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Evaluating and Monitoring Habitat Loss Using Satellite Remote Sensing Imagery

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Evaluating and monitoring habitat loss using satellite remote sensing imagery

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Abstract

Habitat loss is widely acknowledged as the leading cause of extinctions and is occurring at an alarming rate and affecting biodiversity globally. I measured the rate of habitat loss using satellite-based land cover change data. First, I modelled the potential suitable habitat of the Marbled Murrelet on Vancouver Island using two techniques and compared those results to *in-situ* field measurements. Both modelling techniques predicted declines in suitable habitat between the years, although one technique was better at predicting suitable habitat. I also compared rates of habitat loss in areas of high species endangerment to those with lower endangerment over a 15-year period in three Canadian ecozones. In two ecozones, rates of habitat loss were higher in sites of high species at risk richness than those with low richness. These results underscore the importance of using remote sensing data as a monitoring tool critical habitat for species at risk in Canada.

Résumé

La perte d'habitat naturel, reconnue comme la cause principale d'extinction, se produit à un taux inquiétant et a de conséquences négatives importantes pour la biodiversité à l'échelle globale. J'ai mesuré le taux de l'habitat perdu en utilisant des données du changement de l'occupation du sol obtenu des images satellites. Premièrement, j'ai développé un modèle pour prédire les changements dans le montant d'habitat approprié du guillemot marbré à l'Ile de Vancouver, en me basant sur deux techniques ainsi que des données obtenues sur le terrain. Tout deux techniques ont prédit un déclin dans le montant d'habitat approprié entre années, par contre un des deux était meilleur à prédire la localité d'habitat approprié. J'ai comparé les taux de perte d'habitat en régions avec haut niveau d'espèces en péril à ceux ayant un bas niveau d'espèces en péril durant une période de 15 ans dans trois écozones au Canada. Dans deux écozones, les taux de perte d'habitat étaient plus élevés dans les sites où le niveau d'espèces en péril était plus haut comparé aux sites où le niveau était plus bas. Ces résultats soulignent l'importance de l'utilisation de la télédétection comme outil pour la surveillance de l'habitat critique des espèces en péril au Canada.

General Introduction

A majority of Canada's area (~65%) can be classified as wilderness (Sanderson *et al.*, 2002), however, it is an area where latent species extinction risk is high (Cardillo *et al.*, 2006) and whose rates of endangerment are similar to those of developing countries in the Americas (Kerr & Deguise, 2004). Biodiversity loss is largely agreed to be due to anthropogenic effects, such as urban development and natural land being converted to agriculture (UNEP, 2002). These anthropogenic effects coupled with pronounced climate change will undoubtedly affect Canadian biodiversity strongly in the future. It is increasingly apparent that monitoring biodiversity and the factors that affect it across Canada improve the likelihood of building effective conservation strategies (Duro *et al.*, 2007). This premise lies at the heart of existing and proposed monitoring schemes in Canada (Ecological Monitoring and Assessment Network, 2007; Alberta Biodiversity Monitoring Institute, 2008).

Monitoring biodiversity and species habitat across broad areas using surveys and traditional field techniques is logistically difficult and expensive, so researchers need to rely on other options. Satellite remote sensing permits synoptic observation of extensive, inaccessible areas that are practically or economically infeasible using alternative methods. (Kerr and Ostrovsky, 2003). With remote sensing data, researchers are able to obtain spatial data at a variety of spatial and temporal scales (the data that are most often used vary from 1km to 60cm in spatial resolution, with a revisit time varying from daily to bi-monthly), which allows for near real-time monitoring of natural phenomena such as forest fires (Li *et al.*, 2003), avalanches (Huggel *et al.*, 2007), and floods (Overton, 2005; Asante et al., 2007). Remote sensing data have been shown to be useful for species distribution modelling (Venier *et al.*, 2004), determining range shifts of species with respect to changing climate

(Thomas and Lennon, 1999; Parmesan and Yohe, 2003), and for measuring ecosystem function (Jobbagy *et al.*, 2002).

