16	2.665	0.068	2.00E-16	2.31	0.07	2.00E-16
landcover_forest_mixed_coniferousdominant_under7m	3.163	0.109	2.00E-16	2.193	0.109	2.00E-16	2.777	0.070	2.00E-16	2.40	0.07	2.00E-16
landcover_forest_mixed_deciduousdominant_over7m	4.270	0.107	2.00E-16	2.484	0.107	2.00E-16	2.550	0.071	2.00E-16	2.20	0.07	2.00E-16
landcover_forest_mixed_deciduousdominant_under7m	3.065	0.118	2.00E-16	2.277	0.118	2.00E-16	2.410	0.087	2.00E-16	2.34	0.09	2.00E-16
landcover_forest_mixed_unknowndominant_0to4m	-10.921	46.592	0.814685				0.596	0.202	0.00322	2.32	0.21	2.00E-16
landcover_forest_regenerating_forestrydisturb_0to4m	2.547	0.107	2.00E-16	1.796	0.108	2.00E-16	2.249	0.066	2.00E-16	1.82	0.07	2.00E-16
landcover_forest_regenerating_naturaldisturb_0to4m	2.358	0.109	2.00E-16	2.079	0.110	2.00E-16	1.821	0.070	2.00E-16	1.91	0.07	2.00E-16
landcover_forest_thinned_0to4m	2.079	0.201	2.00E-16	0.498	0.198	0.011787	3.369	0.098	2.00E-16	2.67	0.09	2.00E-16
landcover_powerline	-10.697	49.006	0.827205	-10.348	36.295	0.775569	2.514	0.111	2.00E-16	2.21	0.11	2.00E-16
landcover_wetland	-0.335	0.124	0.006867	-0.126	0.125	0.311	1.153	0.067	2.00E-16	1.26	0.07	2.00E-16
25pct_area	0.432	0.017	2.00E-16	0.321	0.081	7.33E-05	0.584	0.014	2.00E-16	0.00	0.05	0.958206
dist_water_0-25	0.662	0.020	2.00E-16	0.209	0.082	0.010762	0.048	0.054	0.38234	-0.39	0.06	1.43E-11
dist_water_25-50	0.303	0.081	0.000184	0.184	0.077	0.016946	-0.354	0.059	2.09E-09	-0.34	0.05	1.32E-10
dist_water_50-100	0.183	0.083	0.026992	0.192	0.074	0.0096	-0.327	0.054	1.97E-09	-0.19	0.05	0.000182
dist_water_100-250	0.224	0.078	0.00395	0.250	0.074	0.000707	-0.203	0.051	7.53E-05	0.02	0.05	0.758769
dist_water_250-500	0.389	0.075	2.21E-07	0.206	0.074	0.005518	-0.088	0.051	8.48E-02	-0.07	0.05	0.194236
dist_water_500-1000	0.484	0.075	9.61E-11	0.345	0.076	5.83E-06	-0.339	0.052	5.03E-11	-0.29	0.06	1.98E-07
dist_water_1000+	0.177	0.075	0.01857	0.303	0.019	2.00E-16	-0.501	0.056	2.00E-16	0.17	0.01	2.00E-16
elevation_midland	0.428	0.077	2.59E-08	0.304	0.015	2.00E-16	0.208	0.015	2.00E-16	0.07	0.02	0.000678
elevation_upland	0.384	0.014	2.00E-16	-0.130	0.014	2.00E-16	0.008	0.020	0.68135	0.12	0.02	3.82E-12
road_density_medium	-0.153	0.016	2.00E-16	-0.060	0.016	0.00013	-0.120	0.015	7.03E-16	-0.35	0.01	2.00E-16
road_denity_high	-0.057	0.021	0.006428	-0.041	0.020	0.039799	-0.893	0.027	2.00E-16	-0.70	0.03	2.00E-16

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GENERAL DISCUSSION OF FINDINGS

General findings of Chapters 1 & 2

The objective of this thesis was to advance understanding about the inclusion of experiential wildlife knowledge into quantitative habitat and population analyses. This was accomplished by contributing a systematic review of the methods, successes, and limitations defining past attempts at experiential wildlife knowledge inclusion and through a case study evaluating the effects of an adapted forest regime on moose habitat selection informed by Cree knowledge. A multi-stakeholder steering committee which guides the moose habitat quality project identified shared goals of assessing how moose are using different land covers, 25% areas, and other habitat types 20 years after the AFR was implemented. To meet these shared goals, I performed a quantitative analysis based on GPS moose location and GIS land data, acting as one phase of a broader project using knowledge co-production methods.

