University of Alberta

Incorporating natural disturbance and heritage sites in protected areas design

by

Shawn James Leroux ©

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

in

Conservation Biology

Department of Renewable Resources

Edmonton, Alberta

Spring 2006

Your file Votre référence
ISBN: 0-494-13840-8
Our file Notre référence
ISBN: 0-494-13840-8

Abstract

Protected areas are the cornerstone of conservation planning. I developed a spatially explicit, dynamic simulation model, CONSERV that simulates vegetation community dynamics and fire and used CONSERV to 1) evaluate the efficacy of conventional protected areas design methods, and 2) develop a theory for the size of protected areas required to maintain ecological processes. I also compared the spatial overlap of indigenous heritage sites and protected areas. I observed that most protected area networks designed to capture conventional conservation targets did not maintain their initial targets through time. I also found low overlap between heritage sites and protected areas in my study region. I proposed the minimum dynamic reserve concept as a quantitative and practical framework for determining the size of protected areas. I conclude that natural disturbance and heritage sites should be considered a priori for effective and comprehensive protected areas design, and provide guidelines to achieve this outcome.

Acknowledgements

My mom and dad have provided unconditional love and support throughout my life. No words could accurately depict the gratitude I have for having such great parents. Mom and dad: thank you so much. My brother has been a great role model throughout my life and has provided great support during my thesis. Darryl, you are an inspiration, I wish you the best in your future plans. I was fortunate to bump into Amy on the basketball courts at the U of A. Amy, you have provided amazing support throughout my degree, whether it be in person or from a distance, you were always there.

I consider myself the luckiest graduate student in the world for having the opportunity to work in Fiona Schmiegelow's lab and on the BEACONs project. Fiona is infinitely insightful, and provided a great learning environment. I could not have a better role model. My committee members Steve Cumming, Eric Butterworth, and Stan Boutin provided timely comments and discussion throughout my program. In particular, my thesis benefited from frequent discussions with Steve, who is a pioneer in natural disturbance modelling. Eric facilitated all duck and wetland components of my thesis. Bob Lessard (a.k.a. VB king) and I spent some late nights developing CONSERV. John Nagy helped with the caribou modelling for my thesis and Jamie Smithson helped with caribou field work. BEACONs; Fiona, Steve, Scott, Kim, Meg, Shasta, Ben, Alastair, Nadele, Dan, Lee, and Trish provided good times and intellectual stimulation that have benefit my training. Amy, Scott, Kim, and Alastair provided excellent comments on earlier drafts of my thesis; thanks! Scott always kept an open door. Our discussions and your timely advice have enriched my graduate experience.

A special thanks to the Schmiegelites, Layla, Joanne, Kim, Meg, Scott, Roger, Theresa, Landon, Shannon, Sophie, Lee, Nicky, Bob, Kathy, Simon, Dan, Trish, Tracy, Nadele and Ben. They have been great friends and have always been willing to discuss thesis stuff. I will not be able to find a better lab group! Thanks to my intramural buddies for the good times, you will not be forgotten. Once a week I put everything aside, grabbed my axe and saluted the gods of Rock n' Roll. To all my band mates, thanks for rockin' with me, I hope you all keep liberating people with your music!

My work would not have been possible without personal funding from the Natural Science and Engineering Research Council of Canada, the Alberta Ingenuity Fund, and the University of Alberta. Additional funding was provided by the Canadian Boreal Initiative. I thank I. Kritsch, S. Higgins, and M. Clark from the Gwich'in Social Cultural Institute for support and use of the heritage data for my analysis and the Gwich'in Renewable Resource Board for supporting caribou data collection and analysis. Ducks Unlimited Canada, Gwich'in Renewable Resource Board (GRRB), Government of the Northwest Territories (GNWT), Inuvialiut Game Council, Environment Canada, and Yukon Territorial Government provided earth cover data used in my analyses. GNWT, GRRB, Habitat Stewardship Program, Polar Continental Shelf Program supported caribou data collection. D. Milne (Government of Yukon) and B. Croft and S. Corey (GNWT) provided forest fire history data.

Finally, I have developed a passion for conservation science by traveling across Canada on many occasions, visiting magical places and interesting people across this country. A special thanks to the landscapes and people that inspire my work: in discovering you, I have discovered myself.

Table of Contents

List of Tables

List of Figures

Chapter 1

1. General Introduction

Four objectives of regional conservation planning have been identified (Noss & Cooperrider 1994): 1) represent, in a system of protected areas, all native ecosystem types and seral stages across their natural range of variations; 2) maintain viable populations of all native species in natural patterns of abundance and distribution; 3) maintain ecological and evolutionary processes, such as natural disturbance regimes, hydrological processes, nutrient cycles, and biotic interactions; and 4) design and manage the system to be resilient to short term and long-term environmental change.

Practitioners of systematic conservation planning (*sensu* Margules & Pressey 2000) have made considerable progress in achieving the first two objectives of conservation planning: representation of natural variation and maintenance of viable populations (Noss et al. 2002; Cowling et al. 2003). However, conventional conservation planning methods provide few examples of how to incorporate the two final objectives of conservation planning: maintenance of ecological and evolutionary processes and system resilience (but see Carroll et al. 2003; Cowling et al. 2003; Mackey et al. 2005; Pyke & Fischer 2005).

Protected areas are the cornerstone of conservation planning across the world (Meir et al. 2004; Rodriguez et al. 2004). Nevertheless, protected areas have historically been selected in an ad hoc manner (Pressey & Tully 1994). Such protected areas are often in areas that were the easiest - politically and economically - to protect (Pressey & Tully 1994). Most protected areas do not achieve regional conservation objectives as defined by Noss & Cooperrider (1994) (Pressey & Tully 1994; Gurd & Nudds 1999; Gurd et al. 2001; Wiersma et al. 2004). Systematic conservation planning methods were developed in response to the shortcomings in ad hoc protected areas design (Margules & Pressey 2000). The strengths of systematic conservation planning are its explicit and structured approach to protected areas design (Margules & Pressey 2000; Pressey 2004). However, a weakness in the conventional application of systematic conservation planning methods is the failure to account for the dynamic nature of ecosystems (Moilanen & Cabeza 2002; Bengtsson et al. 2003; Pressey 2004). The conventional application of systematic conservation planning uses site selection algorithms (Possingham et al. 2000) to identify candidate protected areas that satisfy targets for focal species, environmental representation, and special elements (Cabeza & Moilanen 2001; Margules et al. 2002; Noss et al. 2002; Warman et al. 2004). These candidate protected areas are then considered static entities for planning purposes.

1.1 Site selection algorithms

Site selection algorithms are tools that facilitate the identification and selection of candidate protected areas designed to satisfy a suite of conservation targets. Considerable theoretical research has been conducted on the design of site selection algorithms for protected areas delineation (Possingham et al. 2000; Cabeza & Moilanen 2001; Cabeza 2003; Cerdeira et al. 2005; Moilanen 2005a, 2005b; Williams et al. 2004; Williams et al. 2005). Site selection algorithms are usually designed to identify the minimum number of sites or area to achieve all stated targets (i.e., minimum area problem), although some are designed to maximize representation of stated targets in a limited number of sites or area (i.e., maximal coverage problem) (Possingham et al. 2000; Cabeza & Moilanen 2001). To provide a solution to the minimum area problem, local or global heuristic algorithms can be used (Possigham et al. 2006). Local heuristic algorithms select sites in a stepwise manner, choosing the best site at each step (Pressey et al. 1993; Possingham et al. 2006). Global heuristic algorithms begin by generating a random set of sites. Then, at each step, a site is removed or added to the random set and the value of the new set is compared to the initial one (Possingham et al. 2000; Cabeza & Moilanen 2001; Possingham et al. 2005). Simulated annealing is an example of a global heuristic algorithm, and is the most common algorithm used in protected areas design.

To make these site selection algorithms more relevant to practical protected areas design, several authors have developed software that is freely available on the internet. The most common programs that implement site selection algorithms are Sites (Andelman et al. 1999), C-Plan (National Parks & Wildlife Service 2001), and MARXAN (Ball & Possingham 1999). These tools are being used extensively for conservation planning around the world (Ferrier et al. 2000; Noss et al. 2002; Carroll et al. 2003; Cowling et al. 2003; Leslie et al. 2003; Warman et al. 2004; Oetting et al. 2006) but not without caution (Fischer & Church 2005). There are usually three main conservation features prioritized using site selection tools: focal species, environmental representation, and special elements mapping (Noss & Cooperider 1994; Noss et al. 2002; Beazley et al. 2005).

1.2 Focal species modelling

There is increasing concern over the state of global biomes (Foley et al. 2005; Hoekstra et al. 2005), especially decreases in global biodiversity (Sala et al. 2000; Cardillo et al. 2005). It is no surprise, then, that species are a focal point for conservation across the world. From biodiversity hotspots (Myers et al. 2000) to managed landscapes (Nielsen et al. 2004), focal species receive most conservation attention. Focal species – those planners decide to focus on (*sensu* Armstrong 2002) - have also played a prominent role in protected areas design. It would be impossible to measure all species on the planet; therefore, conservation biologists often use a sub-set of species to focus their conservation strategies on in the hopes that they act as surrogates for other species. The most common focal species used in protected areas design are those with large area requirements (Lambeck 1997; Caro & O'Doherty 1999; Coppolillo et al. 2004; Roberge & Angelstam 2004). Planners assume that protecting species with large area requirements will also provide protection for other species found in the same areas. This assumption is rarely tested and the few studies that have tested this assumption have found conflicting results (Berger 1997; Howard et al. 1998; Fleishman et al. 2001; Suter et al. 2002; Rubino & Hess 2003). For focal species to be useful in protected areas design, assumptions must be tested, and the most appropriate applications are for modelling, monitoring, and marketing (Harrison et al. 2006). Habitat suitability models (Beazly et al. 2005), resource selection models (Noss et al. 2002), and population viability models (Carroll et al. 2003) have been used to identify suitable areas for focal species protection.

1.3 Environmental representation

By incorporating environmental representation into protected areas design, conservation planners attempt to maintain functioning examples of ecosystems, landforms, communities, populations, and species (Noss & Cooperrider 1994; Noss 1997). Conservation planners set environmental representation targets to capture landscape features in protected areas. Environmental representation targets figure prominently in conservation plans (Olson & Dinerstein 1998; Wessels et al. 1999; Noss et al. 2002; Pressey et al. 2003; Beazley et al. 2005; Oetting et al. 2006) but most of these targets were set arbitrarily with little ecological support. For environmental representation targets to be useful in protected areas design, they must be based on ecological principles.

1.4 Special elements mapping

Special elements are usually rare species or other unique features like heritage sites – sites recognized as sacred or important for indigenous peoples (Noss 1997; Groves et al. 2002). Special elements are often localized features that deserve special protection (Beazley et al. 2005). Rare species locations and pristine sites figure prominently in many conservation plans (Noss et al. 2002; Carroll et al. 2003; Cowling et al. 2003; Beazley et al. 2005; Oetting et al. 2006) but heritage sites are often omitted from many conservation plans (Carroll et al. 2003; Warman et al. 2004; Wiersma & Urban 2005; Venevsky & Venevskaia 2005; Deguise & Kerr 2006). This is especially troubling when conservation planning occurs on indigenous lands. For special elements to be useful for protected areas design, local communities, government agencies, and non-government agencies should be involved in mapping special elements in their regions.

1.5 Irreplaceability and vulnerability

Conservation planners input targets for focal species, environmental representation, and special elements into a site selection tool to identify candidate protected areas that satisfy their criteria. The sites with the greatest irreplaceability, i.e., that contribute the most to the conservation targets for the features contained (Ferrier et al. 2000), are considered those sites of highest conservation value (Pressey et al. 1993; Margules & Pressey 2000; Ferrier et al. 2000; Pressey & Taffs 2001). Potential protected areas (i.e., sites of high irreplaceability) are then ranked according to their vulnerability (i.e., status of threats) (Margules & Pressey 2000; Noss et al. 2002) or according to probabilistic scheduling approaches (Drechsler 2005) or dynamic implementation (Meir et al. 2004; Oetting et al. 2006; Wilson et al. 2006), as it is unlikely that all potential protected areas will be selected at once because of limited funds, land-use conflicts or jurisdictional barriers (Costello & Polasky 2004).

Setting targets for focal species, environmental representation, and special elements may not capture all conservation features; however, by combining these analyses, we can design more effective protected areas that maintain a variety of conservation features. A major weakness in the conventional application of systematic conservation planning is that it is static, with little consideration of the dynamic nature of ecological systems (Cabeza & Moilanen 2001; Moilanen & Cabeza 2002; Meir et al. 2004; Pressey 2004) and decision-making (Meir et al. 2004; Oetting et al. 2006; Wilson et al. 2006). Some studies have begun to incorporate population dynamics (Moilanen & Cabeza 2002; Carroll et al. 2003), landscape change (Pressey et al. 2004; Pyke & Fischer 2005), uncertainty modelling (Halpren et al. 2005), and dynamic decision-making (Meir et al. 2004; Oetting et al. 2006; Wilson et al. 2006) into conservation planning but few studies have applied these methods in relatively intact ecosystems. It may be particularly relevant to incorporate system dynamics in protected areas planning in intact areas because intact areas are still shaped largely by dynamic natural disturbances.

1.6 Protected areas size

Reserve size is a fundamental consideration in protected areas design, especially when designing protected areas in regions with large natural disturbances. Pickett and Thompson (1978) proposed the minimum dynamic area concept (MDA) as a guide for the design of protected areas to buffer against landscape disturbance and maintain ecological processes. The MDA concept holds promise but some authors have argued that it is a theoretical construct that is unachievable in practice (Baker et al. 1992; Fries et al. 1998). Several authors; however, have proposed guiding principles for the size of protected area required to maintain a landscape in a quasi-equilibrium state. Shugart & West (1981) proposed that a landscape in quasi-equilibrium must be at least 50 times the mean fire size of a region (but they simulated fires of uniform size), whereas Busing and White (1993) argue that a smaller ratio would result in compositional equilibrium. Peters et al. (1997) proposed that a minimum dynamic area be the largest disturbance event expected in a 500-1000 year period, while Kneeshaw & Gauthier (2003) proposed that protected areas be 100 times the average fire size of a region. These estimates are best guesses with few data to support them. A general, practical theory for protected area size remains a frontier in conservation biology (Poiani et al. 2000), and it is especially important to develop these criteria for the world's remaining intact areas like the boreal region where opportunities for pro-active conservation planning exist.

1.7 Boreal region

Canada's boreal forest supports some of the last wilderness areas in the world (Sanderson et al. 2002), containing approximately one quarter of all intact forests remaining globally (Bryant et al. 1997). Although not a biodiversity hotspot (Myers et al. 2000), the boreal forest retains much of the planet's biomass (Mittermeier et al. 2003), has the last intact mega-faunal assemblages in North America (Laliberté & Ripple 2004), supports over one-third of the breeding populations of North American migratory land birds (Blancher 2003; Blancher & Wells 2005), maintains a variety of ecosystem services (e.g., hydrological control, nitrogen fixation, carbon sequestration), and has recently been identified as one of the world's hotspots of latent extinction risk (Cardillo et al. 2006), underscoring the need for the protection of this ecosystem (Mittermeier et al. 2003). While the boreal region is not highlighted in most global analyses of priority areas for conservation (Myers et al. 2000; Rodriguez et al. 2004; Orme et al. 2005; Lamoreux et al. 2006), recent studies have begun to recognize the boreal region as a unique conservation opportunity (Kareiva & Marvier 2003; Cardillo et al. 2006).

The boreal region is a dynamic system shaped by large-scale natural disturbances like forest fire (Johnson 1992) and insect outbreaks (McCullough et al. 1998). Climate change is also predicted to have significant effects on temperature and precipitation in the boreal region (Melillo et al. 1993; Gitay et al. 2002). Specifically, most climate change scenarios predict higher temperatures and more precipitation across the boreal region (Gitay et al. 2002). Climate change may also increase the frequency and severity of forest fires in the boreal region (Flannigan & Van Wagner 1991; Stocks et al. 1998).

In addition to natural disturbance and climate change, the Canadian boreal region is under increasing pressure from resource development activities (Schneider et al. 2003). Oil and gas exploration in the Mackenzie Valley, Northwest Territories is predicted to increase significantly over the next decade (Cizek and Montgomery 2005; Holroyd and Retzer 2005) yet few studies have investigated the potential effects of this development activity on wildlife and ecosystem integrity in the region (but see Johnson et al. 2005). The cumulative effects of natural disturbance, climate change, and development activities on wildlife and landscapes in the boreal region may be more pronounced than any individual process presently influencing the system dynamics (Nellemann and Cameron 1998; Schneider et al. 2003; Johnson et al. 2005). Protected areas can contribute to pro-active and comprehensive conservation planning for the boreal region and conservation planning theory provides a starting point for the design of protected area networks for the boreal region.

At present only 5.6 % of the boreal region has permanent protection and another 3.6 % has interim protection (Canadian Boreal Initiative 2005). To counter the pressure from resource extraction industries, there has been increasing pressure to establish additional protected areas in this region. The Boreal Conservation Framework, developed by a collective of indigenous communities, industrial companies, and non-governmental organizations from across the boreal region, calls for the protection of at least 50 % of the region with sustainable resource management in the remainder (Canadian Boreal Initiative 2005). Here, I present research on protected areas design in the boreal region of Canada, with case studies in the northern Mackenzie Valley region of Northwest and Yukon Territories (Fig. 1-1). My specific objectives were to 1) evaluate the efficacy of protected areas designed based on conventional conservation planning methods, 2) investigate the complementarity of indigenous heritage sites and protected areas, and 3) enhance the theoretical and practical foundations for determining the size of protected areas required to buffer against landscape disturbance and maintain ecological processes.

1.8 Thesis overview

In Chapter 2, I develop a spatially explicit, dynamic simulation model, CONSERV to evaluate the ability of protected areas designed using conventional conservation planning methods to maintain their initial targets through time. CONSERV models vegetation community dynamics and fire, which has historically been the dominant influence on vegetation change in the study area. I parameterized CONSERV using fire history records. I designed a suite of protected areas networks to capture minimum, medium, and maximum targets for woodland caribou habitat, high quality wetlands, vegetation, and waterbodies and fixed targets for environmentally significant areas and heritage sites. The protected area networks were also designed with two different measures of connectedness: low and high. I evaluated the ability of each protected areas network to maintain the original targets over time under an active natural disturbance regime, as a first step in addressing future landscape change.

In Chapter 3, I evaluate the spatial overlap between indigenous group heritage sites and protected areas designed to capture conventional conservation targets for focal species, high quality wetlands, vegetation types, and environmentally significant areas. There is great potential for protected areas design in the boreal region of Canada but conservation planning can not occur in isolation of current community land-use planning processes.

In Chapter 4, I refine the MDA concept and provide a theoretical and practical foundation for determining the size of protected areas required to buffer against landscape disturbance and maintain ecological processes. My conceptual framework provides quantitative and practical methods for identifying large protected areas that may serve as system-level ecological benchmarks. I apply this framework in a case study and demonstrate the usefulness of this concept for protected areas design.

This thesis provides three important contributions to protected areas design theory. I developed a novel simulation model that can be used to evaluate and design protected areas and demonstrated the usefulness of this modelling framework in protected areas design. I quantified the relationship between heritage sites and protected areas, as a guide to comprehensive conservation planning. Finally, I developed a conceptual framework for determining the size of protected areas required to buffer against landscape disturbance. While my research has been mostly theoretical, the insights gained provide important information for practical conservation planning. I hope to translate my findings into more practical terms through discussions with indigenous communities, government agencies, and non-governmental organizations.

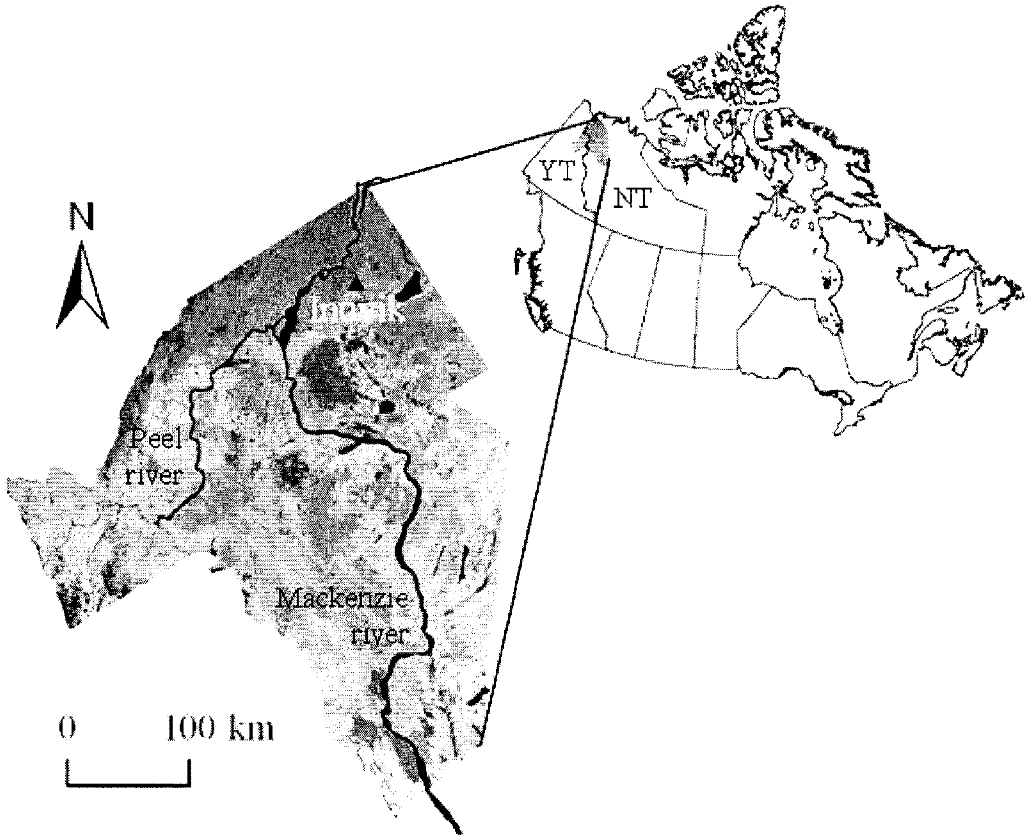

Figure 1-1 Study region used in the three chapters of this thesis. The study region is primarily in the Northwest Territories (NT) with portions in the Yukon Territory (YT). The largest city in the study area is Inuvik. Only portions of the study area were used for chapters 1 and 2.

2. Literature Cited

Andelman, S., I. Ball, F. Davis, and D. Stoms. 1999. SITES v 1.0: an analytical toolbox for designing ecoregional conservation portfolios. The Nature Conservancy, Boise, Idaho. Available from http:www.biogeog.ucsb.edu/projects/tnc/overview.html (accessed September 2003).

Armstrong, D. 2002. Focal and surrogate species: getting the language right. Conservation Biology **16**:285-286.

Baker, W. L. 1992. The landscape ecology of large disturbances in the design and management of nature reserves. Landscape Ecology **7**:181-194.

Ball, I. and H. Possigham. 2000. MARXAN (V1.8.2): marine reserve design using spatially explicit annealing, A manual. The University of Queensland, Brisbane, Australia. Available from http:www.ecology.uq.edu.au/index.html?page=27710 (accessed April 2005).