In this thesis, I explored the use of new techniques to detect land cover conversions in areas where biodiversity impacts of such changes could be serious. In Chapter 1 (Comparing model- and *in situ* based assessments of habitat suitability for a cryptic species at risk: Marbled Murrelets) in southwestern British Columbia), I I developed two niche models for Marbled Murrelet and investigated impacts of recent, extensive losses of potentially critical habitat detected using high resolution satellite data. In Chapter 2 (A comparison of rates of habitat loss in areas of high and low endangered species richness in Canada), I again used remote sensing data to investigate land cover changes over a 15-year period in two hotspots of endangered species richness in Canada (southern Ontario and the Okanagan Valley region in southern British Columbia). Results from this thesis are intended to provide constructive tests of the use of satellite data for biodiversity monitoring in areas of Canada where ongoing land cover conversions appear to be exacerbating current losses of many species.

Chapter 1

Comparing model- and *in situ* based assessments of habitat suitability for a cryptic species at risk: Marbled Murrelets

Abstract

Predictions of the impacts of climate and land use changes on biodiversity rely heavily on models that describe species niche characteristics to predict areas of suitable habitat. These models, however, are rarely tested using field measurements. I modelled the distribution of potentially suitable habitat for the Marbled Murrelet, a coastal seabird found along North America's Pacific coast, on a portion of Vancouver Island, British Columbia for the years 2000 and 2005. I used Maximum Entropy (Maxent) to carry out the modelling using nesting site records and remotely sensed and spatial environmental variables from the two time periods. Habitat suitability models were also built and their performance was compared to the Maxent models. Field work was conducted in select sites throughout Vancouver Island measuring different habitat characteristics and the suitability of Marbled Murrelet habitat. These field measurements were compared to the results of the Maxent and habitat suitability models. Both models predicted a decline in the amount of potentially suitable habitat for the Marbled Murrelet between 2000 and 2005. Measurements in the field had "fair" agreement with the Maxent model, and "substantial" agreement with the habitat suitability model. The loss in habitat between 2000 and 2005 was a result of extensive logging right around the park perimeters. The results highlight the importance of the use of species distribution models in the conservation of threatened species and stress the importance of further testing these predictive models with field data.

Introduction

Niche models, also referred to as species distribution models, are used to predict the distribution of a species by relating occurrence (and sometimes absence and/or abundance) data with a suite of environmental data that represent potential niche parameters to produce a spatially explicit map of suitable habitat for that species. These models are often applied in the context of conservation biology, invasive species and wildlife management, and biogeographical theory (Schadt *et al.*, 2002; Peterson, 2003; Thuiller *et al.*, 2005; Pearce & Lindenmeyer, 1998). These niche models can also be applied through time to predict how changing environmental conditions might affect species' potential ranges (Harte *et al.*, 2004; Thomas *et al.*, 2004; Kharouba *et al,* 2009; Algar *et al,* 2009).

Suitable habitat monitoring is a vital part of biodiversity conservation strategies (Willis *et al.*, 2007). For example, the GAP program in the United States identifies species habitat requirements to give land managers, planners and policy makers key information to make informed decisions for identifying priority areas for conservation (Scott & Schipper, 2006). Likewise in Mexico, CONABIO (National Commission for the Knowledge and Use of Biodiversity) has used similar techniques to conserve biodiversity. In one such study, the potential introduction of a California honey bee was halted after CONABIO's risk assessment found that it would be detrimental for endemic bee species in the area (Koleff, 2004). The Global Biodiversity Information Facility (http://www.gbif.org), the largest biodiversity data portal that currently contains over 150,000,000 occurrence records, has made it easy for many users to create species distribution models in an efficient manner with numerous modelling algorithms (Sutton *et al.*, 2007; Nativi *et al.* 2007).