To identify appropriate methodologies to accomplish this, I conducted a systematic review in Chapter 1 that explored previous methods employed by other primary literature. In this Chapter, we found that across 49 systematically collected articles, a wide variety of methodologies exist and have been used to quantitatively interweave local, expert, and Indigenous knowledge with wildlife science. These methods are used globally and across many taxa, with the majority of studies focusing on similar species to moose: large, charismatic, and hunted species. Articles did discuss knowledge interweaving as being valuable for rare or cryptic species which are hard to observe through structured field surveys, however the dominance of large and charismatic species reflected in this review may indicate that experiential knowledge is most robust with these kinds of wildlife. As such, rare or cryptic species may require supplemental information such as camera trap information, which is not as dependent on human observation. Hunters and trappers were the most frequently involved knowledge holders, and in person interviews were the most common elicitation methods to retrieve data. The development of habitat models based on GLMs or GLMMs developed from habitat relationship networks or model covariate information from knowledge holders was very common. We also identified 6 key benefits, limitations, and improvements for interweaving knowledge into quantitative wildlife science. The identified benefits pertained to recognizing the validity of diverse knowledges, increasing trust in science and management, improving equity between scientists and stakeholders, expanding data, relatively low costs, identifying knowledge gaps, identifying agreements and disagreements, and improving temporal transferability of models. The limitations pertained to the potentially local scale and lack of systematic structure of knowledge, mismatch between knowledge forms, the challenging nature of assessing interviewer and respondent reliability, bias, lack of social science training in researchers, and communication challenges. We conclude by recommending that future articles: 1) avoid assuming that quantitative data can be used to assess the reliability of other knowledge forms; 2) use multiple statistical methods to assess congruence or disagreement between data sources; 3) develop standardized methods to accommodate uncertainty and observer reliability; 4) meaningfully include knowledge holders in more study phases, including study design and member checking; 5) discuss intellectual property rights, knowledge ownership, and knowledge protection; 6) acknowledge and discuss power differences between researchers and knowledge holders; and 7) assess and communicate knowledge holder benefits or negative outcomes in addition to science outcomes.

In Chapter 2, we used the toolbox of methodologies identified in Chapter 1 to accomplish our objectives. We followed a similar methodology to the most commonly used methods described above. After iterative development of an appropriate study structure with the steering committee, we conducted in-person interviews to develop habitat relationship networks and use that knowledge to inform the development of model variables. These variables were then explored in GLM and Manly selection analyses. While the majority of the analytical stage was performed using quantitative data such as GPS collar locations and GIS land data, my project was informed by and developed iteratively with Cree knowledge. In doing so, we assess home range behaviour and habitat selection in summer and winter. We identified summer home ranges as substantially larger than winter home ranges, and identified key similarities and differences in summer and winter habitat use. In both seasons, mixedwood forests, deciduous forests, and coniferous forests were highly used habitat, but in summer shorter forests, flood zones, and pre-commercially thinned or brush cut forests emerged as the most valuable habitat, whereas in winter tall forests were the most valuable. In both seasons, moose selected for regenerating post-forestry stands at a coarse scale, but avoided them at a fine scale. In both seasons of analysis, moose chose to position their home ranges in areas with a higher proportion of 25% areas than was generally available in the study area. We identified the 25% areas as having more upland habitat, less lowland habitat, less water and wetlands, more forested area, and more roads than 75% areas.

Connections between Chapters 1 & 2

We put many of the methods identified and recommendations made in Chapter 1 into practice for Chapter 2, but not all our recommendations were met. Here, we will explore each recommendation and discuss whether or not we achieved it. The first recommendation: *1) avoid* *assuming that quantitative data can be used to assess the reliability of other knowledge forms,* was met. In this analysis, we respected statements from members of the steering committee that indicated that Cree experts were not interested in a comparison of "scientific knowledge" and Cree knowledge, or in using either type of knowledge to assess the accuracy of the other. As such, we did not use the results of the analysis in Chapter 2 to validate or assess the knowledge elicited in the interviews, instead using the knowledge to inform and develop the quantitative analysis. The second recommendation: *2) use multiple statistical methods to assess congruence or disagreement between data sources,* was partially met. We did perform three separate analyses (home range, GLM, and Manly Selection), but did not directly compare them to assess congruence or disagreement, or provide assessment of which method worked best with Cree knowledge. The third recommendation: *3) develop standardized methods to accommodate uncertainty and observer reliability,* was not met. Because we interwove Cree knowledge in the study design, pre-modeling, and follow-up stages and not as directly input data, we did not employ specific methods to address uncertainty. The fourth recommendation: *4) meaningfully include knowledge holders in more study phases, including study design and member checking,* was met. We included knowledge holders in three out of the five study stages we identified in Chapter 1, by including Cree experts in the consultation/study-design, pre-modeling/analytical-approach, and follow-up/member-checking stages. The fifth recommendation: *5) discuss intellectual property rights, knowledge ownership, and knowledge protection,* was not met. Because Chapter 2 did not directly report the knowledge elicited in the interviews or depict the habitat relationship networks, and the networks were used to develop model variables in an unstructured way, we did not feel that Chapter 2 required this. The sixth recommendation: *6) acknowledge and discuss power differences between researchers and knowledge holders,* was not met. This project was developed under the authority of a multi-stakeholder steering committee, and all stages were communicated to and approved by the committee. As per the instruction of the committee, the role of this project was to provide answers only and not present any management recommendations. The seventh recommendation: *7) assess and communicate knowledge holder benefits or negative outcomes in addition to science outcomes*, was not met. Because of the timing of the study, we did not have time to assess the benefits that the results may have on communities. Future research in the broad project that uses the quantitative results developed in Chapter 2 should explore the benefits the results have for the communities.