Beazley, K., L. Smandych, T. Snaith, F. MacKinnon, P. Austen-Smith Jr., and P. Duinker. 2005. Biodiversity considerations in conservation system planning: map-based approach for Nova Scotia, Canada. Ecological Applications **15**:2192-2208.

Bengtsson, J., P. Angelstam, T. Elmqvist, U. Emanuelsson, C. Folke, M. Ihse, F. Moberg, and M. Nystrom. 2003. Reserves, resilience and dynamic landscapes. Ambio **32**:389-396.

Berger, J. 1997. Population constraints associated with the use of black rhinos as an umbrella species for desert herbivores. Conservation Biology **11**:69-78.

Blanchard, P. 2003. Importance of Canada's boreal forest to landbirds. Bird Studies Canada, Port Rowan, Ontario.

Blancher, P. and J. Wells. 2005. North America's bird nursery: the boreal forest and its global responsibility towards sustaining bird populations. Boreal Songbird Initiative, Ottawa, Ontario.

Bryant, D., D. Nielsen, and L. Tangley. 1997. The last frontier forests: ecosystems & economies on the edge. World Resource Institute, Washington D.C.

Busing, R. T., and P. S. White. 1993. Effects of area on old-growth forest attributes: implications for the equilibrium landscape concept. Landscape Ecology **8**:119-126.

Cabeza, M. 2003. Habitat loss and connectivity of reserve networks in probability approaches to reserve design. Ecology Letters **6**:665-672.

Cabeza, M., and A. Moilanen. 2001. Design of reserve networks and the persistence of biodiversity. Trends in Ecology and Evolution **16**:242-248.

Canadian Boreal Initiative. 2005. The boreal in the balance: securing the future of Canada's boreal region: a status report. Canadian Boreal Initiative, Ottawa, Ontario.

Cardillo, M., G. M. Mace, K. E. Jones, J. Bielby, O. R. P. Bininda-Edmonds, W. Sechrest, C. D. L. Orme, and A. Purvis. 2005. Multiple causes of high extinction risk in large mammal species. Science **309**:1239-1241.

Cardillo, M., G. M. Mace, J. L. Gittleman, and A. Purvis. 2006. Latent extinction risk and the future battlegrounds of mammal conservation. Proceedings of the National Academy of Sciences **103**:4157-4161.

Caro, T. M., and G. O'Doherty. 1999. On the use of surrogate species in conservation biology. Conservation Biology **13**:805-814.

Carroll, C., R. F. Noss, P. C. Paquet, and N. H. Schumaker. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications **13**:1773-1789.

Cerdeira, J. O., K. J. Gaston, and L. S. Pinto. 2005. Connectivity in priority area selection for conservation. Environmental Modeling and Assessment **10**:183-192.

Cizek, P., and S. Montgomery. 2005. Cumulative effects modelling of the Mackenzie gas project – scoping and development. Canadian Arctic Resources Committee, Yellowknife, Northwest Territories.

Coppolillo, P., H. Gomez, F. Maisels, and R. Wallace. 2004. Selection criteria for suites of landscape species as a basis for site-based conservation. Biological Conservation **115**:419-430.

Costello, C., and S. Polasky. 2004. Dynamic reserve site selection. Resource and Energy Economics **26**:157-174.

Cowling, R. M., R. L. Pressey, M. Rouget, and A. T. Lombard. 2003. A conservation plan for a global biodiversity hotspot - the Cape Floristic Region, South Africa. Biological Conservation **112**:191-216.

Deguise, I. E., and J. T. Kerr. 2006. Protected areas and prospects for endangered species conservation in Canada. Conservation Biology **20**:48-55.

Drechsler, M. 2005. Probabilistic approaches to scheduling reserve selection. Biological Conservation **122**:253-262.

Ferrier, S., R. L. Pressey, and T. W. Barrett. 2000. A new predictor of the irreplaceability of areas for achieving a conservation goal, its application to real-world planning, and a research agenda for further refinement. Biological Conservation **93**:303-325.

Fischer, D. T., and R. L. Church. 2005. The SITES reserve selection system: a critical review. Environmental Modeling and Assessment **10**:215-228.

Flannigan, M. D., and C. E. Van Wagner. 1991. Climate change and wildfire in Canada. Canadian Journal of Forest Research **21**: 66-72.

Fleishman, E., R. B. Blair, and D. D. Murphy. 2001. Empirical validation of a method for umbrella species selection. Ecological Applications **11**:1489-1501.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science **309**:570-574.

Fries, C., M. Carlsson, B. Dahlin, T. Lamas, and O. Sallnas. 1998. A review of conceptual landscape planning models for multiobjective forestry in Sweden. Canadian Journal of Forest Research **28**:159-167.

Gitay, H., A. Suarez, R. T. Watson, and D. J. Dokken, editors. 2002. Climate change and biodiversity. IPCC, Geneva, Switzerland.

Groves, C. R., D. B. Jensen, L. L. Valutis, K. H. Redford, M. L. Shaffer, J. M. Scott, J. V. Baumgartner, J. V. Higgins, M. W. Beck, and M. G. Anderson. 2002. Planning for biodiversity conservation: putting conservation science into practice. Bioscience **52**:499-512.

Gurd, D. B., T. D. Nudds, and D. H. Rivard. 2001. Conservation of mammals in eastern north american wildlife reserves: how small is too small? Conservation Biology **15**:1355-1363.

Gurd, D. B., and T. D. Nudds. 1999. Insular biogeography of mammals in Canadian parks: a re-analysis. Journal of Biogeography **26**:973-982.

Halpren, B., H. M. Regan, H. P. Possingham, and M. A. McCarthy. 2006. Accounting for uncertainty in marine reserve design. Ecology Letters **9**:2-11.

Harrison, S., S. Leroux, K. Lisgo, S. G. Cumming, F.K.A. Schmiegelow, and B. Olsen. 2006. The 3 Ms as the appropriate use of focal species in conservation. Biological Conservation: in review.

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, and C. Roberts. 2005. Confronting the biome crisis: global disparities of habitat loss and protection. Ecology Letters **8**:23-29.

Holroyd, P., and H. Retzer. 2005. A peak into the future: potential landscape impacts of gas development in Northern Canada. Pembina Institute, Drayton Valley, Alberta.

Howard, P. C., P. Viskanic, T. R. B. Davenport, F. W. Kigenyi, M. Baltzer, C. J. Dickson, J. S. Lwanga, R. A. Matthews, and A. Balmford. 1998. Complementarity and the use of indicator groups for reserve selection in Uganda. Nature **394**:472-475.

Johnson, C. J., M. S. Boyce, R. L. Case, H. D. Cluff, R. J. Gau, A. Gunn, and R. Mulders. 2005. Quantifying the cumulative effects of human developments: a regional environmental assessment for sensitive Arctic wildlife. Wildlife Monographs 160:1-37.

Kareiva, P. and M. Marvier. 2003. Conserving biodiversity coldspots. American Scientist **91**:344-351.

Kneeshaw, D., and S. Gauthier. 2003. Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. Environmental Reviews **11**:S99-S114.

Laliberté, A. S., and W. J. Ripple. 2004. Range contractions of North American carnivores and ungulates. Bioscience **54**:123-138.

Lambeck, R. J. 1997. Focal species: a multi-species umbrella for nature conservation. Conservation Biology **11**:849-856.

Lamoreux, J. F., J. C. Morrison, T. H. Ricketts, D. M. Olson, E. Dinerstein, M. W. McKnight, and H. H. Shugart. 2006. Global tests of biodiversity concordance and the importance of endemism. Nature **440**: 212-214.

Leslie, H., M. Ruckelshaus, I. R. Ball, S. Andelman, and H. P. Possingham. 2003. Using siting algorithms in the design of marine reserve networks. Ecological Applications **13**:S185-S198.

Mackey, B. G., M. E. Soulé, H. A. Nix, H. F. Recher, R. G. Lesslie, J. E. Williams, J. C. Z. Woinarski, R. J. Hobbs, and H. P. Possingham . 2005. Towards a scientific framework for the wildcountry project. In J. W. Wu and R. J. Hobbs, editors. Key topics and perspectives in landscape ecology. Cambridge University Press, Cambridge, UK.

Malcolm, J. R., B. D. Campbell, B. G. Kuttner, and A. Sugar. 2004. Potential indicators of the impact of forest management on wildlife habitat in northeastern Ontario: a multivariate application of wildlife habitat suitability matrices. The Forestry Chronicle **80**:91-106.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. Nature **405**: 243-253.

Margules, C. R., R. L. Pressey, and P. H. Williams. 2002. Representing biodiversity: data and procedures for identifying priority areas for conservation. Journal of Biosciences **27**:309-326.

McCullough, D. G., R. A. Werner, and D. Neumann. 1998. Fire and insects in Northern and boreal forest ecosystems of North America. Annual Review of Entomology **43**:107-127.

Meir, E., S. Andelman, and H. P. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7**:615-622.

Melillo, J. M., A. D. McGuire, D. W. Kicklighter, B. Moore III, C. J. Vorosmarty, and A. L. Schloss. 1993. Global climate change and terrestrial net primary production. Nature **363**: 234-240.

Mittermeier, R. A., C. G. Mittermeier, T. M. Brooks, J. D. Pilgrim, W. R. Konstant, G. A. B. daFonseca, and C. Kormos. 2003. Wilderness and biodiversity conservation. Proceedings of the National Academy of Science **100**:10309-10313.

Moilanen, A., and M. Cabeza. 2002. Single-species dynamic site selection. Ecological Applications **12**:913-926.

Moilanen, A. 2005a. Methods for reserve selection: interior point search. Biological Conservation **124**:485-492.

Moilanen, A. 2005b. Reserve selection using nonlinear species distribution. American Naturalist **165**:695-706.

Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. B. da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature **403**:853-858.

National Parks & Wildlife Service. 2001. C-Plan: Conservation planning software user manual for C-Plan version 3.06. NSW, Australia. Available from http:members. ozemail.com.au/~cplan/ (accessed April 2005).

Nellemann, C., and R. D. Cameron. 1998. Cumulative impacts of an evolving oil-field complex on the distribution of calving caribou. Canadian Journal of Zoology **76**:1425-1430.

Nielsen, S. E., M. S. Boyce, and G. B. Stenhouse. 2004. Grizzly bears and forestry I. Selection of clearcuts by grizzly bears in west-central Alberta, Canada. Forest Ecology and Management **199**:51-65.

Noss, R. F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the greater Yellowstone ecosystem. Conservation Biology **16**:895-909.

Noss, R. F. 1997. Protected areas: how much is enough? National parks and protected areas. Blackwell Science, Cambridge, UK.

Noss, R. F., and A. Y. Cooperrider. 1994. Saving nature's legacy. Island Press. Washington D.C.

Oetting, J. B., A. L. Knight, and G. R. Knight. 2006. Systematic reserve design as a dynamic process: F-TRAC and the Florida Forever program. Biological Conservation **128**:37-46.

Olson, D. M., and E. Dinerstein. 1998. The global 200: A representative approach to conserving the Earth's most biologically valuable ecoregions. Conservation Biology **12**:502-515.

Orme, C. D. L., R. G. Davies, M. Burgess, F. Eigenbrod, M. Pickup, V. A. Olson, A. J. Webster, T. –S. Ding, P. C. Rasmussen, R. S. Ridgely, A. J. Stattersfield, P. M. Bennett, T. M. Blackburn, K. J. Gaston, and I. P. F. Owens. 2005. Global hotspots of species richness are not congruent with endemism or threat. Nature **436**: 1016-1019.

Peters, R. S., D. M. Waller, B. Noon, S. T. A. Pickett, D. Murphy, J. Cracraft, R. Kiester, W. Kuhlmann, O. Houck, and W. J. Snape III. 1997. Standard scientific procedures for implementing ecosystem management on public lands. Pages 320-336 in S. T. A. Pickett, R. S. Ostfeld, M. Shachak, M. and G. E. Likens, editors. The ecological basis of conservation: heterogeneity, ecosystems, and biodiversity. Chapman & Hall, Toronto, Ontario.

Pickett, S. T. A., and J. N. Thompson. 1978. Patch dynamics and the design of nature reserves. Biological Conservation **13**:27-37.

Poiani, K. A., B. D. Richter, M. G. Anderson, and H. E. Richter. 2000. Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. Bioscience **50**:133-146.

Possingham, H., I. Ball, and S. Andelman. 2000. Mathematical methods for identifying representative reserve networks. Pages 291-305 in S. Ferson and M. Burgman, editors. Quantitative methods for conservation biology. Springer-Verlag, New York.

Possingham, H. P., K. A. Wilson, S. J. Andelman, and C. H. Vynne. 2006. Protected areas: goals, limitations, and design. Pages 509-551 in M. J. Groom, G. K. Meffe, and C. R. Carroll, editors. Principles of conservation biology. 3rd ed. Sinauer Associates, Inc., Sunderland, Massachusetts.

Pressey, R. L. 2004. Conservation planning and biodiversity: assembling the best data for the job. Conservation Biology **18**:1677-1681.

Pressey, R. L., R. M. Cowling, and M. Rouget. 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Florisitic Region, South Africa. Biological Conservation **112**:99-127.

Pressey, R. L., C. J. Humphries, C. R. Margules, R. I. Vanewright, and P. H. Williams. 1993. Beyond opportunism - key principles for systematic reserve selection. Trends in Ecology & Evolution **8**:124-128.

Pressey, R. L., and K. H. Taffs. 2001. Scheduling conservation action in production landscapes: Priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss. Biological Conservation **100**:355-376.

Pressey, R. L., and S. L. Tully. 1994. The cost of *ad hoc* reservation: a case study in western New South Wales. Australian Journal of Ecology **19**:375-384.

Pyke, C. R., and D. T. Fischer. 2005. Selection of bioclimatically representative biological reserve systems under climate change. Biological Conservation **121**:429-441.

Roberge, J. -M., and P. Angelstam. 2004. Usefulness of the umbrella species concept as a conservation tool. Conservation Biology **18**:76-85.

Rodriguez, A. S. L., H. R. Akçakaya, S. J. Andelman, M. I. Bakarr, L. Boitani, T. M. Brooks, J. S. Chanson, L. D. C. Fishpool, G. A. B. Da Fonseca, K. J. Gaston, M. Hoffmann, P. A. Marquet, J. D. Pilgrim, R. L. Pressey, J. Schipper, W. Sechrest, S. N. Stuart, L. G. Underhill, R. W. Waller, M. E. J. Watts, and X. Yan. 2004. Global gap analysis: priority regions for expanding the global protected-area network. Bioscience **54**:1092-1100.

Rubino, M. J., and G. R. Hess. 2003. Planning open spaces for wildlife 2: modeling and verifying focal species habitat. Landscape and Urban Planning **64**:89-104.

Sala, O. E., F. S. Chapin III, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. Oesterheld, N. L. Poff, M. T. Sykes, B. H. Walker, M. Walker, and D. H. Wall. 2000. Global biodiversity scenarios for the year 2100. Science **287**:1770-1774.

Sanderson, E. W., M. Jaiteh, M. A. LEvy, K. H. Redford, A. V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. BioScience **52**:891-904.

Schneider, R. R., J. B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the western canadian sedimentary basin: a modeling approach. Conservation Ecology **7**:8.

Shugart, H. H., and D. C. West. 1981. Long-term dynamics of forest ecosystems. American Scientist **69**:647-652.

Stocks, B. J., M. A. Fosberg, T. J. Lynham, L. Mearns, B. M. Wotton, Q. Yang, J. Z. Jin, K. Lawrence, G. R. Hartley, J. A. Mason, and D. W. McKenney. 1998. Climate change and forest fire potential in Russian and Canadian boreal forests. Climate Change **38**: 1-13.

Suter, W., R. F. Graf, and R. Hess. 2002. Capercaillie (*Tetrao urogallus*) and Avian Biodiversity: testing the umbrella-species concept. Conservation Biology **16**:778-788.

Turner, M. G., and W. H. Romme. 1994. Landscape dynamics in crown fire ecosystems. Landscape Ecology **9**:59-77.

Venevsky, S., and I. Venevskaia. 2005. Hierarchical systematic conservation planning at the national level: identifying national biodiversity hotspots using abiotic factors in Russia. Biological Conservation **124**:235-251.

Warman, L. D., D. M. Forsyth, A. R. E. Sinclair, K. Freemark, H. D. Moore, T. W. Barrett, R. L. Pressey, and D. White. 2004. Species distributions , surrogacy, and important conservation regions in Canada. Ecology Letters **7**:374-379.

Wessels, K. J., S. Freitag, and A. S. van Jaarsveld. 1999. The use of land facets as biodiversity surrogates during reserve selection at a local scale. Biological Conservation **89**:21-38.

Wiersma, Y. F., T. D. Nudds, and D. H. Rivard. 2004. Models to distinguish effects of landscape patterns and human population pressures associated with species loss in Canadian national parks. Landscape Ecology **19**:773-786.

Wiersma, Y. F., and D. L. Urban. 2005. Beta diversity and nature reserve system design in the Yukon, Canada. Conservation Biology **19**:1262-1272.

Williams, J. C., C. S. ReVelle, and S. A. Levin. 2004. Using mathematical optimization models to design nature reserves. Frontiers in Ecology and the Environment **2**:98-105.

Williams, J. C., C. S. ReVelle, and S. A. Levin. 2005. Spatial attributes and reserve design models: a review. Environmental Modeling and Assessment **10**:163-181.

Wilson, K. A., M. F. McBride, M. Bode, and H. P. Possingham. 2006. Prioritizing global conservation efforts. Nature **440**: 337-340.

Chapter 2

Incorporating system dynamics in reserve design[1]

1. Introduction

Systematic conservation planning methods (*sensu* Margules & Pressey 2000) were developed in response to shortcomings in historical methods of reserve design that were largely ad hoc. In contrast, systematic conservation planning of reserves uses an explicit and structured approach (Margules & Pressey 2000). While conservation projects around the world have adopted systematic conservation planning, most applications have occurred in substantially altered systems, such as the New South Wales region in Australia (Pressey *et al.* 1993), the Cape Florisitic region in South Africa (Pressey *et al.* 2003), and California State in the United States (Pyke & Fischer 2005).

The conventional approach to systematic conservation planning uses site selection algorithms (Possingham *et al.* 2000) to identify collections of discrete sites, which in aggregate, satisfy a priori conservation targets for focal species habitat, vegetation communities, waterbodies, and special elements (Cabeza & Moilanen 2001; Noss *et al.* 2002; Warman *et al.* 2004). I refer to such collections as reserve networks. Sites with the greatest irreplaceability contribute most to the conservation targets for the features contained (Ferrier *et al.* 2000) and are assigned the highest conservation value (Pressey *et al.* 1993; Margules & Pressey 2000). Potential reserves (*i.e.,* sites of high irreplaceability) are ranked in priority according to their vulnerability (*i.e.,* status of threats) (Margules & Pressey 2000; Noss *et al.* 2002) or according to sequential reserve scheduling approaches (Meir *et al.* 2004; Wilson *et al.* 2006), as it is unlikely that all potential reserves will be established immediately due to limited funds or jurisdictional barriers. While rapid advances in techniques have been made over the past decade, conventional applications of systematic planning have been static, with little consideration of the dynamic nature of ecological systems (Cabeza & Moilanen 2001; Moilanen & Cabeza 2002) or the uncertainty inherent in these systems (Halpern *et al.* 2006). Most conservation plans also

---

[1] This chapter is formatted for submission to *Ecology Letters* and is currently under consideration at *Ecology Letters* by Leroux, S.J., Schmiegelow, F.K.A., Cumming, S.G., Lessard, R.B., Nagy, J. Reprinted with permission from *Blackwell Publishing.*

assume that the spatial distribution of conservation features is well known and constant (Meir *et al.* 2004; Pyke & Fischer 2005).

While recognizing the strengths of systematic conservation planning methods, several authors have identified the need to incorporate system dynamics into systematic conservation plans (Cumming *et al.* 1996; Margules & Pressey 2000; Bengtsson *et al.* 2003; Meir *et al.* 2004; Halpern *et al.* 2006). Large-scale system dynamics (*e.g.*, natural disturbance, climate change) influence vegetation communities and their dependent fauna (Margules and Pressey 2000). Consequently, many features that we attempt to capture in reserve networks are inherently dynamic. Reserve designs that incorporate static targets derived from existing conditions may not convey long-term protection (Bengtsson *et al.* 2003). Accordingly, methods are being developed to incorporate models of species persistence (*e.g.*, Moilanen and Cabeza 2002), landscape change (*e.g.*, Pressey *et al.* 2004; Pyke & Fischer 2005), and uncertainty (*e.g.*, Halpern *et al.* 2006) in conservation plans. However, no methods currently exist for evaluating the efficacy of reserve networks under the influence of large ecological processes. Explicit incorporation of large ecological processes in reserve design is particularly relevant for the world's remaining wilderness areas, such as the boreal and Amazonian regions, which are still structured by large natural processes such as forest fire and flooding regimes, and where opportunities to plan for future landscape change, such as that associated with climate change, remain. Spatially explicit models of system dynamics can be useful tools to assess reserve network efficacy: the ability of reserve networks to maintain their initial conservation targets through time.

I provide a case study of conservation planning in a 66,000-km^2 study region of the Mackenzie Valley, Northwest Territories (NT), Canada (Fig. 2-1), that includes portions of the traditional lands of the Gwich'in First Nation. The anthropogenic footprint on this landscape is low compared to areas in the southern boreal forest, and much lower than most other regions of the world. This region has been identified as a hotspot of latent extinction risk (Cardillo *et al.* 2006). I evaluate the efficacy of reserve networks designed based on systematic conservation planning methods under natural disturbance dynamics, as a first step in addressing future landscape change, and demonstrate the utility of spatially explicit, dynamic simulation models in reserve design.

I use conventional reserve design tools to develop alternate reserve networks with different conservation targets for woodland caribou (*Rangifer tarandus caribou* Gmelin) habitat, high-quality wetlands, vegetation types, waterbodies, and reserve connectedness. I also include fixed targets for Environmentally Significant Areas (ESAs) and heritage sites. I input these reserve networks into CONSERV, an empirically-based, spatially explicit, dynamic simulation model of vegetation community dynamics and natural disturbance, parameterized for the study region, and use simulation experiments to evaluate the efficacy of reserve networks designed with alternate criteria. This study is the first I know of to apply natural disturbance models to reserve design.

2. Methods

2.1 Data

A 30-m resolution, 34 category earth cover map of the study region (Ducks Unlimited 2002, 2003) was reclassified to 10 cover types and re-scaled to 500-m resolution by a majority threshold filter. The 500-m resolution was sufficient for caribou habitat models (below) and minimized the aggregation and loss of unique earth cover types. The earth cover types used include distinct tree, shrub and grass communities, wetlands, and permanent water. I focused on protecting high-density wetlands in the study region because two duck species that breed in this area, Scaup sp. (*Aythya sp.* L. and Eyton) and Scoter sp. (*Melanitta sp.* L.), have experienced significant declines across their range (Decarie 1995; Austin *et al.* 2000) and high-density wetlands are the most productive duck habitat (Johnson & Grier 1988). I used the original 30-m resolution map to calculate per-cell wetland edge density (km/km^2) for the re-scaled map. Cells with densities above the regional median (3.77 km/km^2) were considered high quality wetlands.

I calculated per-cell slope, aspect (flat, N, S, E, W), and elevation from a digital elevation model (Natural Resources Canada 2000). Seismic lines are semi-permanent linear features created by exploration for oil and natural gas. I estimated per-cell seismic line density (km/km^2) from the National Energy Board (2001). I obtained wildfire sizes and locations for the interval 1965-2004 from fire management agency archives (Government of the Northwest Territories 2005; Government of Yukon 2005).