The increasing use of niche models in species conservation has made the validation of the models a crucial step in the process. Errors in the models, especially commission errors (i.e. false-positives), would be detrimental to conservation planning (Loiselle *et al,* 2003). Although crucial, validation of niche models with on-the-ground field measurements are rarely carried out after the models are created (Mackay & Lindenmayer, 2001). Instead of field validation, species niche models are most often tested internally. This is often done by using 30% of the occurrence points to train and create the models, while the remaining points are used to test the model (Fielding & Bell, 1997; split-sample approach of Guisan & Zimmermann, 2000). Although internal validation is the most widely used accuracy assessment method, the points used to test the models sometimes come from a single field campaign, where the points are generally concentrated in a single region or two, and not dispersed throughout a larger geographical extent.

The current study aims to model the potential suitable habitat of the Marbled Murrelet (*Brachyramphus marmoratus),* which is listed as threatened in Canada by COSEWIC (Committee on the Status of Endangered Wildlife in Canada), using Landsat imagery at a 30 metre resolution. Two distinct modelling techniques were used: subjectively built habitat suability models and statistically-derived Maximum Entropy (Maxent) models. The Marbled Murrelet is a coastal seabird that nests in Western North America, often in old growth rainforests. Logging in these forests, along with declining fish stocks (IUCN, 2007; Parfitt, 2008), threaten this species. I modelled distributions for two time periods (2000 and 2005) and measured the change in the modelled distribution of suitable habitat using two empirically derived methods – simple habitat suitability (i.e. models built based on expert opinion) and statistical niche models. The models were tested in two ways: internal validation and field measurements. Field work was conducted in two sites on Vancouver Island, where a variety of habitat characteristics known to affect the habitat suitability for Marbled Murrelets were measured. The two key questions of this research are the following:

1) How have observed environmental changes affected the potential suitable habitat of the Marbled Murrelet from 2000 to 2005?
2) How well do on-the-ground field measurements compare to both expert-built habitat suitability models and statistically derived niche models constructed from occurrence points?

I predict that the Maxent models will be more accurate because of their strong statistical foundation and recent success in the species distribution modelling literature (Elith *et al.*, 2006).

Both questions, although addressing different subjects, are important for future conservation efforts. The reliability of niche models is an area of research in the field that needs to be explored further to ensure that conservation plans are conceived in an accurate manner with optimal results.

Methods

2.1 Study area

The study region is Pacific Rim National Park in British Columbia, Canada (Figure 1.1). The area of the park is 510 km^2, with the greater park ecosystem (GPE) including watersheds flowing into Clayoquot Sound or within 50km of it or flowing into the Strait of

San Juan de Fuca and Barkley Sound. Old growth, coastal temperate rainforest is characteristic of the area. Right outside the park boundaries and elsewhere around Vancouver Island, however, extensive harvesting of old growth forest is affecting the habitat quality of many species (Garman *et al.*, 1999).

2.2 Study species

The Marbled Murrelet is a coastal seabird that is found along the northwestern coast of North America. It is a cryptic species that is rarely observed because of its behaviour and inaccessible nesting sites (Jodice & Collopy, 2000). Marbled Murrelets on Vancouver Island nest in old growth coniferous trees with large, mossy limbs that they use as platforms for flight (Burger & Bahn, 2004). They are occasionally observed in habitats where small trees and shrubs are dominant (DeGange, 1996), or even in unforested areas near the shore (Piatt & Naslund, 1995). Zharikov *et al.* (2006) found that Marbled Murrelets prefer steep slopes, lower elevations, and proximity to ocean in their model of habitat selection. They also observed that the Marbled Murrelets can successfully breed in old growth forests that were fragmented by logging.

Marbled Murrelet nesting site records were obtained from Pacific Rim National Park ecologists. Generally, direct observations of the Marbled Murrelets are over bodies of water, which are not useful for modelling suitable habitat. Biologists thus often have to use radio-tagging and nest-location to identify species occurrences on land. All the modelling done refers to nesting site habitat, and does not cover feeding habitat, which is entirely sea-based. A total of 36 nesting sites were used as inputs for the Maxent modelling. The spatial accuracy of the data is approximately 30m (Zharikov *et al.*, 2006). For radio-tagging methods, refer to Bradley *et al* (2004). All points were taken from areas around Clayoquot Sound, on the western coast of Vancouver Island.