Future directions

Subsequent research on moose habitat quality in Eeyou Istchee could benefit from several approaches focused on improving data, increasing comprehensiveness of results, and greater inclusion of Cree knowledge. To improve data, future studies should focus on adjusting the scale of data to the scale of habitat selection at the third order. Especially in winter, selection behaviour was challenging to identify because of the mismatch between very small home ranges and stand-scale habitat data available. Research could be done using camera collars or wildlife cameras to perform fourth order selection analyses within moose home ranges to assess what resources they are using at a fine scale. Fourth order selection analyses, which assess which resources are used in a stand, could be combined with this second and third order analysis by assigning uses such as feeding and bedding to certain stands based on analysis of camera collar data. This expanded information could be used to separate the GPS collar data into “feeding locations” and “bedding locations” to assess what habitat types are used for certain behaviours in summer and winter using the same GLM and RSF analyses.

Vegetation information could also be improved with the inclusion of LiDAR data in the future, which could expand the information available by refining canopy height data and may provide information on stand structure. Furthermore, because only annual resolution was available for the location and timing of forestry cuts in the study area, refining the estimated date of cutblock logging through automated algorithms or through LiDAR data would be beneficial. In other Canadian forests, new algorithms such as "Shrinking Latency in Multiple Streams" (SLIMS) has been used to detect changes in forests from fire and harvesting combined (Cardille et al. 2022). With time, these algorithms could be retrained for the forest in the AFR to detect the date and location of forest cuts to a higher degree of accuracy.

The comprehensiveness of results could be improved by performing the same analyses for shoulder-season (spring/calving and fall/rut) with the same data used in this analysis. We focused on mid-summer and mid-winter as highly dichotomous seasons in which we expected to see strong selective behaviour because of the very different ecological conditions, and did not explore spring or fall habitat in the interest of time. Expanding this analysis to four seasons would increase the comprehensiveness and understanding of how moose use habitat in the AFR throughout the year. Furthermore, throughout the iterative process of developing the project in steering committee meetings, some project partners raised concerns about the analysis being limited to female moose. In Eeyou Istchee, while female moose and calves are hunted, male moose are hunted more and the sex ratio of the population indicates significantly more hunting pressure on males than females. We focused on female moose because in cervid populations females are known to be primary drivers of population health (Clark & Tait, 1982, Solberg

2002), however expanding the analysis to male moose could improve results applicability for some stakeholders.

Finally, future work on moose habitat selection in Eeyou Istchee should include a mixed methods approach, in which a separate qualitative and quantitative analysis are interwoven. This could be very beneficial to further explore moose habitat use in the region. Our models did not explain the majority of variation in moose habitat selection, and it is possible that this can be attributed to having filtered out variables brought up in interviews with tallymen that did not have available data or would have been challenging to quantify. These themes may have been highly pertinent to moose habitat selection, and a mixed methods approach could explore that dynamic and these variables. Furthermore, such an analysis could provide more opportunities to include knowledge holders in the modeling and post-modeling stage. Our study included knowledge holders in the consultation/study-design, pre-modeling/analytical-approach, and follow-up/member-checking stages. A mixed methods study that does a separate qualitative analysis could be able to interweave Indigenous knowledge in the modeling/data and post-modeling/validation stage.

CONCLUSION

In this thesis, I aimed to use knowledge co-production methods to study moose habitat use in the AFR in Eeyou Istchee. The project was guided by a steering committee comprised of multiple stakeholders, including Cree community representatives. Project objectives were to quantitatively assess how moose were using habitat in the AFR 20 years after it was put into place, with particular focus on land covers, 25% areas, and forested stands. In Chapter 1, I explored 49 articles which accomplished similar goals, and surveyed the broad array of methodologies used to do so. I also identified 5 keys stages of a study in which to involve knowledge holders, and concluded with a set of recommendations. In Chapter 2, I put some of these surveyed methodologies into action, including participating in in-person interviews to develop habitat relationship networks that informed model development and variable selection, and conduct GLM analyses to assess habitat use. Knowledge holders were included in three of the project stages, and we met many of our recommended improvements. Through this process, I identified tall deciduous, mixedwood, and coniferous with fir forest, upland elevation, and areas far from water as important winter habitat. I identified important summer habitat as tall deciduous, mixedwood, and coniferous with fir forest, flood zones, thinned forest, midland habitat, 25% areas, and sites within 25 m of water. These findings were communicated both to the steering committee that developed the project, and the communities in follow up workshops. We recommend further analyses refine the data, expand the analysis to male moose and shoulder seasons, and employ this quantitative analysis in a mixed-methods analysis to increase use of Cree knowledge and increase results applicability.

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