Woodland caribou occurrence data were obtained from GPS and ARGOS satellite collars on 8 female woodland caribou tracked from 1 May 2002 – 13 October 2005 (n = 10,559 locations). Mapped special elements included wildlife areas of special interest (Ferguson 1987), key migratory bird terrestrial habitat sites (Alexander *et al.* 1991), territorial parks, and heritage sites data representing Gwich'in trails and camps (Gwich'in Land Use Planning Board 2003).

2.2 Criteria for 6 reserve networks

I developed 6 reserve networks for the study region based on conventional systematic conservation planning methods, incorporating both species and landscape feature data to exploit the advantages of each. The 6 reserve networks were designed using a site-selection tool that combined targets for 1) suitable woodland caribou habitat, 2) representation of high-quality wetlands, 3) representation of vegetation types and waterbodies, 4) inclusion of special elements, and 5) reserve connectedness.

2.2.1 Modelling suitable woodland caribou habitat

I chose woodland caribou as a focal species in the analyses because they are medium-sized ungulates with large home ranges, distributed across the boreal forest and are listed as threatened in Canada (COSEWIC 2003). There were insufficient data to parameterize population viability models, so I developed the conservation plan to capture targets for suitable caribou habitat based on empirical species occurrence models. A core caribou modelling region was defined by a minimum convex polygon around GPS and ARGOS satellite caribou locations buffered by an additional 3,176 m; the 95^{th} percentile of all distances travelled by radio-tagged caribou in 8 hours (Nagy *et al.* 2006). I modelled caribou habitat use in the winter season (15 January to 30 April) because winter is likely the limiting season for woodland caribou in the study region (Nagy *et al.* 2006). I generated random points at a density of $2/km^2$ within the buffered core area to sample available habitat (n = 60,385).

I developed the caribou habitat model by pooling data for all caribou and using multiple logistic regression to develop a landscape-level resource selection function (RSF) (Johnson *et al.* 2004) based on a suite of earth cover and terrain covariates (Table 2-1). The caribou use and random points were overlaid on the 500-m grid, and habitat variables were assigned to locations.

I used a linear stretch to scale the predicted values of the RSF, w, between 0 and 1 (Johnson *et al.* 2004). The scaled values for the sampled available points and used points were sorted in ascending order, partitioned into 10 equal size bins, and assigned a rank between 1 (lowest decile of w-values) and 10 (highest decile of w-values). I tested for significant habitat selection by a χ^2 test (d.f. = 9) (Nagy *et al.* 2006). Following Nielsen *et al.* (2006), I binned the predicted RSF values for the entire region and considered RSF values larger than the median value to be suitable habitat for caribou.

2.2.2 Reserve connectedness

Connectedness, determined by physical linkages between sites within a reserve network, is a key consideration in reserve design (McDonnell *et al.* 2002; Warman *et al.* 2004). In site selection tools like MARXAN (Ball & Possingham 2000), the user can vary reserve network connectedness by changing the boundary length modifier (Possingham *et al.* 2000; McDonnell *et al.* 2002). This parameter acts as a penalty on the total edge length of selected sites, and higher values tend to result in more contiguous networks.

2.3 Conservation targets

I developed 6 reserve networks with a range of conservation targets. I evaluated conservation targets of 25 %, 50 %, and 75 % protection for the map cells with suitable habitat for caribou and high-quality wetlands. I set representation targets for each re-scaled earth cover type in the study region (see previous description) of 10 %, 20 %, and 30 %. I set targets of maintaining 75 % of wildlife areas of special interest and key migratory bird terrestrial habitat sites and 100 % of territorial parks and key heritage sites in reserves. Finally, I designed three reserve networks with low connectedness and three reserve networks with high connectedness (Table 2-2).

2.4 Reserve network construction

Planning units provide the framework for constructing a conservation plan, but there is no strong theoretical basis for selecting the size and shape of planning units (Pressey & Logan 1998). I used planning units of uniform size and geometry (2 x 2 km; n = 16,454). With planning units of this resolution, I was able to run the site selection tool on the entire study region. Each planning unit contained 16 map cells.

I combined goals for the focal species, high-quality wetlands, vegetation types, waterbodies, special elements, and connectedness into MARXAN (Ball & Possingham

2000) (Fig. 2-2). I used the CLUZ user-friendly interface for MARXAN to build the reserve networks (Smith 2004). MARXAN is a site selection tool that facilitates the identification and selection of candidate reserves designed to satisfy a suite of conservation targets. MARXAN uses a global heuristic algorithm, in this case simulated annealing, to approximate the minimum number of sites or total site area to achieve stated targets (*i.e.*, minimum area problem; Cabeza & Moilanen 2001). A simulated annealing algorithm begins by generating a random set of sites. Then, at each iteration, a site is removed or added to the random set and the value of the new set is compared to the initial one (Possingham *et al.* 2000; Cabeza & Moilanen 2001). After many iterations, simulated annealing algorithms can identify near-optimal solutions to a minimum area problem. I developed 6 different reserve networks based on the minimum, medium, and maximum percent targets for focal species habitat, high-quality wetlands, vegetation types, and waterbodies, and the fixed special elements targets (Table 2-2). For each reserve network, I performed 100 MARXAN runs of 1,000,000 iterations. When developing reserve networks for multiple targets, over-representation of some targets is commonly encountered due to the overlap of resources on the landscape. I refer to the targets I designed the reserve networks to achieve as initial targets (T_i) and I refer to the amount of each feature actually captured in the reserve networks as the realized target (T_a).

2.5 CONSERV

To evaluate reserve network efficacy, I developed CONSERV, a simple grid-based spatial model of vegetation class dynamics and fire. The fire model and user interface of CONSERV are based on an earlier model developed for boreal landscapes (Armstrong and Cumming 2003). I model forest fire in CONSERV because fire is the dominant disturbance that determines vegetation community development in the boreal forest (Johnson 1992).

2.5.1 Vegetation dynamics

CONSERV simulates vegetation dynamics using age-based state transition rules which summarize key features of the study region ecology. There are five main vegetation types: spruce forest, mixed forest, low shrub, tall shrub, and herbaceous. With respect to forested areas, Black & Bliss (1978) described 4 post-fire seral stages for spruce forests in the study region. Stage 1 occurs 3-20 yrs after a burn and is characterized by low shrubs; stage 2 occurs 21- 35 yrs after a burn and is characterized by tall shrub; stage 3 occurs 36-200 yrs after fire and is dominated by open spruce forest, and stage 4 occurs more than 200 yrs after fire and is dominated by a closed spruce forest. Black & Bliss (1978) did not study succession in mixed-wood forests of the study region. I assumed that burned mixed-wood forest follow the same low and tall shrub stages as do spruce forest, but regenerate to the mixed-wood state after 36 years. I also assumed herbaceous patches regenerate 3 yrs post-fire. There is some evidence that closed-canopy spruce forests may senesce to a low-shrub-like state in the prolonged absence of fire (Strang & Johnson 1981). I assume that all forested types revert to low shrub vegetation 300 yrs post-fire, as if burned.

2.5.2 Forest fire

CONSERV models forest fire as a three-stage stochastic process where fires ignite, escape, and spread (Armstrong and Cumming 2003). The outcome of each fire is determined by a random draw from a probability distribution. I used empirical forest fire history data to estimate the distribution parameters for the study region, as described by Armstrong and Cumming (2003). Recent fire frequency data (1987-2004) revealed 357 ignitions across 51,200 km^2 of forest that was at risk of burning; therefore, the mean number of fires/yr/25-ha cell was 1.03 x 10^{-4} which closely approximates the annual per-cell ignition probability. After ignition, fire spread is modelled as a modified percolation process. Only 104/357 fires (29 %) "escaped" or exceeded the 25 ha cell size. Following Armstrong and Cumming (2003), I estimated the initial spread probability to each of 8 adjacent cells to be 0.04.

I calibrated the spread probability of escaped fires by simulation experiments finding the value that produced approximately the mean size of fires and frequency of occurrence of large fires (*i.e.*, > 25 ha) in the full empirical dataset (1965-2004). In the model, fire spreads until it reaches the maximum allowable size or until no further cells are burned. The best solution produced a mean fire size of 9,331 ha (n = 4468). The mean fire size of empirical data was 9,291 ha (n = 184). I compared the simulated mean to the mean fire size of the empirical data by bootstrapping 200 samples from each dataset, 100 times, and performing a Kolmogrov-Smirnov test on each bootstrap run. The mean p-value of these 100 tests was 0.424, and only 9/100 tests has $p < 0.05$. I conclude that the calibrated parameters result in a fire size distribution comparable to the empirical data. The maximum allowable fire size was 224,000 ha, calculated from historical records by methods of Cumming (2001)

2.5.3 Model initialization

CONSERV is initialized with a ASCII file containing a number of map layers including the classified vegetation map, presence/absence of high quality wetlands, seismic line density, the other constant biophysical parameters needed to apply the predictive caribou habitat model, and a reserve network as constructed by MARXAN. Because vegetation dynamics are age dependent, an initial age map is also required. There are limited data on forest ages in the study region; however, detailed fire history records dating back to 1987 were available, as was an earth cover classification map (Ducks Unlimited 2002, 2003). I used time since the last stand-initiating fire as the age of forest patches that recently burned. For forest patches without a fire history, I randomly placed polygons the size of actual fires in the study region, and assigned an age to each polygon by randomly drawing ages from a negative exponential age-class distribution (Van Wagner 1978) with a fire cycle of 100 yrs, which is characteristic of this region of the boreal forest (Johnson 1992).

2.6 CONSERV – Reserve network evaluation

I input the 6 reserve networks into CONSERV and tracked how well each reserve network maintained the initial conservation targets under vegetation dynamics and disturbance (Fig. 2-2). For each annual time step, CONSERV tracked the proportion of suitable habitat for woodland caribou in the reserve network by calculating an RSF value for each map cell using the coefficients of the RSF model (Table 2-1). The proportion of each vegetation type in the reserve network was also computed at each time step. CONSERV did not track high-quality wetlands, water, or special element targets as these were considered static features. I performed 100 simulations of 250 yrs for each of the 6 alternate reserve networks. I reported the percentage of simulation years that each target was lower than T_i and the percentage of simulation years that each target was higher than T_a. I use T_a to remove the effects of the surplus representation that occurred during initial site selection and interpret the effects of system dynamics on reserve efficacy.

3. Results

3.1 Woodland caribou RSF model

The distribution of caribou use and random locations among RSF bins was significantly different than random ($p < 0.001$). The RSF model revealed significant caribou habitat selection relevant to reserve design in dynamic ecosystems. Greater than expected numbers of use locations occurred in the upper 5 bins, with the majority of these occurring in the upper 2 bins, indicating that the model had good predictive ability. Caribou selected areas of open spruce, recent burns, near water, and south facing aspects, and avoided areas with herbaceous vegetation cover, steep slopes, and high densities of seismic lines (Table 2-1). The median RSF value was 0.09. Accordingly, there was 33,000 km^2 of suitable habitat for woodland caribou in the study region at model initialization.

3.2 Static reserve networks

The three reserve networks with low connectedness required 73 - 140 reserves covering 21.1 - 55 % of the landscape to meet their targets, whereas the three reserve networks with high connectedness required 11 - 22 reserves covering 30.6 - 63.3 % of the landscape. All reserve networks, regardless of the scenario, were concentrated in the north-east section of the study region (Fig. 2-3). By construction, the T_i were met in all reserve networks. However, given the multiple criteria applied, the T_i were over-represented for 91 % of the individual targets in the reserve networks.

3.3 CONSERV evaluation results

3.3.1 Woodland caribou targets

In the reserve networks with low connectedness, caribou habitat targets were not satisfied in 15 - 92 % of simulation years, while in reserve networks with high connectedness, caribou habitat targets were not satisfied in 0 – 88 % of simulations years. Reserve networks with high initial targets for woodland caribou habitat (*i.e.*, 75 %) had more years where their targets were not maintained. All reserve networks had 9 – 17 % of years where the amount of suitable woodland caribou habitat was higher than T_a (Fig. 2-4).

3.3.2 Vegetation targets

Success in maintaining the vegetation targets varied, due in part to a unit-sum constraint, where an increase in one vegetation type necessarily equated to a decrease in another. The T_i for closed spruce and open spruce were met for all reserve networks, except the minimum targets network for closed spruce. Tall shrub and low shrub T_i were maintained in some reserve networks but their T_i were not satisfied in the minimum targets, minimum targets connected, medium targets, and maximum targets networks for 0.02 – 7 % of simulation years. Less connected reserve networks had more years where the extent of mixed (0.5 – 17 %), herbaceous (9 – 14 %), and burn (19 – 50 %) was lower than T_i than more connected reserve networks. All reserve networks exceeded the T_a for each vegetation target throughout the simulations. For 1 – 22 % of years, the extent of tall shrub and low shrub in reserve networks was higher than T_a whereas for 4 – 99 % of years, the extent of closed spruce, open spruce, mixed, low shrub, and herbaceous in reserve networks exceeded T_a (Fig. 2-4). The amounts of wetland and water were invariant as they are unaffected by CONSERV's patch dynamics and disturbance rules.

4. Discussion

I developed a spatially explicit dynamic simulation model, CONSERV, to explore the efficacy of reserve networks designed using systematic conservation planning methods under an active natural disturbance regime; in this case, fire. Other studies have incorporated focal species population dynamics (Carroll *et al.* 2003), landscape predictions (Pressey *et al.* 2004; Pyke & Fischer 2005), system uncertainty (Halpern *et al.* 2006), and dynamic implementation (Meir *et al.* 2004; Wilson *et al.* 2006) into reserve design, but I am aware of no other studies that explicitly evaluate the efficacy of reserve networks in dynamic landscapes.

I initially constructed 6 reserve networks with varying targets for focal species, high-quality wetlands, vegetation types, and waterbodies, fixed targets for ESAs and key heritage sites, and two measures of reserve connectedness. Not surprisingly, a greater number of reserves and more area was required to meet higher conservation targets. More connected networks required fewer reserves but more total area to meet the targets at all levels. Simultaneous achievement of multiple targets resulted in over-representation of initial goals for most targets in the 6 reserve networks. Simulation experiments revealed that most reserve networks failed to maintain the initial targets for woodland caribou through time, and reserve networks with low connectedness fared poorly compared to reserve networks with high connectedness at maintaining caribou habitat. The greater efficacy of highly-connected reserve networks may have resulted from the greater area encompassed by the more connected reserve networks, rather than the added benefit of higher physical connectedness between reserves per se. Future analyses could evaluate this by keeping the area encompassed by the different reserve networks constant and varying only the degree of connectedness. Most reserve networks maintained their vegetation targets through time. The minimum-targets network was the least effective with only the extent of open spruce and herbaceous vegetation consistently higher than their T_i. The maximum-targets connected network was the most effective, with only the extent of burn vegetation experiencing years below its T_i. This was not unexpected: as one increases the T_i, more of the landscape is in reserve, effectively capturing a higher proportion of the targets.

By using CONSERV to evaluate different reserve networks, I was able to determine which targets were unlikely to be maintained in reserves under natural system dynamics. If a decrease in the targets over time is unacceptable, conservation planners may consider increasing initial levels for some targets to buffer against the effects of natural dynamics within a static reserve system. This buffering may be particularly useful for maintaining areas for endangered species with specific habitat requirements. Simulation models enable conservation planners to evaluate options and set precautionary targets that will maximize the likelihood of maintaining conservation objectives through time.

In all reserve networks, some targets had years where the extent of each target was larger than the realized targets for some period of time during the 250 year simulation period, indicating over-representation of targets through time. If conservation planners are primarily concerned with protection, this is a positive result, but it can also be viewed as inefficient, given limits on the amount of land and water that can be set aside for protection. Methods for determining cost-effective and efficient designs for reserve networks have been developed elsewhere (Wilson *et al.* 2006), but the quantification of over-representation of targets under a natural disturbance regime also permits conservation planners to account for inefficiencies in a reserve network and improve cost-effectiveness.

Simulation models can be used iteratively to set targets in an uncertain and dynamic world and can be used to estimate the size and configuration of reserves required to capture biodiversity and ecological processes in biomes that are at risk (Hoekstra *et al.* 2005; Cardillo *et al.* 2006). Many conservation plans incorporate seemingly arbitrary targets for focal species, vegetation, waterbodies, and special elements because they lack better data or methods to set ecologically meaningful targets. Futhermore, features targeted in conservation plans are often used as surrogates for other elements that are not directly measured. Simulation models can be used to test assumptions regarding surrogates and target levels, thereby contributing to better conservation plans based on robust and efficient targets that have a higher likelihood of being maintained under natural system dynamics.

Despite the prevalence of simulation models in applied ecology, they have not previously been used to evaluate reserve networks (Pressey & Tafts 2001). The case study demonstrates that spatially explicit, dynamic simulation models can be valuable tools in the evaluation of reserve designs and facilitate pro-active planning. In the Mackenzie Valley, the design of reserve networks to effectively maintain conservation targets is critical because degradation of the surrounding landscape is anticipated due to resource development activities and this region has been identified as a hotspot of latent mammal extinction risk (Cardillo *et al.* 2006). The model allows for the evaluation of a suite of conservation scenarios before final reserve designs are implemented. In the study region, final decisions are made by the land-use planning boards representing the Gwich'in First Nations, and will incorporate a broader range of objectives than I considered. However, spatially explicit, dynamic simulation models can be valuable tools supporting the decision-making process.

4.1 Limitations and implications for other models

Data for natural systems are required to parameterize spatially explicit simulation models. These data are available for forest dynamics in the boreal forest but may not be available for other regions of the world. However, even without sufficient data, one can evaluate a range of parameters using different modelling frameworks (Halpren *et al.*

2006). CONSERV is limited in that it only models two processes; vegetation community dynamics and forest fire. In reality, there are many interacting processes that affect the ability of reserve networks to maintain their targets through time. Future models could be improved by taking into account additional processes and interactions that may affect reserve efficacy. The simple vegetation community dynamic and focal species sub-models I used could also be improved with additional data and more advanced modelling techniques (Guisan & Thuiller 2005; Halpern *et al.* 2006). Future work should also consider the effect of landscape condition outside reserves on the efficacy of reserve networks and address reserve connectivity - the ability of reserves to maintain the flows of energy, materials, and organisms – rather than simply physical connectedness. Whereas I used the model to retrospectively evaluate candidate reserve designs, similar models could be used prospectively to determine the size and configuration of reserves required for the long-term maintenance of conservation targets. Similarly, spatially explicit models could be used to evaluate the gaps in coverage of processes in reserve networks, something currently lacking in systematic conservation planning (Margules and Pressey 2000).

I evaluated the relative efficacy of different static reserve networks to maintain conservation features over time. However, dynamic or floating reserves may be useful for maintaining conservation features in dynamic systems (Cumming *et al.* 1996; Bengtsson *et al.* 2003), in combination with static reserves and sustainable resource management. Spatially explicit, dynamic simulation models could be used to evaluate the potential contribution of static and dynamic reserves to protection of biodiversity and maintenance of ecological processes in dynamic landscapes. Existing sequential decision-making processes (Meir *et al.* 2004; Wilson *et al.* 2006) could be used to schedule the reserve rotation. At the very least, it would be worthwhile to explore the management of dynamic reserves in a simulation environment.

4.2 Implications for systematic conservation planning

Margules and Pressey (2000) proposed 6 stages of systematic conservation planning which have guided many reserve planning exercises: 1) compile data, 2) identify conservation goals, 3) review existing conservation areas, 4) select additional conservation areas, 5) implement conservation actions, and 6) maintain required values of conservation areas. This study, and others that incorporate elements of system dynamics in reserve design (*e.g.*, Moilanen & Cabeza 2002; Pressey *et al.* 2004; Pyke & Fischer 2005; Halpren *et al.* 2006), support enhancement of the fourth stage to *evaluate* additional conservation areas. At this stage, spatially-explicit, dynamic simulation models could be used to refine conservation targets and candidate reserves selected in earlier stages of the planning process. Natural resource management models such as LANDIS (Mladenoff 2004), SELES (Fall & Fall 2001), and TARDIS (Cumming & Armstrong 2001), are used to quantify landscape dynamics and evaluate resource management policy, but few conservation plans make use of such tools. I suggest that application of such models should be an integral part of conservation planning because they can contribute to a better understanding of the potential efficacy of reserve networks prior to implementation, and they can be used for developing robust conservation targets that incorporate environmental change. In this manner, enhancement of stage 4 provides direct links to stages 5 and 6, through exploration of implementation options and providing guidance for monitoring within an adaptive management framework (Walters 1986).

Future research on reserve design should incorporate models of population dynamics, climate change, uncertainty and natural disturbance, and dynamic implementation, into a common framework to improve theory and inform the design and evaluation of effective reserve networks. Spatially explicit, dynamic simulation models such as CONSERV can form the framework for this integration.

Figure 2-1 Study region in the northern boreal region of Canada shown as black polygon. The study region is 66,000 km^2, located mostly in the Northwest Territories (NT) with parts in the Yukon Territory (YT).

Table 2-1 Coefficients and test statistics of the RSF model for woodland caribou selection of habitat variables during winter (15 January to 30 April) in the study region.

Variable	Coefficient	Standard Error	P>z
Slope (%)	-0.2646	0.0162	0.0000
Aspect flat	0		
Aspect north	0.8722	0.2025	0.0000
Aspect east	0.9811	0.2022	0.0000
Aspect south	1.4474	0.2013	0.0000
Aspect west	1.0633	0.2012	0.0000
Median elevation (m)	-0.0005	0.0002	0.0100
Seismic lines (km/km^2)	-0.1640	0.0227	0.0000
Closed spruce	0		
Open spruce	2.3836	0.3804	0.0000
Mixed	0.6512	0.4652	0.1620
Tall shrub	0		
Low Shrub	0.5039	0.3906	0.1970
Herbaceous	-0.5089	0.8050	0.5270
Burn	1.5721	0.3906	0.0000
Wetland	0		
Water	1.5135	0.3864	0.0000
Other	0.7149	0.4187	0.0880
Constant	-5.5883	0.4256	0.0000

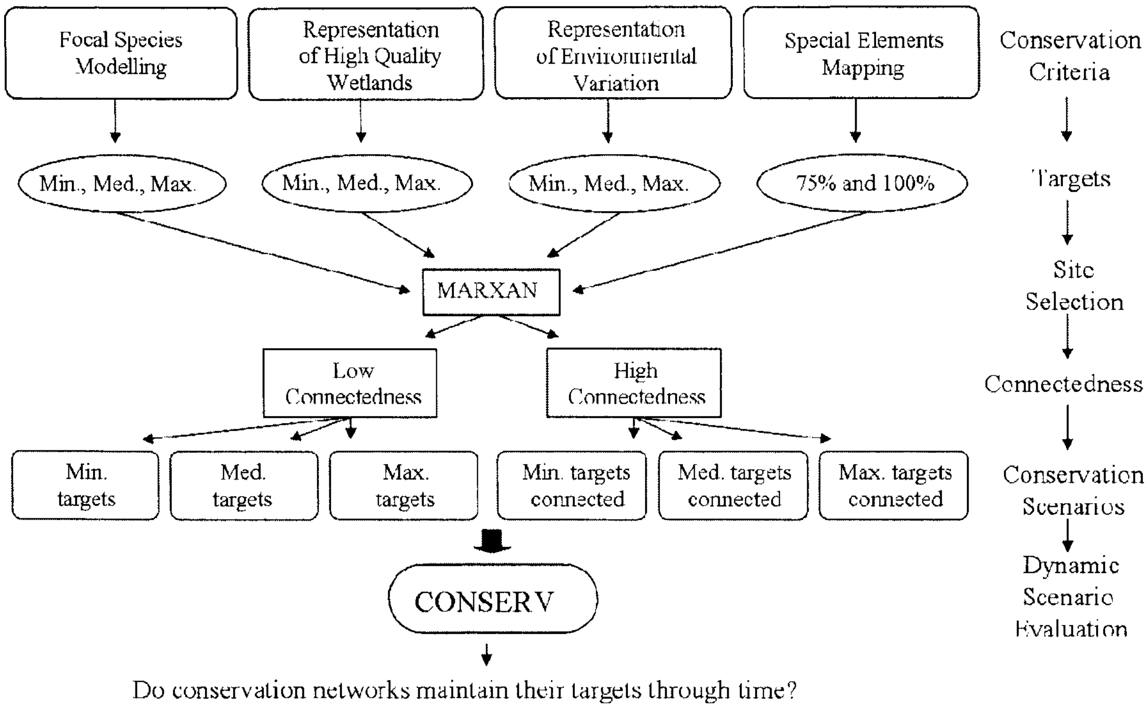

Figure 2-2 Flow-chart describing the methods used in this study.