2.3 Environmental variables

Six environmental variables were selected for niche model construction based on autecological knowledge of the Marbled Murrelets (Meyer & Miller, 2002; Burger & Bahn, 2004). The variables included: distance to old growth conifers, distance to water, slope, aspect, elevation and land cover. Marbled Murrelets often nest in old growth, conifer forests with sitka spruce, hemlock, and Western red cedar being the preferred tree species (Burger & Bahn, 2004). Aspect was also chosen because adult Murrelets prefer south facing slopes so that young chicks can take advantage of the sunlight that heats them up (Burger & Bahn, 2004). Additionally, Murrelets typically nest in low-elevation areas that are generally around 150m above sea level (Burger & Bahn, 2004). All environmental variables at the landscape scale were derived from both Landsat land cover classifications and a digital elevation model (DEM) provided from Geobase (www.geobase.ca), a federal, provincial and territorial government initiative that provides access to up-to-date Canadian, geospatial data. Land cover classifications for the years 2000 and 2005 were produced by Fraser *et al* (2007) using two Landsat-5 scenes from path/row 48/26 and 49/26, covering most of the GPE. Although the accuracy of these particular scenes were not assess, similar techniques at St-Lawrence Islands and Nahanni National Parks yielded accuracy results of 94% and 83% agreement, respectively (Fraser *et al*, In press). The classification was radiometrically normalized to produce a seamless mosaic of the two Landsat images. All geographic information system (GIS) manipulations were carried out using ArcInfo and ArcView

(Environmental Systems Research Institute, 2006) and Geomatica 9.0 (PCI Geomatics, 2003).

2.4 Maximum Entropy niche modelling

I used Maximum Entropy (Maxent) to construct niche models (Phillips *et al.*, 2006). Maxent is a relatively new, machine learning, software that has been shown to perform quite well against other, more established methods such as Generalized Linear Models and Generalized Additive Models (Elith *et al.*, 2006). Maxent works by finding the probability distribution that is closest to uniform such that any of the constraints (i.e. environmental variables used) satisfy the approximation of that distribution. Maxent output consists of a cumulative value for each grid cell (ranging from 0-100), representing the modelled suitability of habitat for the focal species. To build the models, 70% of the occurrence data were used and the remaining 30% were used to independently evaluate the fit of the Maxent model (Fielding & Bell, 1997). Maxent was run 10 times to assess the average behaviour of the algorithm and to reduce the magnitude of possible errors arising from individual model outputs. This is the standard approach and has been the basis of many informative studies (Algar *et al*, 2008; Buermann *et al*, 2008). Environmental variables used in the models were the same variables used in the construction of the habitat suitability models.

Each Maxent model is internally tested using the 30% of occurrence data that were held back during the model construction phase. Models were evaluated using the area under the receiver operator characteristic curve (AUC), a common measure of model accuracy (Fielding & Bell, 1997). Peterson *et al* (2008) and Lobo *et al* (2008) point out some potential pitfalls of the method including equal weighing of omission and commission errors and its lack of information regarding the spatial distribution of model errors. The method, although imperfect, is still informative and a useful tool for accuracy assessment. AUC values range from 0-1, with a value of 0.5 indicating a prediction no better than random and a value of 1 indicating perfect discrimination (Fielding & Bell, 1997). The AUC is often the chosen method of accuracy assessment because its values are independent of any particular threshold of presence/absence and incorporates errors of omission and commission. Omission error would require data on nesting site absences, which were not available for this study. Instead, Maxent creates background pixels, randomly generated points, to act as pseudo-absence points.

Models were converted to binary form representing suitable and unsuitable habitat. There are several objective methods of determining an appropriate binary threshold. I took the average habitat suitability of the map (Cramer, 2003), which was chosen because of the ease of calculation and its reliability (see Liu *et al.*, 2005 for a review on threshold selection). Although information is lost in the conversion to binary maps, they offer a more intuitive and simpler way of interpreting results. Also, the field sites were classified to either suitable or unsuitable habitat based on habitat characteristics, so comparisons will be consistent.

These models can never prove a species presence, but rather make predictions about the suitability of a location with respect to environmental factors known to influence its distribution. Species distribution models have a strong track record and are quite a useful ecological tool (Kharouba *et al*, 2009). For example, Raxworthy *et al.* (2003) discovered seven new species of chameleon in Madagascar based on areas of overprediction from models of other chameleon species.

2.5 Habitat suitability models

Habitat suitability models attempt to quantify habitat quality using characteristics that are important to the species. Habitat suitability models are built by assigning suitability scores to various characteristics (ex. 100 for optimal habitat, 50 for sub-optimal, and 0 for unsuitable, etc.) and aggregating all scores together (Lancia *et al.*, 1982). These maps are simply based on expert opinion so an understanding of the biology of the species in question is crucial so that models can be built as accurately as possible. In that sense, the models are subjectively built, one of the limitations to the technique. These models do confer an advantage, though, over other statistical and machine learning techniques - species with or without occurrence points can be modelled, so long as the biology of the species is understood, and that biology can be translated into spatial measurements. This will be increasingly important for rare species, which oftentimes have very few occurrence records available making other modelling techniques inappropriate.

Like the Maxent models, habitat suitability models were transformed to binary maps using the average habitat suitability value as the threshold. An inherent assumption of both the habitat suitability and Maxent models is that a species will select and use areas that are optimal to its survival, and that it is most likely to be found in higher quality habitat (Schamber & O'Neil, 1986).

2.6 Field validation

Field work was conducted in September of 2007 in two regions within and around

Pacific Rim National Park on Vancouver Island – Bamfield and, north of Barclay Sound, around Tofino and Ucluelet (Figure 1.2). A total of 41 sites were visited, 6 of which were located within the park boundaries while the remaining sites were located in the GPE. The sampling sites were in different geographical regions than the points used to create the models. I measured a variety of habitat characteristics in lieu of detecting the presence of Marbled Murrelets, since Murrelet sightings are extremely rare. The characteristics surveyed, aspect, slope, elevation, general land cover, dominant tree species, diameter at breast height of 5 biggest trees, tree height of representative trees, qualitative canopy closure, canopy complexity, number of mossy limbs, were based on the findings of Burger & Bahn (2004) and known habitat requirements of the Marbled Murrelet (Jodice & Collopy, 2000; Meyer *et al.*, 2002; Meyer & Miller 2002). Each site that was sampled for the above characteristics was also georeferenced (+/- 5 m average precision) with a Garmin GPS unit so that the exact locations could later be compared with the same location in the models.

Of the 41 sites measured, each was classified to either suitable or unsuitable habitat based on the Murrelets' habitat requirements. This step, like the construction of the habitat suitability models, is subjective but based on a thorough understanding of the biology of the species and its habitat requirements. The binary field scores assigned in this stage are treated as the best estimate of true suitable habitat so they are compared to the binary scores of the 2005 habitat suitability and Maxent models (comparisons to the 2000 models would be unreliable because of the potential changes that could have occurred over 7 years). Cohen's kappa statistic of agreement, a statistical measure of the agreement between two binary classifications, was used to compare the field scores to those of the two modelling techniques (Cohen, 1960). Values range from 0 -1, with values greater 0.8 considered "excellent" (see Table 1.1 for Kappa value interpretation). All statistical analyses were carried out using SAS v.9.1 (SAS, 2003).

The usual technique of model testing is to observe the species in the wild and compare field absences and presences to model absences and presences. This method, however, would not be reliable for the Marbled Murrelet as both the species and their nesting habitat are incredibly difficult to find. Instead, the indirect approach of measuring habitat suitability as a proxy of species presence/absence was used. Both modelling techniques measure the suitability of the habitat based on environmental parameters, so the method of measuring the suitability of the habitat in the field is appropriate as a means of testing them.