Table 2-2 Six sets of conservation targets used in developing the 6 reserve networks based on conventional systematic conservation planning methods. I used a range of targets for focal species, high-quality wetlands, environmental representation and two measures of connectedness to evaluate a range of conservation options. The special elements targets are fixed at 100 % of heritage sites and existing territorial parks and 75 % of key migratory bird sites and wilderness areas. The reserve networks range from minimum-targets to maximum-targets connected.

Scenario	Focal Species & Wetland (%)	Vegetation (%)	Connectedness (%)	Special Elements (%)
Minimum-targets	25	10	Low	100 & 75
Minimum-targets connected	25	10	High	100 & 75
Medium-targets	50	20	Low	100 & 75
Medium-targets connected	50	20	High	100 & 75
Maximum-targets	75	30	Low	100 & 75
Maximum-targets connected	75	30	High	100 & 75

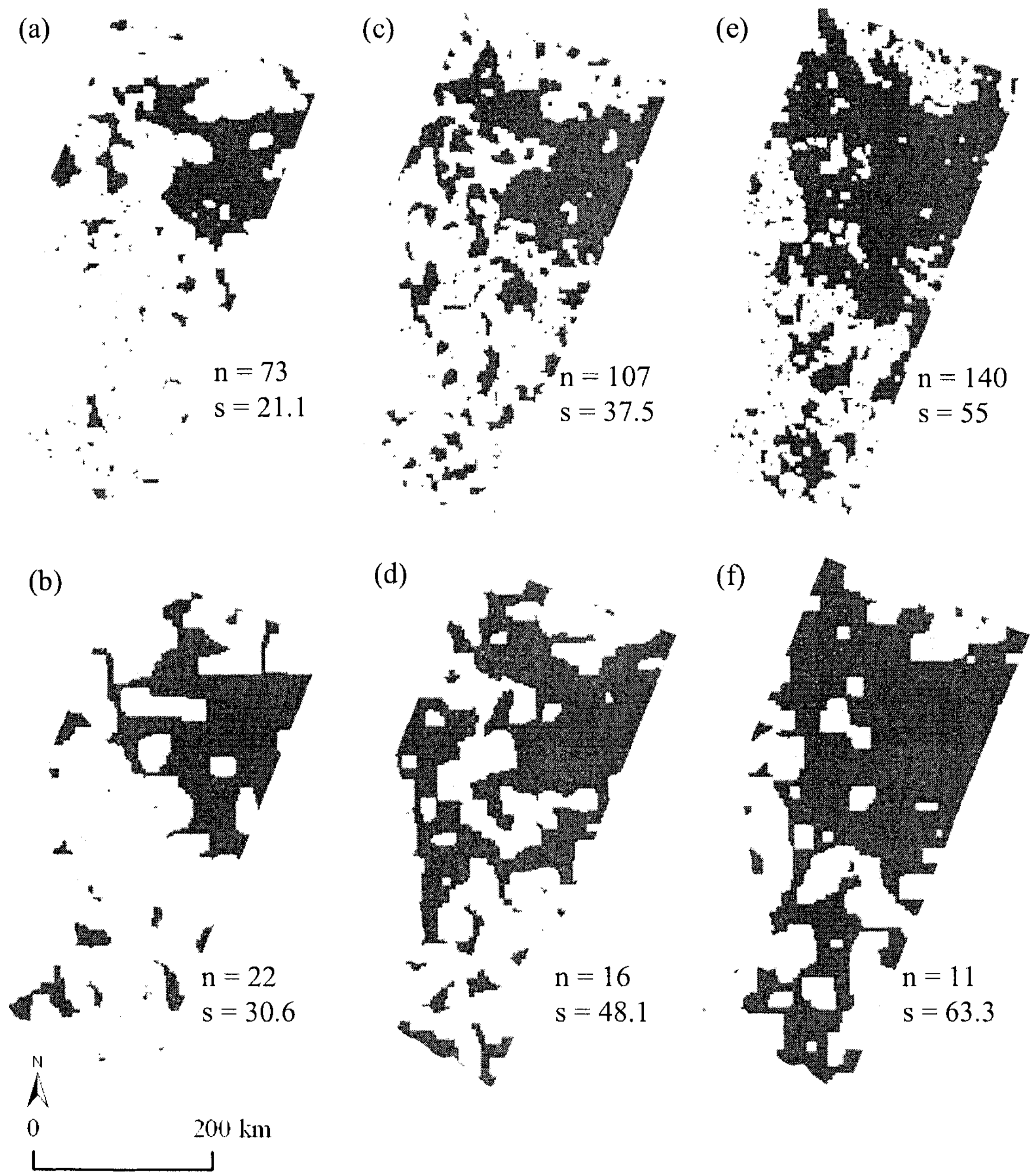

Figure 2-3 Hypothetical reserve networks designed to capture suitable habitat for woodland caribou, high quality wetlands, vegetation types, waterbodies, wildlife areas of special interest, key migratory bird terrestrial habitat sites, territorial parks, and key heritage sites. The number of reserves (n) and percent of the landscape in reserve (s) of the 6 reserve networks developed with varying targets using MARXAN: (a) minimum-targets, (b) minimum-targets connected, (c) medium-targets, (d) medium-targets connected, (e) maximum-targets, and (f) maximum-targets connected. Dark areas are reserve networks and light areas are non-reserve areas.

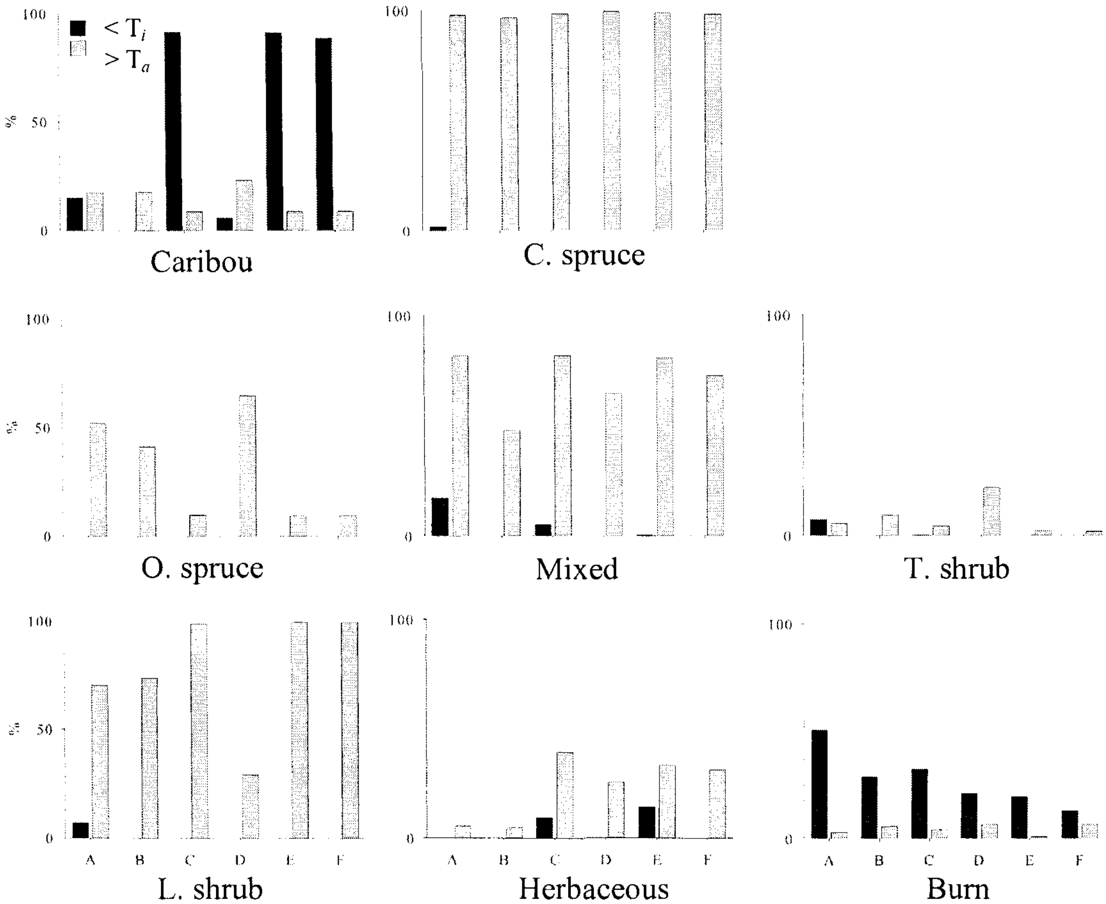

Figure 2-4 Percent of simulation years that each target was lower than T_i (< T_i) and the percent of simulation years that each target was higher than T_a (> T_a) throughout CONSERV's 100 simulations of 250 years. Results are shown for the 6 reserve networks: A (minimum targets), B (minimum targets connected), C (medium targets), D (medium targets connected), E (maximum targets), and F (maximum targets connected). Suitable woodland caribou habitat (Caribou), Closed spruce (C. spruce), open spruce (O. spruce), mixed-wood (Mixed), tall shrub (T. shrub), low shrub (L. shrub).

5. Literature Cited

Alexander, S.A., Ferguson, R.S. & McCormick, K.J. (1991). *Key Migratory Bird Terrestrial Habitat Sites in the Northwest Territories.* Canadian wildlife service, Ottawa, Ontario.

Armstrong, G.W. & Cumming, S.G. (2003). Estimating the cost of land base changes due to wildfire using shadow prices. *For. Sci.*, 49, 719-730.

Austin, J.E., Afton, A.D., Anderson, M.G., Clark, R.G., Custer, C.M., Lawrence, J.S., Pollard, J. B. & Ringelman, J.K. (2000). Declining scaup populations: issues, hypotheses, and research needs. *Wildlife Soc. B.,* 28, 254-263.

Ball, I. & Possingham, H. (2000). MARXAN (v1.8.2): Marine Reserve Design Using Spatially Explicit Annealing, A manual. [WWW document] URL http://www. ecology.uq.edu.au/ index.html?page=27710

Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., Moberg, F. & Nystrom, M. (2003). Reserves, resilience and dynamic landscapes. *Ambio,* 32, 389-396.

Black, R.A. & Bliss, L.C. (1978). Recovery sequence of *Picea mariana - Vaccinium uliginosum* forests after burning near Inuvik, Northwest Territories, Canada. *Can. J. Botany* 56, 2020-2030.

Cabeza, M. & Moilanen, A. (2001). Design of reserve networks and the persistence of biodiversity. *Trends Ecol. Evol.* 16, 242-248.

Cardillo, M., Mace, Georgina, M.M., Gittleman, J.L., & Purvis, A. (2006). Latent extinction risk and the future battlegrounds of mammal conservation. *Proc. Natl. Acad. Sci. USA.,* 103, 4157-4161.

Carroll, C., Noss, R.F., Paquet, P.C. & Schumaker, N.H. (2003). Use of population viability analysis and reserve selection algorithms in regional conservation plans. *Ecol. Appl.,* 13, 1773-1789.

COSEWIC. (2003). *COSEWIC Assessment and Update Status Report on the Woodland Caribou (Rangifer tarandus caribou) in Canada.* Committee on the status of endangered wildlife in Canada, Ottawa, Ontario.

Cumming, S.G., Burton, P.J. & Klinkenberg, B. (1996). Boreal mixedwood forests may have no "representative" areas: some implications for reserve design. *Ecography,* 19, 162-180.

Cumming, S.G. (2001). A parametric model of fire-size distribution. *Can. J. Forest Res.*, 31, 1297-1303.

Cumming, S.G. & Armstrong, G.W. (2001). Divided land base and overlapping forest tenure in Alberta, Canada: a simulation study exploring costs of forest policy. *Forest. Chron.*, 77, 501-508.

Decarie, R. (1995). Habitat use by brood-rearing waterfowl in subarctic Quebec. *Arctic,* 48, 383-393.

Ducks Unlimited. (2002). *Lower Mackenzie River Delta, NT, Earth Cover Classification User's Guide*. Ducks Unlimited, Rancho Cordova, California.

Ducks Unlimited. (2003). *Peel Plateau Project Earth Cover's User's Guide*. Ducks Unlimited, Rancho Cordova, California.

Fall, A. & Fall, J. (2001). A domain-specific language for models of landscape dynamics. *Ecol. Model.,* 141, 1-18.

Ferguson, R.S. (1987). *Wildlife Areas of Special Interest to the Department of Renewable Resources.* Department of Renewable Resources, Government of the Northwest Territories, Yellowknife, Northwest Territories.

Ferrier, S., Pressey, R.L. & Barrett, T.W. (2000). A new predictor of the irreplaceability of areas for achieving a conservation goal, its application to real-world planning, and a research agenda for further refinement. *Biol. Conserv.,* 93, 303-325.

Government of the Northwest Territories. (2005). *Northwest Territories Fire History Data*. Forest Management Division, Resources, Wildlife and Economic Development, Yellowknife, Northwest Territories.

Government of Yukon. (2005). *Yukon Territory Fire History Data*. Wildland Fire Management, Department of Community Services, Whitehorse, Yukon.

Guisan, A. & Thuiller, W. (2005). Predicting species distribution: offering more than simple habitat models. *Ecol. Lett.*, 8, 993-1009.

Gwich'in Land Use Planning Board. (2003). *Working for the Land.* Gwich'in land use planning board, Inuvik, Northwest Territories.

Halpren, B., Regan, H.M., Possingham, H.P., & McCarthy, M. A. (2006). Accounting for uncertainty in marine reserve design. *Ecol. Lett.,* 9, 2-11.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H. & Roberts, C. (2005). Confronting the biome crisis: global disparities of habitat loss and protection. *Ecol. Lett.,* 8, 23-29.

Johnson, C.J., Seip, D.R. & Boyce, M.S. (2004). A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. *J. Appl. Ecol.,* 41, 238-251.

Johnson, D.H. & Grier, J.W. (1988). Determinants of breeding distribution of ducks. *Wildlife Monogr.,* 100.

Johnson, E.A. (1992). *Fire and Vegetation Dynamics: studies from the North American Boreal Forest.* Cambridge University Press, New York.

Margules, C.R. & Pressey R.L. (2000). Systematic conservation planning. *Nature,* 405, 243-253.

McDonnell, M.D., Possingham, H.P., Ball, I.R. & Cousins, E.A. (2002). Mathematical methods for spatially cohesive reserve design. *Environ. Model. Assess.,* 7, 107-114.

Meir, E., Andelman, S., & Possingham, H.P. (2004). Does conservation planning matter in a dynamic and uncertain world? *Ecol. Lett.*, 7, 615-622.

Mladenoff, D.J. (2004). LANDIS and forest landscape models. *Ecol. Model.* 180, 7-19.

Moilanen, A. & Cabeza, M. (2002). Single-species dynamic site selection. *Ecol. Appl.,* 12, 913-926.

Nagy, J.A., Derocher, A., Nielsen, S., Wright, W. & Heikkila, J. (2006). *Modelling Seasonal Habitats of Boreal Woodland Caribou at the Northern Limits of Their Range: a Preliminary Assessment of the Lower Mackenzie River Valley, Northwest Territories, Canada.* Department of Environment and Natural Resources, Yellowknife, Northwest Territories.

National Energy Board. (1999). *Mainland and Beaufort Delta Historical Seismic Lines.* National Energy Board, Calgary, Alberta.

Natural Resources Canada. (2000). *Canadian Digital Elevation Data (CDED), 1:250,000, NTS map sheet 106D-P, 107A-B, 116H-I,P, 096E,L.* Centre for Topographic Information, Sherbrooke, Quebec.

Nielsen, S.E., Stenhouse, G.B., & Boyce, M.S. (2006). A habitat-based framework for grizzly bear conservation in Alberta. *Biol. Conserv.*, in press.

Noss, R.F., Carroll, C., Vance-Borland, K. & Wuerthner, G. (2002). A multicriteria assessment of the irreplaceability and vulnerability of sites in the greater Yellowstone ecosystem. *Conserv. Biol.,* 16, 895-909.

Possingham, H., Ball, I. & Andelman, S. (2000). Mathematical methods for identifying representative reserve networks. In: *Quantitative Methods for Conservation Biology* (eds. Ferson, S. & Burgman, M.). Springer-Verlag, New York, pp. 291-305.

Pressey, R.L., Humphries, C.J., Margules, C.R., Vanewright, R.I. & Williams, P.H. (1993). Beyond opportunism - key principles for systematic reserve selection. *Trends Ecol. Evol.,* 8, 124-128.

Pressey, R.L. & Logan, V.S. (1998). Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in western New South Wales. *Biol. Conserv.,* 305-319.

Pressey, R.L. & Taffs, K.H. (2001). Scheduling conservation action in production landscapes: priority areas in western New South Wales defined by irreplaceability and vulnerability to vegetation loss. *Biol. Conserv.,* 100, 355-376.

Pressey, R.L., Watts, M.E. & Barrett, T.W. (2004). Is maximing protection the same as minimizing loss? Efficiency and retention as alternative measures of the effectiveness of proposed reserves. *Ecol. Lett.,* 7, 1035-1046.

Pyke, C.R. & Fischer, D.T. (2005). Selection of bioclimatically representative biological reserve systems under climate change. *Biol. Conserv.,* 121, 429-441.

Smith, R.J. (2004) Conservation Land-Use Zoning (CLUZ) software. [WWW document] URL http://www.mosaic-conservation.org/cluz.

Strang, R.M. & Johnson, A.H. (1981). Fire and climax spruce forests in central Yukon. *Arctic,* 34, 60-61.

Van Wagner, C.E. (1978). Age-class distribution and the forest fire cycle. *Can. J. Forest Res.,* 8, 220-227.

Walters, C.J. (1986). *Adaptive Management of Renewable Resources.* Macmillan, New York.

Warman, L.D., Sinclair, A.R.E., Scudder, G.G.E., Klinkenberg, B. & Pressey, R.L. (2004). Sensitivity of systematic reserve selection to decisions about scale, biological data, and targets: case study from Southern British Columbia. *Conserv. Biol.,* 18, 655-666.

Wilson, K.A., McBride, M.F., Bode, M., & Possingham, H.P. (2006). Prioritizing global conservation efforts. *Nature,* 440, 337-340.

Chapter 3

Do heritage sites and protected areas overlap?[2]

1. Introduction

Systematic conservation planning (sensu Margules & Pressey 2000) is an explicit and structured approach to protected areas design that has been adopted around the world (Pressey et al. 1993; Noss et al. 2002; Cowling & Pressey 2003; Venevsky & Venevskaia 2005). The conventional application of systematic conservation planning uses site selection algorithms (Possingham et al. 2000) to identify regions of high conservation priority based on conservation targets for focal species, landscape features, and special elements (Cabeza & Moilanen 2001; Margules et al. 2002; Noss et al. 2002; Warman et al. 2004). The special elements in these analyses can include areas with particular ecological significance, such as rare species occurrences and pristine sites, or other unique features such as heritage sites - sites recognized as sacred or important for indigenous peoples (Noss 1996; Groves et al. 2002). Special elements are a prominent part of many conservation plans (Groves et al. 2002; Noss et al. 2002; Pressey et al. 2003); however, the focus of most conservation planning studies has been on protecting conservation features like biodiversity hotspots. Consequently, conservation plans developed in regions overlapping indigenous lands often fail to incorporate heritage sites in their analysis (e.g., Carroll et al. 2003; Warman et al. 2004; Venevsky & Venevskaia 2005; Wiersma & Urban 2005).

Heritage sites are meant to capture areas of cultural and social significance but some authors have suggested that indigenous cultures also support the protection of biodiversity and ecological processes because areas of high cultural diversity may correspond with areas of high biodiversity (Alcorn 1993; Oveido & Brown 1999; Garibaldi & Turner 2004). I might expect; therefore, that the protection of heritage sites also confers protection to areas of more conventional conservation priority (e.g., high biodiversity, focal species habitat, environmental representation; Watson et al. 2003). Likewise, the protection of areas of conventional conservation priority may provide protection of traditional sites and activities (Huntington 2002). Investigating the

---

[2] This chapter is formatted for submission to *Conservation Biology* and is currently in review at *Conservation Biology* by Leroux, S.J., Schmiegelow, F.K.A., Nagy, J. Reprinted with permission from *Conservation Biology*.

relationship between sites recognized as sacred or important for indigenous peoples and sites identified for conventional conservation is critical in regions that have high overlap with indigenous lands.

In Canada, 12 % of national, provincial, and territorial parks overlap with indigenous reserves, settlement areas, or land claims. Indigenous reserves, settlement areas, or land claims also cover 40 % of the Canadian landscape. Indigenous peoples play a particularly important role in boreal Canada because there are more than 600 indigenous communities in this region (Canadian Boreal Initiative 2005). Canada's boreal region has high conservation potential because it supports some of the last wilderness areas in the world (Sanderson et al. 2002), containing approximately one quarter of all intact forests remaining globally (Bryant et al. 1997). Traditional Ecological Knowledge held by local peoples can improve conservation planning (Berkes et al. 2000; Huntington 2000; Chhatre & Saberwal 2005; Danby & Slocombe 2005; Drew 2005; Herrmann 2006). However, many conservation agencies have alienated indigenous peoples (Wells & McShane 2004), the primary users and communities of protected areas on indigenous lands, resulting in disaffection of indigenous peoples towards existing protected areas (Stadel et al. 2002). In the Canadian boreal region, the current and expected expansion of resource extraction (Schneider et al. 2003) is being countered with increasing efforts to identify more areas for permanent protection (Canadian Boreal Initiative 2005). Nevertheless, many indigenous communities in the boreal region seek to manage their lands through community-based land-planning processes (Deh Cho 2001; Sahtu Land Use Planning Board 2002; Gwich'in Land Use Planning Board 2003; Taku River 2003). Consequently, protected areas design in the boreal region must be done in conjunction with community-based land-use planning

Community-based land-use planning involves communities in every step of the planning process (Sahtu Land Use Planning Board 2002; Gwich'in Land Use Planning Board 2003). In general, it is a non-systematic approach that reflects a host of values including social, cultural, and economic interests, and incorporates both traditional and scientific knowledge (Sahtu Land Use Planning Board 2002; Gwich'in Land Use Planning Board 2003). The Government of the Northwest Territories (GNWT), Canada, has developed a protected areas strategy following these principles (Northwest Territories

Protected Areas Advisory Committee 1999; Stadel et al. 2001). The primary goals of the strategy are to 1) protect special natural and cultural areas, and 2) protect core representative areas in each ecoregion. The protected areas strategy has worked with communities to identify key natural and cultural areas (i.e., community heritage sites) and plans to complement these assessments with conventional conservation planning to identify additional representative areas in each ecoregion. A better understanding of the relationship between community heritage sites and sites identified to protect conventional conservation features could inform the GNWT Protected Areas Strategy.

In this chapter, I quantify the relationship between heritage sites and sites independently identified using conventional conservation targets. To assess this, I undertake a case study in the Gwich'in settlement area in the Northwest Territories, comparing community heritage sites identified by the Gwich'in with protected areas that I designed to protect focal species, environmental representation, and environmentally significant areas. My goal is to better understand the similarities, differences, and complementarities in the spatial configuration of sites recognized as sacred or important for the Gwich'in and sites recognized for their conventional conservation value.

2. Methods

2.1 Study area

The study area is 22,000 km^2 in the north-west region of the Northwest Territories, located in the northern region of the Gwich'in settlement area (Fig. 3-1). The three main towns in the area are Inuvik, Fort McPherson, and Tsiigetchic. The northern boundary of the study area was determined by the northern boundary of the Gwich'in settlement area, the eastern boundary by the extent of available earth cover data, and the western and southern boundaries by the extent of the Gwich'in heritage site analysis. The study area is bordered by the Inuvialuit Settlement Region to the north and the Sahtu Settlement Area to the east. The landscape is flat, wet, and dominated by black spruce bogs and scattered lakes. Permafrost is continuous. The dominant tree species is black spruce (*Picea mariana* Mill.), followed by white spruce (*Picea glauca* Moench), white birch (*Betula papyrifera* Marsh.), and tamarack (*Laryx laricina*). Shrubs are abundant in the area; the main shrubs species are bog birch (*Betula glandulosa* Michx.), labrador tea (*Ledum spp.* L.), bearberry (*Arctostaphylos rubra* Rehd. & Wils.), and blueberries

(*Vaccinium spp.* L.). The area has long, cold winters (-20°C to -30°C) and short cool summers (10°C to 15°C). Fire is the dominant natural disturbance on the landscape (Nagy et al. 2005).

2.2 Community heritage sites

The Gwich'in First Nation assembled a database on heritage sites in the Mackenzie Valley region of their land claim (Andre & Kritsch 1992). Community members were interviewed and asked to describe and map heritage sites. All heritage sites were later digitized using a geographic information system. The data include camps (n = 299), interconnected trails (n = 299), and culturally-significant places (n = 201). These heritage sites cover 14 % of the study area. I do not present figures showing the location of the heritage sites because these data are sensitive and proprietary.

2.3 Conventional protected areas network

I developed a hypothetical protected areas network for the study region based on conventional protected areas planning methods. The protected areas network was designed using a heuristic algorithm that combined targets for 1) the relative probability of occurrence of woodland caribou (*Rangifer tarandus caribou*), 2) the representation of high-quality wetlands, vegetation types, and waterbodies, and 3) the inclusion of environmentally significant areas (i.e., migratory bird areas, terrestrial parks, and wildlife areas) (Fig. 3-2). I used species and earth cover data, both of which have proven valuable in protected areas design (Brooks et al. 2004; Cowling et al. 2004; Higgins et al. 2004; Pressey 2004).

Planning units provide the framework for constructing a protected areas network, but there is no strong theoretical basis for selecting the size and shape of planning units (Pressey & Logan 1998). I used planning units of uniform size and geometry (2 x 2 km; n = 5,675) for the protected areas site selection. With planning units of this resolution, I was able to run the heuristic algorithm on the entire study area. The conservation features included in the site selection were analyzed at a finer resolution (500 x 500 m; n = 87,430) to capture the finer details of these features.

2.3.1 Relative probability of occurrence of woodland caribou

I chose boreal woodland caribou as a focal species in the analyses because they are wide ranging, medium-sized ungulates that are distributed across the boreal forest and are listed as threatened in Canada (COSEWIC 2003). I developed the conservation plan using a habitat selection model. The core modelling study area was defined by generating a minimum convex polygon around GPS and ARGOS satellite caribou locations (animals, n = 8; locations, n = 10,559) from 1 May 2002 – 13 October 2005. I buffered this core area by an additional 3,176 m (the 95th percentile distance travelled by GPS collared caribou in 8 hours; Nagy et al. 2006). Although part of the buffered MCP fell outside the study area, I used all data in the buffered MCP to develop better habitat selection models. I developed a habitat model for the winter season (15 January to 30 April) because the availability of winter habitat is likely most limiting for woodland caribou in the study area (Nagy et al. 2005). I generated random points (n = 60,385) at a density of 2/km^2 within the buffered core modelling area to represent available habitat.

I developed the caribou habitat model by pooling data for all caribou and generating a landscape-level resource selection function (RSF) (Manly et al. 2002) based on a suite of earth cover and terrain covariates (Table 3-1). By grouping similar earth cover classes I was able to reduce the number of classes from 34 to 10. I then re-classed the 30-m resolution earth cover map produced by Ducks Unlimited (2002, 2003) and re-scaled the map to 500-m resolution using a majority threshold filter (Parody & Milne 2004). With this resolution I was able to develop a woodland caribou RSF model for the entire study area while minimizing the aggregation and loss of unique earth cover types. A National Topographic System based digital elevation model for the area provided information on slope, aspect, and elevation (Natural Resources Canada 2000) while seismic line data was obtained from the National Energy Board (1999). In a geographic information system (GIS), I calculated the seismic line density (km/km^2) in each re-scaled 500-m grid cell. The caribou use and random points were overlaid on the 500-m resolution map grid, and habitat variables were assigned to locations.

I used logistic regression to fit a RSF model (Boyce et al. 2002), taking the form:

$$w(x) = \exp(\exists_1 x_1 + \exists_2 x_2 + ... + \exists_n x_n) \qquad (3\text{-}1)$$

where covariates x_1 to x_n represent possible combinations of earth cover and terrain covariates. I then used a linear stretch of the form:

$$w = ((w(x) - w_{\min})/(w_{\max} - w_{\min})) \qquad (3\text{-}2)$$

where $w(x)$ is the product of Eq.1 and w_{min} and w_{max} represent the smallest and largest RSF value available on the landscape, respectively (Johnson et al. 2004). The w values for the random data were sorted in ascending order, partitioned into 10 equal size bins, and assigned a bin rank between 1 and 10 (1 = lowest w values, 10 = highest w values). The use locations were assigned to the appropriate bins and assigned the appropriate bin rank. I used the χ^2 test to determine if the distribution of caribou-use locations among the 10 bins was significantly different than would be expected if the caribou had used habitats in proportion to the availability.

I separated the RSF values from the model into 10 quantile bins that represented an index of increasing relative habitat use for caribou (Nielsen et al. 2006). Similar to Nielsen et al. (2006), I considered RSF values in bins 5 to 10 to be suitable habitat for caribou, and I set conservation targets of 50 % protection for the planning units represented by bins 5 to 10. I regard 50 % as a moderate conservation target for suitable woodland caribou habitat.

2.3.2 Representation of high-quality wetlands, vegetation types, and waterbodies

In a GIS, I extracted the wetland vegetation classes from the original 30-m earth cover classification raster map. Then, I converted the wetland raster to polyline and calculated the wetland line density (km/km^2) in each re-scaled 500-m grid cell (see previous description for re-scaling process). I defined wetland areas as high quality if the wetland density in a grid cell was higher than the study area median of 3.769 km/km^2. I focused on protecting high-density wetlands in the study region because two duck species, Scaup sp. (*Aythya sp.* L. and Eyton) and Scoter sp. (*Melanitta sp.* L.), have experienced significant declines across their range (Decarie 1995; Austin et al. 2000) and high-density wetlands are the most productive duck habitat (Johnson & Grier 1988; Johnson et al. 1999). I set a moderate conservation target of protecting 50 % of high-quality wetlands in protected areas.

I also set moderate representation targets of 20% for each re-scaled earth cover type in the study area (see previous description) in the protected areas network.

2.3.3 Mapping special elements

The special elements in this analysis are environmentally significant areas, including territorial parks, wildlife areas of special interest (Ferguson 1987), and key migratory bird terrestrial habitat sites (Alexander et al. 1991). I set a target of capturing 75 % of all environmentally significant areas in the protected areas network.

2.3.4 Protected areas connectedness

Connectedness, determined by physical linkages between reserves, is a key consideration in reserve design (McDonnell et al. 2002; Warman et al. 2004). In site selection tools like MARXAN (Ball & Possingham 2000), the user can modify the level of connectedness desired for each reserve network by changing the boundary length modifier (Possingham et al. 2000; McDonnell et al. 2002). I designed the protected areas network with high connectedness as there is evidence that more connected protected areas networks better maintain their conservation targets (McDonnell et al. 2002; Warman et al. 2004).

2.3.5 Site selection

I combined targets for focal species, high-quality wetland, vegetation types, waterbodies, and special elements (Table 3-2) into MARXAN v 1.8.2 (Ball & Possingham 2000). I used the CLUZ v.1.11 user-friendly interface for MARXAN to build the protected areas networks (Smith 2004). MARXAN is a site selection tool that facilitates the identification and selection of candidate protected areas designed to satisfy a suite of conservation targets. MARXAN uses a global heuristic algorithm, in this case simulated annealing, to identify the minimum number of sites or area to achieve all stated targets (i.e., minimum area problem; Cabeza & Moilanen 2001). A simulated annealing algorithm begins by generating a random set of sites. Then, at each iteration, a site is removed or added to the random set and the value of the new set is compared to the initial one (Possingham et al. 2000; Cabeza & Moilanen 2001). The site is either added or removed from the set based on the comparison. I performed 100 runs of 1,000,000 iterations of the simulated annealing algorithm in MARXAN to select the near-minimum amount of area needed to capture stated conservation targets and identify the most irreplaceable sites (i.e., sites that contribute most to the conservation targets for the features contained) for the protected areas network.

2.4 Comparison of heritage sites and conventional protected areas

To determine the spatial overlap of Gwich'in heritage sites and protected areas identified to capture conventional conservation targets, I used a Jaccard coefficient defined as ([number of shared grid cells/(number of shared grid cells + number of additional grid cells for heritage sites + number of additional grid cells for protected areas)] x 100) (van Jaarsveld et al. 1998; Warman et al. 2004). The study area was comprised of 87,430 grid cells, including 12,439 grid cells encompassing heritage sites and 40,041 grid cells encompassing protected areas. I randomly selected 100 sets of 12,439 and 40,041 grid cells, representing the same number of grid cells as the heritage sites and the protected areas network. I generated Jaccard values for these 100 random sets and compared the random distribution of Jaccard values to the observed Jaccard value to determine the probability of obtaining the observed value by chance. I was interested in the relative similarity between heritage sites and the 13 conservation features (i.e., woodland caribou, high quality wetlands, vegetation, environmentally significant areas) used in the designation of the protected areas network; therefore I calculated Jaccard coefficients for heritage sites and all conventional conservation features.

To explore the overlap further, I calculated the percentage of each conservation feature in heritage sites and protected areas. I also determined the difference between the percentage of the heritage sites and protected areas composed of each conventional conservation feature and the percentage of the landscape composed of each conventional conservation feature in order to explore the composition of heritage sites and protected areas.

3. Results

3.1 Woodland caribou RSF model

The RSF model provides useful information on caribou habitat selection. Caribou selected areas of open spruce, near water, recent burns, and south facing aspects and avoided areas with herbaceous vegetation cover, steep slopes, and high densities of seismic lines (Table 3-1). Grid cells with an RSF value > 0.09, the cut-off for inclusion in bin 5, were considered grid cells with suitable habitat for caribou, resulting in 15,688 km^2 of suitable habitat for woodland caribou in the study area. The distribution of caribou use and random locations among RSF bins were significantly different than random (χ^2 $p < 0.001$). Greater than expected numbers of use locations occurred in the upper 5 bins, with the majority of these occurring in the upper 2 bins, indicating that the model had good predictive ability.

3.2 Conventional protected areas network

The protected areas network included 6 areas covering 46 % of the study area to meet the targets (Fig. 3-3). The protected areas were distributed across the study area, with 5 small and 1 large site. Most of the initial targets were over-represented in the network with the exception of that for woodland caribou habitat (Fig. 3-4).

3.3 Spatial overlap of heritage sites and protected areas network

The degree of spatial overlap (calculated as Jaccard coefficient) between heritage sites and the conventional protected areas network was 10.88, which was significantly lower ($p < 0.0001$) than random spatial overlap (mean = 12.17).

The degree of spatial overlap between heritage sites and most conventional conservation features was also low (< 10). The heritage sites overlapped most with water (24.56), suitable woodland caribou habitat (13.10), high quality wetlands (7.75), and open spruce (7.67).

The heritage sites included in this analysis (14 % of the study area) were composed of a relatively high percentage of the available migratory bird areas (60.93), water (35.26), and terrestrial parks (30) and a low percentage (< 20 %) of the remaining conventional conservation features (Fig. 3-4). At a landscape level, protected areas were composed of a lower percentage of low shrub (-5.52) and a higher percentage of suitable woodland caribou habitat (6.59) and wildlife areas (5.20) than the study area (Fig. 3-5). The remaining conservation features represented similar percentages (< ± 5 % difference) of the protected areas and the study area. Heritage sites were composed of a lower percentage of open spruce (-17.6) and high quality wetlands (-6.1) and a higher percentage of water (26.68) than the study area (Fig. 3-5), whereas the remaining conservation features were represented by similar percentages (< ± 5 % difference).

4. Discussion

The conventional application of systematic conservation planning acknowledges the importance of incorporating special elements like indigenous heritage sites into conservation plans (Groves et al. 2002; Noss et al. 2002; Pressey et al. 2003). Many conservation planners; however, have omitted heritage sites in their protected areas design (e.g., Carroll et al. 2003; Warman et al. 2004; Venevsky & Venevskaia 2005; Wiersma & Urban 2005). I have provided a quantitative assessment of the overlap between heritage sites and protected areas designed to capture conventional conservation targets in a boreal landscape in northern Canada. I expected to observe high overlap because some authors have argued that areas of high cultural diversity are often associated with areas of conventional conservation priority (Oveido & Brown 1999; Garibaldi & Turner 2004). Instead, I found that the overlap between heritage sites and protected areas was significantly lower than random in the study area. The results suggest that protected areas networks designed to capture conventional conservation features like focal species and environmental variation may not effectively protect key heritage sites. Similarly, heritage sites designed to capture culturally significant features may not effectively capture other conservation features.

I observed that Gwich'in heritage sites were concentrated on or near waterbodies, likely because these features are important for harvesting fish and animals, as well as for travel and other cultural reasons. The protected areas network was not designed to capture a high proportion of waterbodies; however, by incorporating heritage sites into protected areas design, better protection of aquatic elements would be afforded. The Gwich'in heritage sites also included trails and entire lakes, which have a high degree of connectedness. The protected areas network was highly connected by conventional criteria and covered 46 % of the landscape, yet it did not have the connectivity required for the heritage sites in the study area. The case study has shown that heritage sites may capture different features than conventional protected area designs; therefore explicit incorporation of heritage sites is necessary if conservation planning is to address both cultural and ecological values.

If conservation planners incorporate heritage sites into protected areas plans, continuation of traditional activities that occur in heritage sites should be permitted. While some conservationists may resist allowing traditional activities in protected areas, on indigenous lands, some consider indigenous peoples as an essential part of the ecosystem (Schwartzman et al. 2000; Huntington 2002; Herrmann 2006). Further, establishment of protected areas can be an effective method for preserving the relationship between humans and ecosystems (Watson et al. 2003). Protected areas design could proceed independently of community-based planning, but I believe that such efforts are likely to fail, alienate indigenous communities, and foster resentment towards protected areas. For example, Parks Canada has recently begun working with aboriginal communities to identify Aboriginal Cultural Landscapes on indigenous lands (Parks Canada 2004) in order to overcome problems associated with previous protected areas designations that largely excluded aboriginal interests.

These findings agree with suggestions that other types of knowledge can contribute to developing conservation plans that more effectively protect natural and cultural features (Ludwig et al. 2001; Pfister 2002). To better capture community heritage sites, conventional conservation planning methods can explicitly incorporate representation targets based on areas of importance to local communities. In areas where indigenous activities are prevalent, heritage sites could be afforded the highest conservation priority (e.g., Folke 2004). In isolation; however, heritage sites in the study area did not capture a high percentage of other conservation features, including suitable woodland caribou habitat, high quality wetland areas, and certain earth cover types (e.g., open spruce and low shrub). If communities wish to protect such features, they may need to identify additional areas for protection to complement their heritage sites. Conventional protected areas design methods can be used to identify such areas (e.g., Deh Cho 2001; Taku River 2003).

It is challenging for conservation planners to incorporate the interests of remote communities, but to achieve conservation, local interests cannot be ignored (Alcorn 1993; Schwartzman et al. 2000; Brosius 2004). To facilitate the exchange of information between conservation biologists and communities, I suggest developing co-operative working groups with government agencies and other organizations that are linked to communities to better understand community interests. By forging these relationships, conservation planners will be able to design protected areas that will more effectively capture both cultural and natural features. Likewise, communities that seek to protect biodiversity and ecological processes may benefit from using conventional protected areas design methods. Collaboration and exchange of information among conservation biologists and local peoples can be mutually beneficial (Drew 2005).

The GNWT has worked with non-governmental organizations and local communities to incorporate community interests at the beginning of the conservation planning process (Stadel et al. 2002; Northwest Territories Protected Areas Advisory

Committee 1999; Northwest Territories Protected Areas Strategy Secretariat 2003). The GNWT also uses other protected areas criteria and analysis to complement potential protected areas identified by communities. This process draws on the strengths of community land-use planning and systematic conservation planning to identify potential protected areas networks with a high probability of being implemented, as opposed to mere paper parks (Alcorn 1993; Schwartzman et al. 2000). The Northwest Territories Protected Areas Strategy could be a model for other conservation agencies attempting to design effective protected areas networks on indigenous lands and for indigenous communities attempting to protect biodiversity and ecological processes on their lands. This study has clearly identified the advantages of such complementarities in achieving a broad range of conservation objectives.

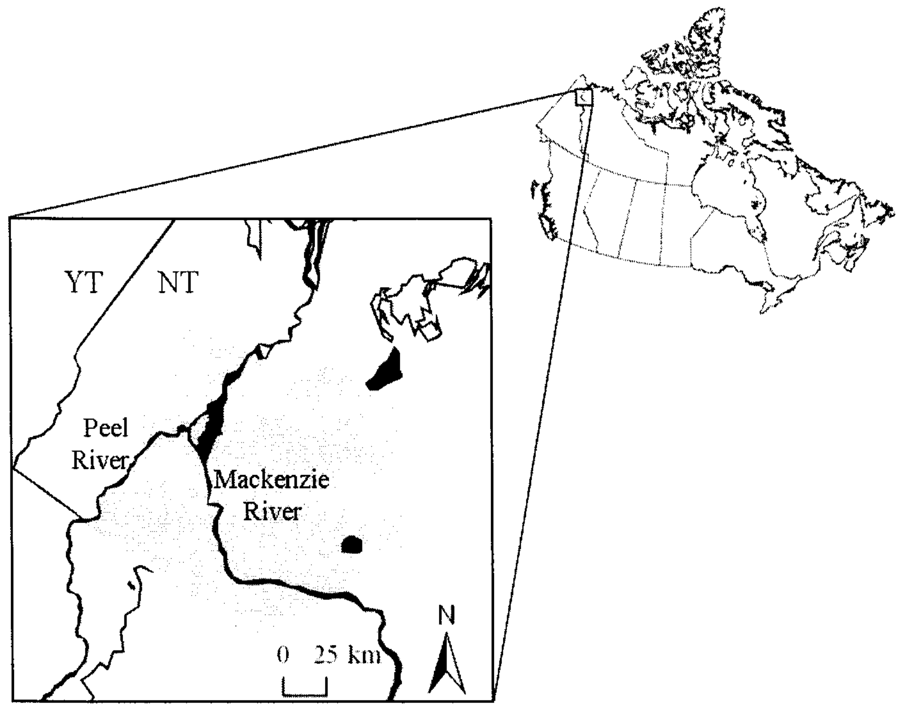

Figure 3-1 Study area in the northern boreal region of Canada shown as grey polygon. The study area is 22,000 km^2 and is mostly in the Northwest Territories (NT) with parts in the Yukon Territory (YT).

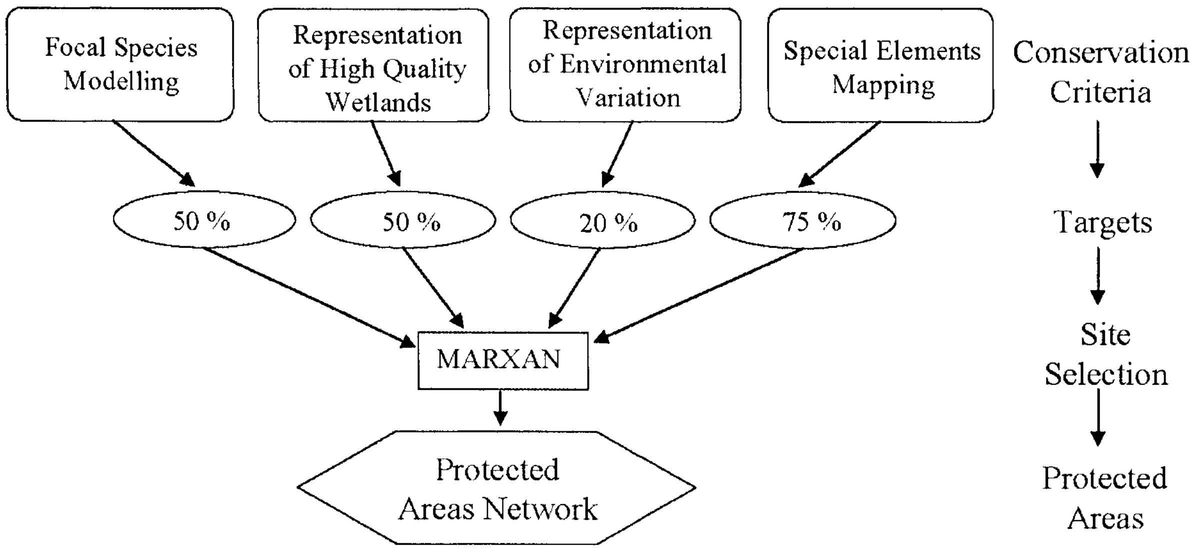

Figure 3-2 Conservation planning approach adopted to design the protected areas network. I combined targets for focal species, representation of high quality wetlands, representation of earth cover types, and special elements mapping into the site-selection software, MARXAN, to identify a protected areas network that met the targets.

Table 3-1 Coefficients and test statistics of the RSF model for woodland caribou selection of habitat variables during mid/late winter (15 January to 30 April).

Variable	Coefficient	Standard Error	P>z
Slope (%)	-0.2646	0.0162	0.0000
Aspect flat	0		
Aspect north	0.8722	0.2025	0.0000
Aspect east	0.9811	0.2022	0.0000
Aspect south	1.4474	0.2013	0.0000
Aspect west	1.0633	0.2012	0.0000
Median elevation (m)	-0.0005	0.0002	0.0100
Seismic lines (km/km^2)	-0.1640	0.0227	0.0000
Closed spruce	0		
Open spruce	2.3836	0.3804	0.0000
Mixed	0.6512	0.4652	0.1620
Tall shrub	0		
Low Shrub	0.5039	0.3906	0.1970
Herbaceous	-0.5089	0.8050	0.5270
Burn	1.5721	0.3906	0.0000
Wetland	0		
Water	1.5135	0.3864	0.0000
Other	0.7149	0.4187	0.0880
Constant	-5.5883	0.4256	0.0000

Table 3-2 Conservation targets used in developing the protected areas network for the study area. I set targets for capturing suitable woodland caribou habitat, high quality wetlands, vegetation, waterbodies, and environmentally significant areas. I used a high measure of connectedness in the protected areas network..

Feature	Target (%)
Closed spruce	20
Open spruce	20
Mixed	20
Tall shrub	20
Low shrub	20
Herbaceous	20
Burn	20
Wetland	20
Water	20
Woodland caribou	50
High density wetland	50
Territorial parks	75
Migratory bird areas	75
Wildlife areas	75

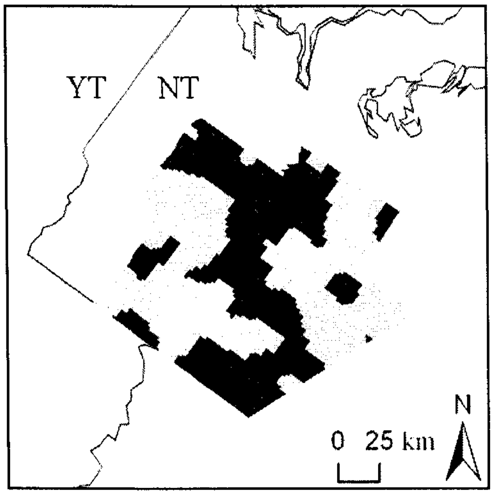

Figure 3-3 Hypothetical protected areas network (black) designed to capture suitable habitat for woodland caribou, high quality wetlands, vegetation types, waterbodies, wildlife areas of special interest, key migratory bird terrestrial habitat sites, and territorial parks. The protected areas network is composed of six protected areas that cover 46 % of the study area.

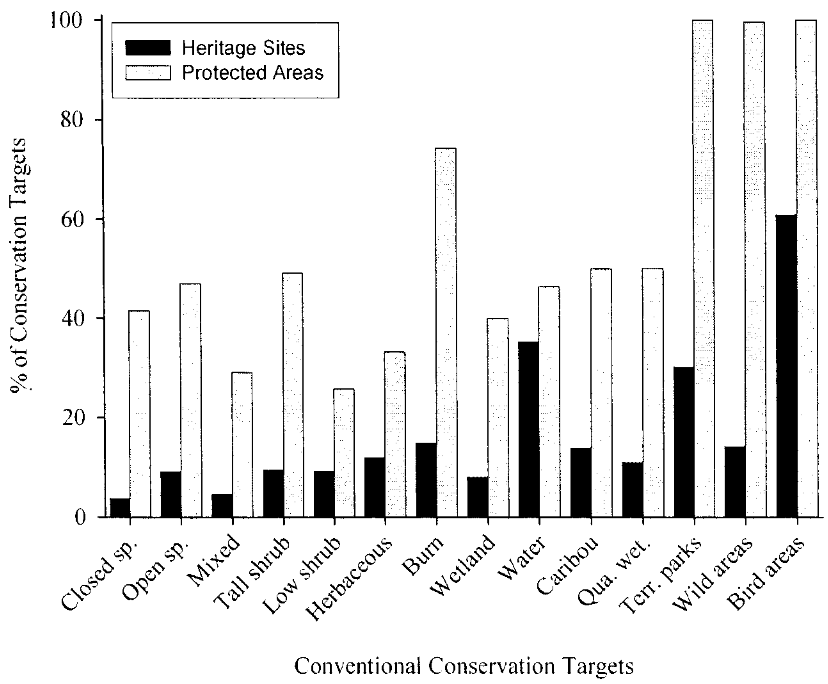

Figure 3-4 Percentage of each conservation feature in heritage sites and protected areas. Closed spruce (Closed sp.), open spruce (Open sp.), suitable woodland caribou habitat (Caribou), high-quality wetlands (Qua. wet.), territorial parks (Terr. parks), wilderness areas of special interest (Wild areas), and key migratory bird terrestrial habitat sites (Bird areas).

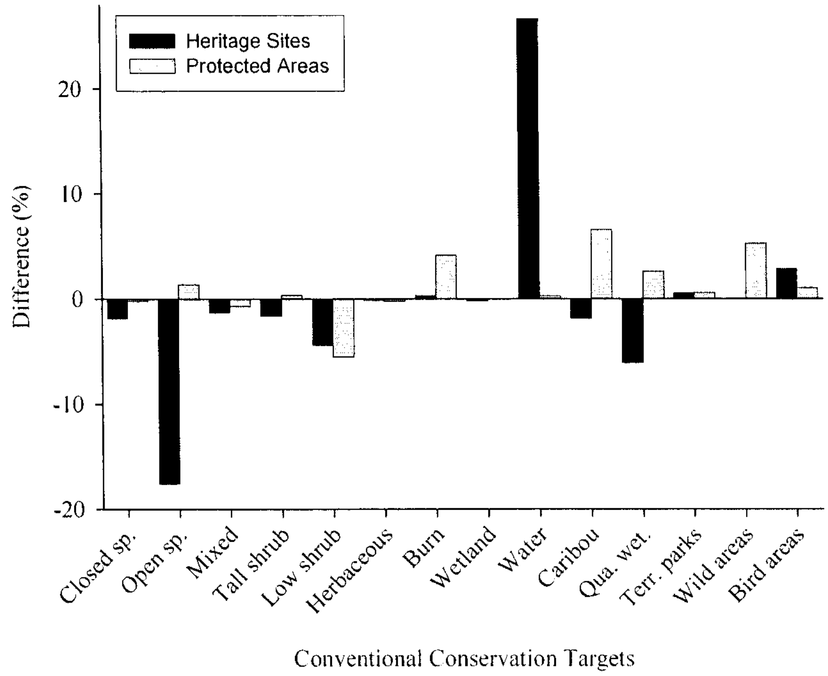

Figure 3-5 Difference between the percentage of the heritage sites and protected areas composed of each conventional conservation feature and the percentage of the landscape composed of each conventional conservation feature. Positive values indicate that the conservation feature made up a higher percentage of the heritage sites or protected areas than the landscape. Closed spruce (Closed sp.), open spruce (Open sp.), suitable woodland caribou habitat (Caribou), high-quality wetlands (Qua. wet.), territorial parks (Terr. parks), wilderness areas of special interest (Wild areas), and key migratory bird terrestrial habitat sites (Bird areas).

5. Literature Cited

Alcorn, J. B. 1993. Indigenous peoples and conservation. Conservation Biology **7**:424-426.

Alexander, S. A., R. S. Ferguson, and K. J. McCormick. 1991. Key migratory bird terrestrial habitat sites in the Northwest Territories. Canadian Wildlife Service, Ottawa, Ontario.

Andre, A. and I. Kritsch. 1992. The traditional use of the Travaillant Lake area using trails and place names of the Gwichya Gwich'in from Arctic Red River, N.W.T. Gwich'in Social Cultural Institute, Arctic Red River, Northwest Territories.

Austin, J. E., A. D. Afton, M. G. Anderson, R. G. Clark, C. M. Custer, J. S. Lawrence, J. B. Pollard, and J. K. Ringelman. 2000. Declining scaup populations: issues, hypotheses, and research needs. Wildlife Society Bulletin **28**:254-263.

Ball, I. and H. Possigham. 2000. MARXAN (v1.8.2): marine reserve design using spatially explicit annealing, A manual. The University of Queensland, Brisbane, Australia. Available from http:www.ecology.uq.edu.au/index.html?page=27710 (accessed April 2005).

Berkes, F., J. Colding, and C. Folke. 2000. Rediscovery of traditional ecological knowledge as adaptive management. Ecological Applications **10**:1251-1262.

Boyce, M. S., P. R. Vernier, S. E. Nielsen, and F. K. A. Schmiegelow. 2002. Evaluating resource selection functions. Ecological Modelling **157**: 281-300.

Brooks, T., G. A. B. Da Fonseca, and A. S. L. Rodrigues. 2004. Species, data, and conservation planning. Conservation Biology **18**:1682-1688.

Brosius, J. P. 2004. Indigenous peoples and protected areas at the World Parks Congress. Conservation Biology **18**:609-612.

Bryant, D., D. Nielsen, and L. Tangley. 1997. The last frontier forests: ecosystems & economies on the edge. World Resource Institute, Washington D.C.

Cabeza, M., and A. Moilanen. 2001. Design of reserve networks and the persistence of biodiversity. Trends in Ecology and Evolution **16**:242-248.

Canadian Boreal Initiative. 2005. The boreal in the balance: securing the future of Canada's boreal region: a status report. Canadian Boreal Initiative, Ottawa, Ontario.

Carroll, C., R. F. Noss, P. C. Paquet, and N. H. Schumaker. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications **13**:1773-1789.

Chhatre, A., and V. Saberwal. 2005. Political incentives for biodiversity conservation. Conservation Biology **19**:310-317.

COSEWIC. 2003. COSEWIC assessment and update status report on the woodland caribou (*Rangifer tarandus caribou*) in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, Ontario.

Cowling, R. M., and R. L. Pressey. 2003. Introduction to systematic conservation planning in the Cape Florisitic Region. Biological Conservation **112**:1-13.

Cowling, R. M., A. T. Knight, D. P. Faith, S. Ferrier, A. T. Lombard, A. Driver, M. Rouget, K. Maze, and P. G. Desmet. 2004. Nature conservation requires more than a passion for species. Conservation Biology **18**:1674-1676.

Danby, R. K., and D. S. Slocombe. 2005. Regional ecology, ecosystem geography, and transboundary protected areas in the St. Elias Mountains. Ecological Applications **15**:405-422.

Decarie, R. 1995. Habitat use by brood-rearing waterfowl in subarctic Quebec. Arctic **48**:383-393.

Deh Cho First Nations, Government of Canada, and Government of the Northwest Territories. 2001. Deh Cho First Nations interim measures agreement. Deh Cho First Nations, Fort Simpson, Northwest Territories.

Drew, J. A. 2005. Use of traditional ecological knowledge in marine conservation. Conservation Biology **19**:1286-1293.

Ducks Unlimited. 2002 . Lower Mackenzie River delta, NT, earth cover classification user's guide. Ducks Unlimited, Rancho Cordova, California.

Ducks Unlimited. 2003 . Peel plateau project earth cover's user's guide. Ducks Unlimited, Inc., Rancho Cordova, California.

Ferguson, R. S. 1987. Wildlife areas of special interest to the department of renewable resources. Department of renewable resources, government of the Northwest Territories, Yellowknife, Northwest Territories.

Folke, C. 2004. Traditional knowledge in social-ecological systems. Ecology and Society **9**:7.

Garibaldi, A., and N. Turner. 2004. Cultural keystone species: implications for ecological conservation and restoration. Ecology and Society **9**:1.

Groves, C. R., D. B. Jensen, L. L. Valutis, K. H. Redford, M. L. Shaffer, J. M. Scott, J. V.

Baumgartner, J. V. Higgins, M. W. Beck, and M. G. Anderson. 2002. Planning for biodiversity conservation: putting conservation science into practice. Bioscience **52**:499-512.

Gwich'in Land Use Planning Board. 2003. Working for the land. Gwich'in Land Use Planning Board, Inuvik, Northwest Territories.

Herrmann, T. M. 2006. Indigenous knowledge and management of *Araucaria araucana* forest in the Chilean Andes: implications for native forest conservation. Biodiversity and Conservation **15**: 647-662.

Higgins, J. V., T. H. Ricketts, J. D. Parrish, E. Dinerstein, G. Powell, S. Palminteri, J. M. Hoekstra, J. Morrison, A. Tomasek, and J. Adams. 2004. Beyond Noah: saving species is not enough. Conservation Biology **18**:1672-1673.

Huntington, H. P. 2000. Using traditional ecological knowledge in science: methods and applications. Ecological Applications **10**:1270-1274.

Huntington, H. P. 2002. Can traditional ecological knowledge and wilderness benefit one another? Pages 64-68 in A. E. Watson, L. Alessa, and J. Sproull, editors. Wilderness in the circumpolar north: search for compatibility in ecological , traditional, and ecotourism values. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, Utah.

van Jaarsveld, A. S., S. Freitag, S. L. Chown, C. Muller, S. Koch, H. Hull, C. Bellamy, M. Kruger, S. Endrody-Younga, M. W. Mansell, and C. H. Scholtz. 1998. Biodiversity assessment and conservation strategies. Science **279**:2106-2108.

Johnson, C. J., D. R. Seip, and M. S. Boyce. 2004. A quantitative approach to conservation planning: using resource selection functions to map the distribution of mountain caribou at multiple spatial scales. Journal of Applied Ecology **41**:238-251.

Johnson, D. H., and J. W. Grier. 1988. Determinants of breeding distribution of ducks. Wildlife Monographs **100**.

Johnson, R. R., K. F. Higgins, D. E. Naugle, and J. A. Jenks. 1999. A comparison of sampling techniques to estimate number of wetlands. Wildlife Society Bulletin **27**:103-108.

Ludwig, D., M. Mangel, and B. Haddad. 2001. Ecology, conservation, and public policy. Annual Reviews of Ecology and Systematics **32**:481-517.

Manly, B. F. J., L. L. McDonald, D. L. Thomas, T. L. McDonald, and W. P. Erickson. 2002. Resource selection by animals: statistical design and analysis for field studies. Kluwer Academic Publishers. Dordrecht, the Netherlands.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. Nature **405**: 243-253.

Margules, C. R., R. L. Pressey, and P. H. Williams. 2002. Representing biodiversity: data and procedures for identifying priority areas for conservation. Journal of Biosciences **27**:309-326.

McDonnell, M. D., H. P. Possingham, I. R. Ball, and E. A. Cousins. 2002 . Mathematical methods for spatially cohesive reserve design. Environmental Modeling and Assessment **7**:107-114.

Nagy, J. A., D. Auriat, W. Wright, T. Slack, I. Ellsworth, and M. Kienzler. 2005. Ecology of boreal woodland caribou in the lower Mackenzie Valley, NT. Department of Resources, Wildlife, and Economic Development, Yellowknife, Northwest Territories.

Nagy, J. A., A. Derocher, S. Nielsen, W. Wright, and J. Heikkila. 2006. Modelling seasonal habitats of boreal woodland caribou at the northern limits of their range: a preliminary assessment of the lower Mackenzie River valley, Northwest Territories, Canada. Department of Environment and Natural Resources, Yellowknife, Northwest Territories.

National Energy Board. 1999. Mainland and Beaufort Delta Historical Seismic Lines. National Energy Board, Calgary, Alberta.

Natural Resources Canada. 2000. Canadian digital elevation data (CDED), 1:250,000, NTS map sheet 106D-P, 107A-B, 116H-I,P, 096E,L. Centre for Topographic Information, Sherbrooke, Quebec.

Nielsen, S. E., G. B. Stenhouse, and M. S. Boyce. 2006. A habitat-based framework for grizzly bear conservation in Alberta. Biological Conservation: in press.

Northwest Territories Protected Areas Advisory Committee. 1999. NWT Protected Areas Strategy: a balanced approach to establishing protected areas in the Northwest Territories. Northwest Territories Protected Areas Strategy Secretariat, Yellowknife, Northwest Territories.

Northwest Territories Protected Areas Strategy Secretariat. 2003. Mackenzie valley five year action plan (2004-2009): conservation planning for pipeline development. Northwest Territories Protected Areas Strategy Secretariat, Yellowknife, Northwest Territories.

Noss, R. F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the greater Yellowstone ecosystem. Conservation Biology **16**:895-909.

Noss, R. F. 1996. Protected areas: how much is enough? Pages 91-120 in R.G. Wright and J.Lemons, editors. National parks and protected areas: their role in environmental protection. Blackwell Science, Cambridge.

Oveido, G., and J. Brown. 1999. Building alliances with indigenous peples to establish and manage protected areas. Pages 99-108 in S. Stolton and N. Dudley, editors. Partnership for protection: new strategies for planning and management of protected areas. Earthscan Publications, London.

Parks Canada. 2004. An approach to Aboriginal Cultural Landscapes. Parks Canada, Ottawa, Ontario. Available from http:www.pc.gc.ca/docs/r/pca-acl/index_E.asp (accessed February 2006).

Parody, J. M., and B. T. Milne. 2004. Implications of rescaling rules for multi-scaled habitat models. Landscape Ecology **19**:691-701.

Pfister, R. E. 2002. Collaboration across cultural boundaries to protect wild places: the British Columbia experience. Pages 27-35 in A. E. Watson, L. Alessa, and J. Sproull, editors. Wilderness in the circumpolar north: search for compatibility in ecological , traditional, and ecotourism values. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, Utah.

Possingham, H., I. Ball, and S. Andelman. 2000. Mathematical methods for identifying representative reserve networks. Pages 291-305 in S. Ferson and M. Burgman, editors. Quantitative methods for conservation biology. Springer-Verlag, New York.

Pressey, R. L. 2004. Conservation planning and biodiversity: assembling the best data for the job. Conservation Biology **18**:1677-1681.

Pressey, R. L., R. M. Cowling, and M. Rouget. 2003. Formulating conservation targets for biodiversity pattern and process in the Cape Florisitic Region, South Africa. Biological Conservation **112**:99-127.

Pressey, R. L., C. J. Humphries, C. R. Margules, R. I. Vanewright, and P. H. Williams. 1993. Beyond Opportunism - Key Principles for Systematic Reserve Selection. Trends in Ecology and Evolution **8**:124-128.

Pressey, R. L., and V. S. Logan. 1998. Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in western New South Wales. Biological Conservation **85**:305-319.

Sahtu Land Use Planning Board. 2002. Sahtu preliminary draft land use plan. Sahtu Land Use Planning Board, Fort Good Hope, Northwest Territories.

Sanderson, E. W., J. Malanding, M. A. Levy, K. H. Redford, A. V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. Bioscience **52**:891-904.

Schneider, R. R., J. B. Stelfox, S. Boutin, and S. Wasel. 2003. Managing the cumulative impacts of land uses in the western canadian sedimentary basin: a modeling approach. Conservation Ecology **7**:8.

Schwartzman, S., A. Moreira, and D. Nepstad. 2000. Rethinking tropical forest conservation: perils in parks. Conservation Biology **14**:1351-1357.

Smith, R. J. 2004. Conservation Land-Use Zoning (CLUZ) software v.1.11. Durrell Institute of Conservation and Ecology, Canterbury, UK. Available from http:www.mosaic-conservation.
org/cluz/index.html (accessed February 2005).

Stadel, A., R. Taniton, and H. Heder. 2002. Northwest Territories protected areas strategy: how community values are shaping the protection of wild spaces and heritage places. Pages 20-26 in A. E. Watson, L. Alessa, and J. Sproull, editors. Wilderness in the circumpolar north: search for compatibility in ecological , traditional, and ecotourism values. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, Utah.

Taku River Tlingit First Nation. 2003. Our land is our future: Taku River Tlingit First Nation vision and management direction for land and resources. Taku River Tlingit First Nation, Taku River, British Columbia.

Venevsky, S., and I. Venevskaia. 2005. Hierarchical systematic conservation planning at the national level: identifying national biodiversity hotspots using abiotic factors in Russia. Biological Conservation **124**:235-251.

Warman, L. D., A. R. E. Sinclair, G. G. E. Scudder, B. Klinkenberg, and R. L. Pressey. 2004. Sensitivity of systematic reserve selection to decisions about scale, biological data, and targets: case study from southern British Columbia. Conservation Biology **18**:655-666.

Watson, A., L. Alessa, and B. Glaspell. 2003. The relationship between traditional ecological knowledge, evolving cultures, and wilderness protection in the Circumpolar north. Conservation Ecology **8**:2.

Wells, M. P., and T. O. McShane. 2004. Integrating protected area management with local needs and aspirations. Ambio **33**:513-519.

Wiersma, Y. F., and D. L. Urban. 2005. Beta diversity and nature reserve system design in the Yukon, Canada. Conservation Biology **19**:1262-1272.

Chapter 4

Minimum Dynamic Reserves: a theory for reserve size[3]

1. Introduction

Conservation planning currently lacks general methods for determining the size of reserves required to maintain ecological processes and, in particular, to buffer against natural disturbance (Peters *et al.* 1997; Poiani *et al.* 2000). Conventional reserve design methods draw from the theories of island biogeography (Mac Arthur & Wilson 1967) and meta-population dynamics (Levins 1969), and often incorporate habitat requirements of focal species and the spatial configuration of landscape features and special elements (Noss *et al.* 2002) to inform reserve size. However, these methods do not provide explicit guidance for the size of reserves required to capture dynamic features on a landscape (Pressey *et al.* 2003). Pickett & Thompson (1978) defined a minimum dynamic area (MDA) as "the smallest area with a natural disturbance regime, which maintains internal recolonization sources, and hence minimizes extinction". Patch dynamics influence vegetation communities and their dependent fauna (Margule and Pressey 2000). The conceptual linkage between habitat patch dynamics and population persistence is of relevance to reserve design (*e.g.*, Poiani *et al.* 2000; Groves *et al.* 2002; Pressey *et al.* 2003); however, MDAs remain a theoretical construct with little empirical support (Baker 1992; Fries *et al.* 1998; Bengtsson *et al.* 2003). The MDA concept focuses on the general design principles for self-sufficient reserves, but no explicit or quantitative criteria on how to construct a MDA have been established, although dynamic simulation models (Peters *et al.* 1997) and temporal reconstruction of patch mosaics using forest history data (Romme 1982; Baker 1989) have been proposed.

The MDA concept combines elements of island biogeography theory with patch dynamics to identify the size of reserve "islands" that will support a quasi-equilibrium landscape given a particular disturbance regime. A landscape in a quasi-equilibrium state is one where disturbances are frequent and relatively small-scale compared to the landscape area, resulting in fairly constant populations and processes over the whole area (Sprugel 1991). Minimum area concepts (Jaccard 1902; Dress 1954; Barkman 1989) and

[3] This chapter is formatted as an Ideas and Perspectives article for submission to *Ecology Letters* by Leroux, S.J., Schmiegelow, F.K.A., Lessard, R.B., Cumming, S.G.

the shifting-mosaic steady state theory (Bormann & Likens 1979; Heinselman 1981) present similar conceptual frameworks. The size of area required to exhibit quasi-equilibrium or MDA characteristics has been expressed as various multiples of maximum or mean disturbance size (see Cumming *et al.* 1996; Kneeshaw & Gauthier 2003). Although there is some evidence that landscapes can exist in a state of quasi-equilibrium (Bormann & Likens 1979; Shugart & West 1981; Sprugel & Borman 1981; Frelich & Lorimer 1991; Mueller-Dombois 1991; Busing & White 1993), other studies indicate an absence of this pattern (Romme 1982; Baker 1989; Sprugel 1991; Turner & Romme 1994; Cumming *et al.* 1996). Nevertheless, the MDA concept remains a frontier notion in conservation biology (Poiani *et al.* 2000). In the two and a half decades since Pickett and Thompson (1978) published their seminal work, only a few studies have attempted to estimate the area required for an actual MDA (Leck 1979; Peters *et al.* 1997; Peterken 1999; Kneeshaw & Gauthier 2003). It appears that the existence of landscape equilibrium and MDA conditions are the exception, and not the rule (White & Pickett 1985; Sprugel 1991; Cumming *et al.* 1996), which limits their application for reserve design. There is an imminent need for quantitative, yet practical criteria to guide reserve size for long-term maintenance of the world's remaining biodiversity because of increasing concerns regarding the state of global biomes (Foley *et al.* 2005; Hoekstra *et al.* 2005; Cardillo *et al.* 2006). By simplifying the restrictive conditions of the MDA concept and making criteria more explicit, it may be possible to identify practical approaches to the design of large reserves to buffer against natural disturbance and maintain ecological processes.

I introduce the minimum dynamic reserve (MDR) in an effort to operationalize the concept of MDAs, by providing practical and quantitative criteria for reserve size. I begin by defining the MDR concept and present a conceptual framework for the size of MDRs. I then describe a method to estimate the actual size and location of candidate MDRs and describe how to evaluate these candidate MDRs using spatial simulation models. Next, I apply this theory and simulation methods to estimate the size and location of a candidate MDR in a 115,000 km^2 study area in the Mackenzie Valley, Northwest Territories, Canada (Fig. 4-1), and test if a candidate MDR maintains its recolonization sources. In exploring the MDR concept, I hope to stimulate renewed discussion on practical principles for determining reserve size.

2. Definition and Scope of the Minimum Dynamic Reserve Concept

I define a MDR as the minimum area required to buffer against natural disturbance and maintain ecological processes. In contrast to an MDA; however, an MDR is not meant to achieve all conservation goals on its own. Rather, I intend that MDRs act as an anchor or benchmark within a reserve network that will capture the dynamic elements of a landscape. I assume that static elements (*e.g.*, geological features) are adequately planned for using conventional reserve design methods (*e.g.*, Noss *et al.* 2002).

A natural disturbance is "any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment" (White & Pickett 1985). Characteristic disturbance sizes are a consideration in reserve design because disturbances restructure the physical characteristics of landscapes and thus influence the species and processes that depend on these structures (Poiani *et al.* 2000; Groves *et al.* 2002). The MDR concept is similar to the MDA concept because both consider the size of disturbance and recolonization sources in their definition. These concepts differ in that MDA concept assumes that a quasi-equilibrium landscape is sufficient to guarantee the persistence of animal and plant species within an MDA. These criteria are too restrictive to achieve in practice (Baker 1992; Cumming *et al.* 1996; Fries *et al.* 1998; Bengtsson *et al.* 2003). The MDR concept is intended to resolve this by relaxing the condition of quasi-equilibrium and by providing criteria for reserve size based on the minimum representation required to maintain minimal areas of communities through time, but otherwise permits internal fluctuations in their abundance or extent. I define communities as suites of species from a single taxonomic class in different life stages that may be differentially affected by landscape disturbance in a region. The MDR concept is a refinement of the MDA concept that has the clear benefits of being explicit and quantitative, and offers practical applications for conservation planning.

After identifying the natural disturbance regime and communities used in a MDR analysis, there are three steps to identifying a MDR: 1) estimating the minimum size of a MDR based on the largest expected disturbance event, 2) estimating the actual size and location of a candidate MDR in light of the composition of species communities on a landscape, and 3) testing if a candidate MDR maintains its recolonization sources through time under natural disturbance.

2.1 Estimating the minimum size of a candidate MDR

I elaborate the MDR concept under the simplifying assumption that the natural disturbance regime is approximately spatially homogenous and temporally stationary (*i.e.* similar frequency and size of disturbances). In other words, I assume the relationship between disturbance probability and communities to be relatively homogeneous. In target regions that encompass multiple disturbance regimes, at least one candidate MDR should be identified for each disturbance regime. A candidate MDR must be contiguous, at least as large as the largest expected disturbance, and contain minimum proportional representation of all communities. These conditions are meant to ensure that the reserve has a high probability of maintaining recolonization sources. The minimum size (M) of a candidate MDR is given by:

$$M = \sum_{i=1}^{n} y_i \qquad (4\text{-}1)$$

where y_i are the relative proportional areas of the n communities defined as:

$$y_i = \frac{xa_i}{a_{\max}} \qquad (4\text{-}2)$$

where x is the extent of the largest expected disturbance, a_i is the area occupied by the *i-th* community, and a_{max} is the area occupied by the community with the largest total area.

Historical records of disturbance can be used to estimate x (Cumming 2001). The observed values of y_i depend on the state of the system at the time of observation. State-independent estimates of y_i and M can be obtained by Monte Carlo simulations using landscape models to estimate a mean a_i and a_{max} through time. For the candidate MDR to be successful in maintaining internal recolonization sources, the extent of a_i and $a_{max} \geq \delta_i$ at all times, where δ is the minimum area of community i required to maintain internal recolonization sources. Here, for simplicity, I assume that $\delta_i = 1$ unit area.

2.2 Estimating the actual size and location of a candidate MDR

M is the minimum size of a candidate MDR but it is unlikely that the composition of all communities on the landscape is such that a random MDR of size M is not guaranteed to satisfy the minimum area requirements for all communities. Consequently, we need to estimate the actual size and location of a candidate MDR given the configuration of communities on the landscape. The actual size is the size of the smallest area $\geq M$ that satisfies the minimum area requirements over time. The actual size depends on the spatial distribution of communities in the region and how these groups transition over time or after disturbance. This can be achieved in the following steps:

i) Partition the region into as many approximately equally sized planning units, P_i, of size at least M as possible. P_i can become larger than M (see Step iii).
ii) Determine which P_i satisfy all the minimum requirements of y_i.
iii) If P_I does not meet the minimum requirements of y_i, iteratively increase the size of P_i and calculate the area of each community for each P_i with minimum requirements of y_i, until a candidate MDR is found.

If a landscape does not have a P_i that meets the minimum requirements, this landscape does not have a candidate MDR. If multiple candidate MDRs are found, order the candidates according to ecological criteria. For example, a diversity index could be used to rank the candidate MDRs.

2.3 Testing if a candidate MDR maintains its recolonization sources

Spatially explicit, dynamic simulation models can be used to test whether a candidate MDR is likely to maintain its recolonization sources, by modelling the dominant disturbance type and its effect on communities on the landscape. To test if the candidate MDR maintains a_i and $a_{max} > 0$ at all times, Monte Carlo simulations of the disturbance dynamics can be used to calculate the amount of a_i and a_{max} in the candidate MDR through time. If at any point during the simulation a_i or $a_{max} = 0$ in the candidate MDR, the MDR is deemed to be ineffective with respect to maintaining potential recolonization sources. In this case, the process is re-iterated with larger P_is until a candidate MDR is found that satisfies a_i and $a_{max} > 0$ at all times or the maximum size is reached.

3. Forest Fire and Boreal Forest Recolonization: An Application of the Minimum Dynamic Reserve Concept

The dominant disturbance agent responsible for vegetation community development in most areas of the boreal region of Canada is forest fire (Johnson 1992). Consequently, the MDR for this region considered forest fire as the landscape disturbance and communities composed of different plant species and seral stages. There are five communities composed of different species and seral stages that may be differentially affected by fire in the study area: closed spruce, open spruce, mixed-wood, tall shrub, and low shrub forests (Johnson 1992) (Table 4-1). The data used for the analysis are from Ducks Unlimited's (2002; 2003; 2006) earth cover classification for the study area. I dissolved the original 37 vegetation classes at 30 m resolution into 10 similar communities and re-scaled these to 500 m (25 ha) grid cells using a majority threshold filter analysis. I only modelled changes of the 5 communities significantly affected by forest fire in the MDR analysis.

3.1 Estimating the minimum size of a candidate MDR

The study area, although very large, falls within a relatively homogenous fire region with similar burn rates and fire sizes (Cumming, Mackey & Schmiegelow unpublished data); therefore I searched for one candidate MDR on the landscape. To estimate M I determined the largest expected fire, 2,370 km^2, from methods in Cumming (2001). To provide a robust estimate of a_i and a_{max} for eq. (2), I ran 100 Monte Carlo simulations of 250 yrs using CONSERV, a 25 ha grid-based, landscape simulation model that uses the fire model and user interface of an earlier model developed for boreal landscapes (Armstrong & Cumming 2003). I subsequently estimated y_i, and M using these data. At each time step (1 yr), CONSERV ignites and spreads fires, and vegetation community succession occurs.

3.1.1 CONSERV - Parameters

I reconstructed the initial age of forest stands using time-since the last stand initiating fire for patches that had fire history records. I generated random fires on forest patches that did not have fire history data, assigning an age to each fire by drawing from a negative exponential age-class distribution (van Wagner 1978) with a fire cycle of 100 yrs. CONSERV simulates vegetation community dynamics based on time since fire. Black & Bliss (1978) described 4 stages of succession following fire in the spruce forests in the study area. After fire, a low shrub community develops. This community succeeds to a tall shrub community at approximately 21 yrs, followed by an open spruce community from 36 to 200 yrs. Closed spruce is the final seral stage from 201 to 300 yrs but some evidence suggests that closed spruce forests may senesce without fire and return to a low shrub-like vegetation (Strang & Johnson 1981). Consequently, in CONSERV, when cells reach 301 yrs without burning, they succeed to low shrub vegetation, as if burned. Black & Bliss (1978) did not study succession in mixed-wood forests of the study area. I assumed that areas of mixed-wood forest that burned succeeded through low shrub and tall shrub stages and returned to a mixed-wood forest in 36 yrs. I also assumed that if a herbaceous patch burned, it returned to a herbaceous patch 3 yrs after the fire. I did not use herbaceous in the MDR analysis as it is a rare vegetation type on the landscape and it has a very low fire frequency.

CONSERV models forest fire as a stochastic process where fires ignite, escape, and spread. I used empirical forest fire history data to parameterize CONSERV's forest fire model, applying methods described in Armstrong and Cumming (2003). The ignition rate was 8.9 x 10^{-5} /cell/yr and the escape probability was 0.04, resulting in approximately 10 fires/yr larger than 25 ha. I estimated the spread rate iteratively until I had parameters that produced a mean fire size similar to approximate the historical mean fire size. The best parameter estimates produced a mean fire size of 10,780 ha (n = 4,434). The mean fire size of empirical data was 10,761 ha (n = 351). I compared the simulated mean to the mean fire size of the empirical data by bootstrapping 100 samples from each dataset, 100 times, and performing a Kolmogrov-Smirnov test on each bootstrap run. The mean p-value of these 100 tests was 0.395, and 12/100 tests had $p < 0.05$, indicating that the spread parameters produced a similar fire size distribution as the empirical fire size distribution.

3.2 Estimating the actual size and location of a candidate MDR

I implemented the steps for estimating the actual size and location of a candidate MDR into a reserve builder in CONSERV. I constructed a regular grid of planning units, P_i, beginning with a size of M. At each iteration, the first P_i was drawn from the top-left corner of the study area. Subsequent P_is were tiled afterwards. If no candidate MDR was identified, I increased the size of P_i and repeated the procedure. I used the Brillouin index of diversity (B_i) (Brillouin 1962) to rank multiple candidate MDRs. The Brillouin index (B_i) takes the form:

$$B_i = \frac{1}{Z_k^s} \log\left(\frac{Z_k^s!}{z_1!z_2!\ldots z_n!}\right) \qquad (4\text{-}3)$$

where Z is the sum of the occurrences of the communities in the *k-th* planning unit of size s, z_i is the number of occurrences of the *i-th* community, and n is the total number of communities. For the case study, an occurrence is equivalent to one 25 ha grid cell. The B_i measures the diversity of each P_i and it rewards evenness in community richness. For example, a study region with 10 occurrences of three communities will have a $B_i = 0.425$, whereas a region with 5 occurrences of two communities and 20 occurrences of the third community will have a lower $B_i = 0.330$.

3.3 Testing if a candidate MDR maintains its recolonization sources

I tracked the area of each community at each time step in a candidate MDR subjected to an active disturbance regime. If at any point during the simulation a_i or $a_{max} = 0$ in the candidate MDR, the candidate MDR was deemed to be ineffective at maintaining recolonization sources. I ran 100 Monte Carlo simulations of 250 years to test the effectiveness of candidate reserves.

4. Results

4.1 Estimating the minimum size of a candidate MDR

After 100 CONSERV simulations of 250 years, the mean extent of the five communities available to burn in the study area varied between a minimum of 2,567 km^2 for mixed-wood to 55,800 km^2 for the dominant community, open spruce (Table 4-2).

Based on the estimates of the mean a_i and a_{max} and given $x = 2{,}370$ km^2 I estimated y_i from eq. (2) to be $y_1 = 413$ km^2, $y_2 = 2{,}370$ km^2, $y_3 = 109$ km^2, $y_4 = 529$ km^2, and $y_5 = 748$ km^2 (Table 4-2). Using results for x and y_i and eq. (1), I estimated $M = 4{,}169$ km^2. M is the theoretical minimum size of a candidate MDR in the study area. I estimated the actual size and location of a candidate MDR based on the spatial distribution of communities on the landscape.

4.2 Estimating the actual size and location of a MDR

Beginning with $P_1 = M$, I iteratively increased the size of P_i until I identified a P_i of a size and location that met all minimum requirements of y_i. I found one candidate MDR, which was 6,481 km^2 and was located in the east-central portion of the study area (Fig. 4-2). Because I only found one candidate MDR of this size, I did not use the B_i to rank the candidate MDRs.

4.3 Testing if a candidate MDR maintains its recolonization sources

After 100 CONSERV simulations of 250 yrs, a_i and $a_{max} > 0$ at all times; therefore, I concluded that the candidate MDR was large enough to maintain its potential recolonization sources through time (Table 4-3). The minimum extent of any a_i during the simulations was 10 km^2 of tall shrub. Applying the conceptual framework provided practical support of a MDR.

5. Discussion

The MDR concept provides a refined framework for determining the minimum size of reserve required to buffer against landscape disturbance and maintain ecological processes. The conceptual framework is based on the size of natural disturbance and the relationship between natural disturbance and communities. The communities are composed of different species from a single taxonomic class at varying life history stages that may be differentially affected by natural disturbance. The MDR concept incorporates elements of the MDA concept to provide practical criteria for guiding reserve size. The strengths of the MDR concept are its explicit and quantitative criteria that can be applied to real ecosystems. While the size of a potential MDR may seem daunting for implementation, in Canada alone, there are 10 National Parks larger than the MDR identified through the case study and there have been calls for mega-reserves in other parts of the World (Peres 2005). The large reserves in Canada; however, were not designed with explicit criteria for buffering against the dominant disturbance in their system or maintaining internal recolonization sources, nor are they representative of land cover.

To illustrate the conceptual framework, I provided an application of the MDR concept in the boreal region of Canada. In this case, forest fire was considered the dominant natural disturbance affecting communities composed of different plant species. I estimated the actual size and location of a candidate MDR in the study area and used CONSERV to simulate landscape change and track the amount of each community over time. The candidate MDR maintained its potential recolonization sources through time. The ability of MDRs to capture other ecological processes, such as predator-prey dynamics, nutrient flow, and productivity that may be related to recolonization potential of boreal forest communities requires careful evaluation.

Single reserves considered in isolation, even very large ones such as MDRs, are unlikely to ensure the long-term protection of biodiversity and the ecological processes upon which biodiversity depends (Gove *et al.* 2005; Scholes & Biggs 2005). Areas adjacent to reserves can make significant contributions towards achieving conservation goals even if adjacent areas are relatively small (Gove *et al.* 2005). A MDR, however, can act as an anchor for a comprehensive conservation network. Combined with smaller reserves and sustainable management of intervening areas, a MDR can contribute to an overall conservation network that will buffer against natural disturbance, capture focal species and their habitat, and maintain environmental representation, special elements, and ecological processes. Unifying these goals is the most effective method to achieve comprehensive conservation planning.

5.1 Limitations

The methodology I described and applied requires data sets that span large spatial extents. I was fortunate to have high quality earth cover classification data and fire history data. Other regions may not possess such information, although global data sets are increasingly available (*e.g.*, MODIS, Global Landcover 2000). I also assumed that a single occurrence of a community was sufficient to maintain the potential for recolonization. In the case study, one occurrence of a community encompassed 25 ha. This may be sufficient to maintain recolonization of boreal forest plant communities but other systems might require a larger amount of each group for recolonization potential to be maintained (*e.g.*, for area-demanding vertebrate species). Conversely, some communities might maintain their recolonization potential even when they are not present on the landscape (*e.g.*, semi-serotinous cones of some coniferous trees). I suggest identifying one candidate MDR for each homogenous disturbance regime. I offer this as a guideline but it may not be appropriate in all regions. Local ecologists are still best positioned to determine the number of candidate MDRs required in any given region.

To evaluate a candidate MDR, I suggested the use of simulation models. Although spatially explicit, dynamic simulation models are increasingly available (*e.g.*, LANDIS, SELES, TARDIS), most of these have not been applied to reserve design questions. Where regionally appropriate simulation models are not available, historical disturbance records can be used to evaluate candidate MDRs. To determine the effectiveness of the candidate MDR I ran 100 CONSERV simulations of 250 years. I could provide a higher probability that the test results are representative of the effectiveness of our candidate MDR by increasing the number of simulations. The approach to identifying a candidate MDR using P_is that were tiled may have influenced the results, thus alternative methods, such as applying a moving window, should be explored. Finally, while I have provided criteria for identifying the size and location of MDRs, I have not provided criteria for the configuration of a MDR. Ideally, the perimeter of a MDR should follow the shape of natural features like watersheds and vegetation boundaries, to maintain natural connectivity.

5.2 Future directions

A priority for future work on the MDR concept is to test the model for generality in other systems with different disturbance regimes. The MDR concept may be particularly useful in forest ecosystems, but I believe it may also be applied to other terrestrial and aquatic systems. Further, natural disturbances do not act in isolation on landscapes; therefore, it would be interesting to investigate inclusion of multiple disturbances and their interactions into the MDR concept. Similarly, for simplicity, I provided a conceptual framework for a single taxonomic class, but I believe the MDR concept could be expanded to look at multiple taxonomic classes and their interactions. For example, forest fire plays a key role in the lives of many mammal (*e.g.*, moose) and bird (*e.g.*, woodpeckers) species. The spatial requirements of these species could be included in future analyses of MDRs in the boreal forest of Canada.

The framework provides criteria for identifying one large MDR but it would useful to investigate if several smaller reserves could maintain the minimum requirements of a MDR. A series of small reserves may be effective in regions that have already been significantly transformed, where large tracts of contiguous land are not available. The MDR concept could also be used as a gap analysis tool to evaluate the effectiveness of existing reserves to buffer against natural disturbance and maintain ecological processes. Ecological benchmarks can provide reference sites for the detection of change in ecosystems and for understanding ecosystem processes and the natural range of variation of biodiversity and ecosystems (Arcese & Sinclair 1997). The MDR concept could be used to identify system-level ecological benchmarks large enough to capture large natural disturbances. The MDR concept could also be integrated with current dynamic reserve design theory (Carroll *et al.* 2003; Pyke & Fischer 2005; Halpren *et al.* 2006) and the process of systematic conservation planning (Margules & Pressey 2000). Finally, empirical tests of the effectiveness of the MDR concept are required to refine the concept. To do so, we need to implement MDRs and monitor communities over time. Retrospective approaches using data from existing reserves could also be pursued. Without this empirical substantiation, the MDR concept is likely to languish as a theoretical construct.

6. Conclusion

I believe that the MDR concept may be particularly useful in the world's remaining wilderness areas that are still shaped by large natural disturbances. The Amazon region, and boreal regions of Siberia and Canada contain most of the world's remaining forests and are future battlegrounds for conservation (Mittermeier *et al.* 2003; Cardillo *et al.* 2006); however, only small fractions of these regions have permanent protection. The MDR concept could be used to identify mega-reserves (*e.g.*, Peres 2005) that would act as anchors of a comprehensive reserve network in the World's remaining intact areas. Other disturbance regimes and systems that are strong candidates for applying the MDR concept include but are not limited to: 1) tropical storms and coral reefs, 2) treefalls in tropical forests, 3) climate change in arctic regions, 4) windstorms in sand dune communities, 5) insect outbreaks in temperate and boreal forests, 6) disease in African ungulate herds, and 7) large wave action on rocky inter-tidal communities. I believe that the MDR concept is a promising approach towards developing a general theory for reserve size. While still in relatively early stages of development, I hope that this paper will stimulate further discussions to help refine related concepts.

Figure 4-1 Study area in the northern boreal region of Canada shown as black polygon. The study area is 115,000 km^2 and is mostly in the Northwest Territories (NT) with parts in the Yukon Territory (YT).

Table 4-1 Description of the five plant communities used in the analysis of a candidate MDR in the study area. The groups are composed of different species and seral stages, and may be differentially affected by forest fire.

Community	Description*,†	Fire Frequency‡
Closed spruce	Dense needleleaf forests, mostly composed of white spruce (*Picea glauca*), black spruce (*Picea mariana*), and tamarack (*Larix lariciana*) with sparse under story vegetation including some low shrubs and lichens.	High
Open spruce	Open needleleaf forests, mostly composed of white spruce (*Picea glauca*), black spruce (*Picea mariana*), and tamarack (*Larix lariciana*) with dense under story vegetation including shrubs, forbs, grasses, sedges, horsetails, mosses, and lichens.	High
Mixed wood	Deciduous forest dominated by paper birch (*Betula papyrifera*), aspen (*Populus tremuloids*), and balsam poplar (*Populus balsimefera*) with sparse under story vegetation of low shrubs.	Moderate
Tall shrub	Tall shrubs dominated by willow sp. (*Salix sp.*), dwarf birch (*Betula pumila*), and alder (*Alnus crispa*) with sparse under story vegetation.	Low
Low shrub	Low shrubs dominated by blueberry sp. (*Vaccinium sp.*), labrador tea (*Ledum groenlandicum.*), moss, dwarf willow sp. (*Salix sp.*), dwarf birch (*Betula pumila*), dwarf alder (*Alnus crispa*), forbs, and graminoids.	Low

* Ducks Unlimited (2002, 2003, 2006)
† Cody (2002) for species authority
‡ Johnson (1992)

Table 4-2 Mean and standard deviation of the extent of five plant communities in the study area obtained from 100 simulation runs of 250 years using the spatially explicit, dynamic simulation model, CONSERV. Based on these estimates of a_i and a_{max}, I calculated the parameters for eqs. (1) and (2). I subsequently used these parameters to estimate the minimum size (M) of a candidate MDR.

Community*	Mean (km^2)	$\sigma^{\dagger}$ (km^2)	y_i (km^2)
1 Closed spruce	9,726	3,007	413
2 Open spruce	55,800	3,886	2,370
3 Mixed-wood	2,567	216	109
4 Tall shrub	12,457	2,247	529
5 Low shrub	17,612	2,354	748

* Integer denotes the community's index i

† Standard deviation of patch areas

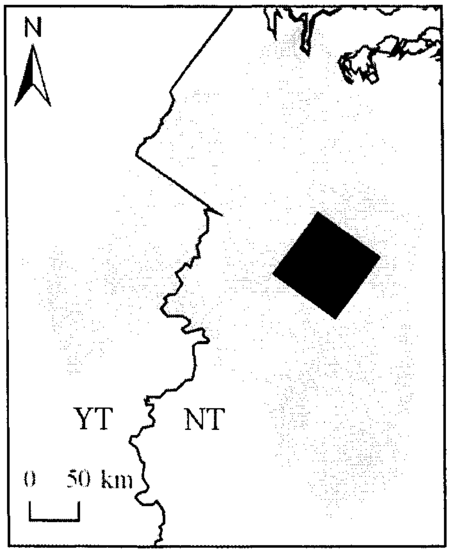

Figure 4-2 Candidate MDR in black overlaid on a grey polygon of the study area. This candidate MDR is 6,480 km^2. The candidate MDR was estimate using an iterative approach beginning with planning units of size *M* (eq. 4-1), until I found a candidate MDR of a size and location that met all the minimum requirements of eqs. (4-1) and (4-2). This candidate MDR represents a hypothetical reserve for this study area.

Table 4-3 Results of the 100 simulation runs of 250 years to determine if the candidate MDR met a_i and $a_{max} > 0$ at all times throughout the simulations. The minimum estimates of each a_i and $a_{max} > 0$, suggesting that the MDR is a suitable one. I also provide the mean and standard deviation for each a_i and a_{max}.

Community*	Minimum (km^2)	Mean† (km^2)	σ‡ (km^2)
1 Closed spruce	95	462	215
2 Open spruce	1,948	3,407	476
3 Mixed-wood	54	118	15
4 Tall shrub	10	705	346
5 Low shrub	40	947	407

* Integer denotes the community's index i

† Mean of total patch areas a_i

‡ Standard deviation of total patch areas a_i

7. Literature Cited

Arcese, P. & Sinclair, A.R.E. (1997). The role of protected areas as ecological baselines. *J. Wildlife Manage.*, 61, 587-602.

Armstrong, G.W. & Cumming, S.G. (2003). Estimating the cost of land base changes due to wildfire using shadow prices. *For. Sci.*, 49, 719-730.

Baker, W.L. (1992). The landscape ecology of large disturbances in the design and management of nature reserves. *Landscape Ecol.*, 7, 181-194.

Baker, W.L. (1989). Landscape ecology and nature reserve design in the boundary waters canoe area, Minnesota. *Ecology,* 70, 23-35.

Barkman, J.J. (1989). A critical evaluation of minimum area concepts. *Vegetatio,* 85, 89-104.

Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., Moberg, F. & Nystrom, M. (2003). Reserves, resilience and dynamic landscapes. *Ambio,* 32, 389-396.

Black, R.A. & Bliss, L.C. (1978). Recovery sequence of *Picea mariana - Vaccinium uliginosum* forests after burning near Inuvik, Northwest Territories, Canada. *Can. J. Botany,* 56, 2020-2030.

Bormann, F.H. & Likens, G.E. (1979). Catastrophic disturbance and the steady state in northern hardwood forests. *Am. Sci.,* 67, 660-669.

Brillouin, L. (1962). *Science and Information Theory.* Academy Press, New York.

Busing, R.T. & White, P.S. (1993). Effects of area on old-growth forest attributes: implications for the equilibrium landscape concept. *Landscape Ecol.,* 8, 119-126.

Cardillo, M., Mace, G.M., Gittleman, J.L. & Purvis, A. (2006). Latent extinction risk and the future battlegrounds of mammal conservation. *Proc. Natl. Acad. Sci. USA*, 103, 4157-4161.

Carroll, C., Noss, R.F., Paquet, P.C. & Schumaker, N.H. (2003). Use of population viability analysis and reserve selection algorithms in regional conservation plans. *Ecol. Appl.,* 13, 1773-1789.

Cody, J.C. (2000). *Flora of the Yukon Territory.* 2nd edn. NRC Research Press, Ottawa, Ontario.

Cumming, S.G. (2001). A parametric model of fire-size distribution. *Can. J. Forest Res.*, 31, 1297-1303.

Cumming, S.G. (2004). *TARDIS User's Guide*. Boreal Ecosystems Research, Ltd., Edmonton, Alberta.

Cumming, S., Demarchi, D. & Walters, C. (1998). *A Grid-Based Spatial Model of Forest Dynamics Applied to the Boreal Mixedwood Region.* Sustainable Forest Management Network, Edmonton, Alberta.

Cumming, S.G., Burton, P.J. & Klinkenberg, B. (1996). Boreal mixedwood forests may have no "representative" areas: some implications for reserve design. *Ecography,* 19, 162-180.

Drees, E.M. (1954). The minimum area in tropical rain forest with special reference to some types in bangka (Indonesia). *Vegetatio,* 5/6, 517-523.

Ducks Unlimited I. (2002). *Lower Mackenzie River Delta, NT, Earth Cover Classification User's Guide.* Ducks Unlimited, Inc., Rancho Cordova, California.

Ducks Unlimited I. (2003). *Peel Plateau Project Earth Cover's User's Guide.* Ducks Unlimited, Inc., Rancho Cordova, California.

Ducks Unlimited I. (2006). *Middle Mackenzie Project Earth Cover Classification.* Ducks Unlimited, Inc., Rancho Cordova, California.

Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., & Snyder, P.K. (2005). Global consequences of land use. *Science,* 309, 570-574.

Frelich, L.E. & Lorimer, C.G. (1991). Natural disturbance regimes in hemlock-hardwood forests of the upper great lakes region. *Ecol. Monogr.,* 61, 145-164.

Fries, C., Carlsson, M., Dahlin, B., Lamas, T. & Sallnas, O. (1998). A review of conceptual landscape planning models for multiobjective forestry in Sweden. *Can. J. Forest Res.,* 28, 159-167.

Gove, A.D., Majer, J.D. & Rico-Gray, V. (2005). Methods for conservation outside of formal reserve systems: the case of ants in the seasonally dry tropics of Veracruz, Mexico. *Biol. Conserv.,* 126, 328-338.

Groves, C.R., Jensen, D.B., Valutis, L.L., Redford, K.H., Shaffer, M.L., Scott, M., Baumgartner, J.V., Higgins, J.V., Beck, M.W. & Anderson, M.G. (2002). Planning for biodiversity conservation: putting conservation science into practice. *Bioscience*, 52, 499-512.

Halpren, B., Regan, H.M., Possingham, H.P. & McCarthy, M. A. (2006). Accounting for uncertainty in marine reserve design. *Ecol. Lett.,* 9, 2-11.

Heinselman, M.L. (1981). Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In: *Forest Succession: Concepts and Application* (eds. West, D.C., Shugart, H.H. & Botkin, D.B.). Springer-Verlag, New York, pp. 374-405.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H. & Roberts, C. (2005). Confronting the biome crisis: global disparities of habitat loss and protection. *Ecol. Lett.,* 8, 23-29.

Jaccard, P. (1902). Lois de distribution florale dans la zone alpine. *Bull. Soc. Vaud. Sci. Nat.,* 38, 6-130.

Johnson, E.A. (1992). *Fire and Vegetation Dynamics: Studies from the North American Boreal Forest.* Cambridge University Press, New York.

Kneeshaw, D. & Gauthier, S. (2003). Old growth in the boreal forest: a dynamic perspective at the stand and landscape level. *Environ. Rev.,* 11, S99-S114.

Leck, C.F. (1979). Avian extinctions in an isolated tropical wet-forest preserve, Ecuador. *Auk,* 96, 343-352.

Levins, R. (1969). Some demographic consequences of environmental heterogeneity for biological control. *Bull. Ent. Soc. Amer.*, 15, 237-240.

Mac Arthur, R.H. & Wilson, E.O. (1967). *The Theory of Island Biogeography.* Princeton University Press, Princeton, New Jersey.

Margules, C.R. & Pressey, R.L. (2000). Systematic conservation planning. *Nature*, 405, 243-253.

Mittermeier, R.A., Mittermeier, C.G., Brooks, T.M., Pilgrim, J.D., Konstant, W.R., daFonseca, G.A.B. & Kormos, C. (2003). Wilderness and biodiversity conservation. *Proc. Natl. Acad. Sci. USA*, 100, 10309-10313.

Mueller-Dombois, D. (1991). The mosaic theory and the spatial dynamics of natural dieback and regeneration in pacific forests. In: *The Mosaic-Cycle Concept of Ecosystems* (ed. Remmert, H.). Springer-Verlag, New York, pp. 46-60.

Noss, R.F. & Cooperrider, A.Y. (1994). *Saving Nature's Legacy.* Island Press, Washington D.C.

Noss, R.F., Carroll, C., Vance-Borland, K. & Wuerthner, G. (2002). A multicriteria assessment of the irreplaceability and vulnerability of sites in the greater Yellowstone ecosystem. *Conserv. Biol.,* 16, 895-909.

Peres, C.A. (2005). Why we need megareserves in Amazonia. *Conserv. Biol.,* 19, 728-733.

Peterken, G.F. (1999). Applying natural forestry concepts in an intensively managed landscape. *Global Ecol. Biogeogr.,* 8, 321-328.

Peters, R.S., Waller, D.M., Noon, B., Pickett, S.T.A., Murphy, D., Cracraft, J., Kiester, R., Kuhlmann, W., Houck, O. & Snape III, W.J. (1997). Standard scientific procedures for implementing ecosystem management on public lands. In: *The Ecological Basis of Conservation: Heterogeneity, Ecosystems, and Biodiversity* (eds. Pickett, S.T.A., Ostfeld, R.S., Shachak, M. & Likens, G.E.). Chapman & Hall, Toronto, pp. 320-336.

Pickett, S.T.A. & Thompson, J.N. (1978). Patch dynamics and the design of nature reserves. *Biol. Conserv.,* 13, 27-37.

Poiani, K.A., Richter, B.D., Anderson, M.G. & Richter, H.E. (2000). Biodiversity conservation at multiple scales: functional sites, landscapes, and networks. *Bioscience*, 50, 133-146.

Pressey, R.L., Cowling, R.M. & Rouget, M. (2003). Formulating conservation targets for biodiversity pattern and process in the Cape Florisitic Region, South Africa. *Biol. Conserv.,*112, 99-127.

Pyke, C.R. & Fischer, D.T. (2005). Selection of bioclimatically representative biological reserve systems under climate change. *Biol. Conserv.,* 121, 429-441.

Romme, W.H. (1982). Fire and landscape diversity in subalpine forests of Yellowstone National Park. *Ecol. Monogr.,* 52, 199-221.

Scholes, R.J. & Biggs, R. (2005). A biodiversity intactness index. *Nature*, 434, 45-49.

Shugart, H.H. & West, D.C. (1981). Long-term dynamics of forest ecosystems. *Am. Sci.,* 69, 647-652.

Sprugel, D.G. (1991). Disturbance, equilibrium, and environmental variability: what is 'natural' vegetation in a changing environment? *Biol. Conserv.,* 58, 1-18.

Sprugel, D.G. & Bormann, F.H. (1981). Natural disturbance and the steady state in high-altitude balsam fir forests. *Science,* 211, 390-393.

Strang, R.M. & Johnson, A.H. (1981). Fire and climax spruce forests in central Yukon. *Arctic,* 34, 60-61.

Turner, M.G. & Romme, W.H. (1994). Landscape dynamics in crown fire ecosystems. *Landscape Ecol.,* 9, 59-77.

Van Wagner, C.E. (1978). Age-class distribution and the forest fire cycle. *Can. J. Forest Res.,* 8, 220-227.

White, P.S. & Pickett, S.T.A. (1985). Natural disturbance and patch dynamics: an introduction. In: *The Ecology of Natural Disturbance and Patch Dynamics* (eds. Pickett, S.T.A. & White, P.S.). Academic Press, Inc, Orlando, Florida, pp. 3-13.

Chapter 5

1. Thesis Synopsis

Twelve percent of the world's surface is included in protected areas (Rodriguez et al. 2004). Nevertheless, many of the world's ecosystems are under threat from anthropogenic pressures such as resource extraction, agriculture, urban sprawl, and tourism (Foley et al. 2005). Many of these activities result in land conversion which is altering the state of global biomes (Foley et al. 2005; Hoekstra et al. 2005). With the advent of systematic methods for identifying protected area networks, and the incorporation of population dynamics (Carroll et al. 2003), landscape change (Pressey et al. 2004; Pyke and Fischer 2005), uncertainty modelling (Halpren et al. 2006) and dynamic decision-making (Meir et al. 2004; Oetting et al. 2006; Wilson et al. 2006) into protected areas design, conservation planners around the world are designing more effective protected areas. In this thesis, I investigated protected areas design in a relatively intact ecosystem: the boreal region of Canada. I had three goals: 1) evaluate the efficacy of protected areas designed based on conventional conservation planning methods, 2) investigate the complementarity of indigenous heritage sites and protected areas, and 3) enhance the theoretical and practical foundations for determining the size of protected areas required to buffer against landscape disturbance and maintain ecological processes.

In Chapter 2, I developed a spatially explicit, dynamic simulation model, CONSERV, that enabled me to evaluate the ability of protected areas networks designed to capture conventional conservation targets to maintain their initial targets through time under a natural disturbance regime. None of the candidate protected area networks maintained all of their targets through time, highlighting the importance of considering system dynamics a priori in protected areas design. Simulation models can be effective tools for setting robust conservation targets and for incorporating natural disturbance into protected areas design. These models may also provide a unifying framework for incorporating other systems dynamics into protected areas design (e.g., population dynamics; Carroll et al. 2003, climate change modelling; Pyke and Fischer 2005, uncertainty modelling; Halpren et al. 2006) and dynamic decision-making (Meir et al. 2004; Oetting et al. 2006; Wilson et al. 2006). I argue that spatially explicit, dynamic simulation models should become an indispensable tool in the conservation planner's toolbox.

In Chapter 3, I compared the spatial overlap between Gwich'in heritage sites and protected areas designed to capture conventional conservation targets, including focal species habitat, high-quality wetlands, representation of vegetation types and waterbodies, and environmentally significant area. I expected high spatial overlap between the two because several authors suggest that there is a positive relationship between heritage sites, biodiversity, and supporting ecological processes (Alcorn 1993; Oveido & Brown 1999; Garibaldi & Turner 2004). Instead, I observed low spatial overlap between the Gwich'in heritage sites and the protected areas designed to capture conventional conservation targets. Additional analyses suggested the prominence of waterbodies in Gwich'in heritage sites may have driven these results. In general, my results highlight that protected areas planning should explicitly consider indigenous heritage sites in order to adequately capture these features. Likewise, indigenous communities that want to protect cultural and natural features could use conventional conservation planning methods to identify additional areas for protection to complement their heritage sites. Integration of both planning methods would provide more comprehensive conservation plans on indigenous lands.

In Chapter 3, I proposed the minimum dynamic reserve concept; a theory for the size of reserve required to buffer against natural disturbance and maintain ecological processes. I proposed this conceptual framework as a refinement of the minimum dynamic area concept (sensu Pickett and Thompson 1978). The strengths of the MDR concept are its quantitative, explicit, and practical criteria for identifying large protected areas to act as anchors of a comprehensive protected areas network. I developed a conceptual framework for identifying MDRs on any landscape under the influence of natural disturbance. I tested this framework with an application in the Mackenzie valley, Northwest Territories and found a successful MDR that was sufficiently large to buffer against landscape disturbance and maintain its recolonization sources through time. I advance the MDR concept as a step towards developing a comprehensive theory for protected areas size.

2. Implications for Systematic Conservation Planning

The results of this thesis provide three key implications for systematic conservation planning (sensu Margules and Pressey 2000). First, I have demonstrated that natural disturbance can influence the efficacy of protected areas and that spatially explicit, dynamic simulation models can be useful tools in evaluating the effectiveness of candidate protected areas and in setting effective conservation targets. Along with other studies that have begun to incorporate system dynamics in protected areas design (Carroll et al. 20003; Pressey et al. 2004; Pyke and Fischer 2005; Halpern et al. 2006), my results suggest the need to incorporate an additional stage within systematic conservation planning where candidate protected areas are evaluated before they are implemented. Second, I have provided evidence that heritage sites may not overlap with protected areas designed to capture conventional conservation targets. Consequently, protected areas design that occurs on indigenous lands should explicitly incorporate indigenous heritage sites into their design criteria to effectively capture these cultural features in comprehensive protected area networks. Finally, I have proposed a conceptual framework that can be used to identify large protected areas that can act as anchors of protected area networks. The MDR concept combined with systematic conservation planning methods can be used to identify comprehensive protected area networks that capture a range of conservation features and are robust to anticipated natural disturbances.

3. Limitations and Future Directions

Two chapters of my thesis use a spatially explicit, dynamic simulation model. These models require considerable amounts of data for parameterization. I was fortunate to have access to such data but data availability is certainly a limitation when applying these models to other landscapes. The advantage of using these models is that it enables conservation planners to evaluate protected areas under different landscape scenarios. Throughout my thesis, I used case studies from the Mackenzie Valley, Northwest Territories. Further evidence from other study areas is necessary to substantiate and refine the general implications of my three data chapters.

Protected areas design is a complex task and while I provide recommendations based on my work, my thesis has been theoretical with little consideration for the applicability of these results on the ground. In a perfect world, I would have been able to work more closely with people undertaking protected areas design and implementation in my study area, but the requirements of a graduate program do not facilitate this sort of exchange. I have discussed the general objectives of my work with the Secretariat of the Government of the Northwest Territories Protected Areas Strategy and the Gwich'in Social Cultural Institute and hope to work with them to transmit the essential findings of my thesis into information that may be useful for their protected areas policy and implementation plan.

The results of my thesis have raised more questions than answers. For Chapter 2, the primary future direction would be to integrate population dynamics modelling (Carroll et al. 2003), climate change modelling (Pyke and Fischer 2005), uncertainty modelling (Halpren et al. 2006), and dynamic decision-making (Meir et al. 2004; Oetting et al. 2006) with spatial explicit, natural disturbance modelling to provide a comprehensive tool for protected areas evaluation and to develop a general theory for protected areas design in a dynamic and uncertain world. For Chapter 3, future directions involve testing the relationship between indigenous heritage sites and protected areas in other regions as well as developing explicit criteria for incorporating indigenous heritage sites into protected areas design early in the planning process. For Chapter 4, I have only taken a small step towards developing a general theory for reserve size. Future work should test the MDR concept in other landscapes with different disturbance regimes, and should work to refine the theory by incorporating multiple taxa and interacting disturbances.

4. Final Thoughts

Although protected areas may act as anchors for conservation, alone, they will not be enough to curb the current biodiversity crisis (e.g. Hoekstra et al. 2005; Soares-Filho et al. 2006). To ensure the survival of the world's biodiversity and its supporting ecological processes, we must adopt a comprehensive conservation planning approach such as that embodied in the reverse-matrix model (sensu Schmiegelow et al. 2005). The reverse-matrix model combines ecological benchmarks, additional protected areas, and sustainable resource management within an adaptive management framework to address comprehensive conservation planning. We will make considerable progress towards achieving conservation goals if we pursue such an approach. In the boreal region of Canada, there is a willingness from industry, First Nations, and non-governmental organizations to work towards a reverse-matrix model for conservation planning. Given the vast extent and relatively intact nature of these systems, we still have the opportunity to design effective conservation plans. Both national (Senate Sub-Committee on the Boreal Forest 1999; National Round Table on the Environment and the Economy 2005) and international organizations (IUCN 2005) have called on the Canadian Government to protected its boreal region; let us not wait until it is too late.

5. Literature Cited

Alcorn, J. B. 1993. Indigenous peoples and conservation. Conservation Biology **7**:424-426.

Carroll, C., R. F. Noss, P. C. Paquet, and N. H. Schumaker. 2003. Use of population viability analysis and reserve selection algorithms in regional conservation plans. Ecological Applications **13**:1773-1789.

Foley, J. A., R. DeFries, G. P. Asner, C. Barford, G. Bonan, S. R. Carpenter, F. S. Chapin, M. T. Coe, G. C. Daily, H. K. Gibbs, J. H. Helkowski, T. Holloway, E. A. Howard, C. J. Kucharik, C. Monfreda, J. A. Patz, I. C. Prentice, N. Ramankutty, and P. K. Snyder. 2005. Global consequences of land use. Science **309**:570-574.

Garibaldi, A., and N. Turner. 2004. Cultural keystone species: Implications for ecological conservation and restoration. Ecology and Society **9**:1.

Halpren, B., H. M. Regan, H. P. Possingham, and M. A. McCarthy. 2006. Accounting for uncertainty in marine reserve design. Ecology Letters **9**:2-11.

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, and C. Roberts. 2005. Confronting the biome crisis: Global disparities of habitat loss and protection. Ecology Letters **8**:23-29.

IUCN. 2005. Resolutions and Recommendations. IUCN, Gland, Switzerland and Cambridge, United Kingdom.

Margules, C. R., and R. L. Pressey. 2000. Systematic conservation planning. Nature **405**:243-253.

Meir, E., S. Andelman, and H. P. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7**:615-622.

National Round Table on the Environment and the Economy. 2005. Boreal futures: governance, conservation and development in Canada's boreal. National Round Table on the Environment and the Economy, Ottawa, Ontario.

Oetting, J. B., A. L. Knight, and G. R. Knight. 2006. Systematic reserve design as a dynamic process: F-TRAC and the Florida Forever program. Biological Conservation **128**:37-46.

Oveido, G., and J. Brown. 1999. Building alliances with indigenous peples to establish and manage protected areas. Partnership for protection: New strategies for planning and management of protected areas. Earthscan Publications, London.

Pickett, S. T. A., and J. N. Thompson. 1978. Patch dynamics and the design of nature reserves. Biological Conservation **13**:27-37.

Pressey, Robert L., M. E. Watts, and T. W. Barrett. 2004. Is maximizing protection the same as minimizing loss? Efficiency and retention as alternative measures of the effectiveness of proposed reserves. Ecology Letters **7**:1035-1046.

Pyke, C. R., and D. T. Fischer. 2005. Selection of bioclimatically representative biological reserve systems under climate change. Biological Conservation **121**:429-441.

Rodriguez, A. S. L., H. R. Akçakaya, S. J. Andelman, M. I. Bakarr, L. Boitani, T. M. Brooks, J. S. Chanson, L. D. C. Fishpool, G. A. B. Da Fonseca, K. J. Gaston, M. Hoffmann, P. A. Marquet, J. D. Pilgrim, R. L. Pressey, J. Schipper, W. Sechrest, S. N. Stuart, L. G. Underhill, R. W. Waller, M. E. J. Watts, and X. Yan. 2004. Global gap analysis: priority regions for expanding the global protected-area network. Bioscience **54**:1092-1100.

Schmiegelow, F. K. A., S. G. Cumming, S. Harrison, S. Leroux, K. Lisgo, and B. Olsen. 2005. Conservation beyond crisis management: the matrix reclaimed. University of Alberta, Edmonton, Alberta.

Senate Subcommittee on the Boreal Forest. 1999. Competing realities: the boreal forest at risk. Report of the Sub-Committee on Boreal Forest of the Standing Senate Committee on Agriculture and Forestry, Ottawa, Ontario.

Soares-Filho, B. S., D. C. Nepstad, L. M. Curran, G. C. Cerqueira, R. A. Garcia, C. A. Ramos, E. Voll, A. McDonald, P. Lefebvre, and P. Schlesinger. 2006. Modelling conservation in the Amazon basin. Nature **440**:520-523.

Wilson, K. A., M. F. McBride, M. Bode, and H. P. Possingham. 2006. Prioritizing global conservation efforts. Nature **440**: 337-340.