Results

Both the Maxent and habitat suitability models predicted an overall decrease in the amount of suitable Marbled Murrelet habitat between the years 2000 and 2005. While Maxent predicted an overall decrease of 29 km^2, the habitat suitability models predicted an overall decrease of 6 km^2.

Assessment of Maxent model accuracy tests using the area under the receiver operator characteristic curve yielded an average AUC value of 0.94 (Figure 1.3), which is considered "excellent" according to Swets (1988). A *post-hoc* comparison of the two modelling techniques for both time periods showed very little agreement, with Kappa = 0.0831, $p \ll 0.001$ for the 2000 models and Kappa = 0.0876, $p \ll 0.001$ for the 2005 models (Table 1.2). These low Kappa values signify major differences in what the two modelling techniques are predicting as suitable and unsuitable habitat. A visual inspection of the maps

(Figure 1.4) indicates that Maxent tends to predict suitability further inland relative to the habitat suitability maps.

Field validation of both 2005 models yielded considerably different results. According to the criteria set forth by Landis and Koch (1977), there was "fair" agreement between the suitability scores of the 2005 Maxent model and the field data (Kappa = 0.384, $p = 0.0049$, N = 41), whereas the 2005 habitat suitability map showed "substantial" agreement with the field data (Kappa = 0.656, $p < 0.0010$, N = 41) (Table 1.3).

Discussion

Marbled Murrelets are known to depend on old growth forests, and the loss of those forests likely reduces their nesting habitat. Raphael *et al.* (2002) found that the amount and distribution of suitable nesting habitat played was correlated with the number of Marbled Murrelets in 10 river drainages on the Olympic Peninsula, in Washington. Both the habitat suitability and Maxent models predicted a decrease in the amount of suitable habitat between the years 2000 and 2005, which is most likely due to the vast amounts of old growth forest being harvested in areas adjacent to the park (Figure 1.5). Habitat loss threatens the survival of this species (Nelson, 1997; IUCN, 2007), so further losses of habitat documented here will almost certainly reduce Murrelet populations further, a conclusion supported by models developed here.

There is a large discrepancy in the accuracy values of the 2005 Maxent models between the internal validation method (e.g. AUC) and the field scores. AUC values were very high for that particular index (0.94), while the Kappa statistic scores from the field indicated less agreement (0.384). This indicates that the Maxent models fit known nesting sites well but that it extrapolates poorly to unsampled regions. Nesting sites that both trained and tested the models were generally in the same geographical region, which could explain the inflated AUC values. Also, use of pseudo-absence points can inflate AUC scores if they are generated in a more environmentally distant location from those of the known presences (Lobo *et al.,* 2008). Another pitfall of using AUC as a method of accuracy assessment is that it weighs omission and commission errors equally. This mismatch in error weighting can also have practical implications. For example, the designation of reserves should weigh commission errors (misclassification of absences) more seriously than omission errors. Although AUC is the most widely used method of accuracy assessment, there are some issues that remain to be resolved (Lobo *et al.,*2008; Peterson *et al.,* 2008).

Habitat is changing rapidly right outside park boundaries, which is likely detrimental to Marbled Murrelets. Any conservation planning for the area should include buffer zones, extended areas of protection around park boundaries, as we know that clear cuts are being done right up to the park boundaries. Buffer zones are viewed as important for the maintenance of biodiversity (Jongman, 1995; Schafer, 1999) and would be especially important for Marbled Murrelets. Buffer zones serve two main functions: 1) to reinforce the reserves by extending its total area and 2) to reduce negative impacts on the park from its surroundings (Batisse, 1997). Buffer zones are particularly important for small parks, such as Pacific Rim National Park, which tend to have a high proportion of edge habitat (Matlack & Litvaitis, 1999). Edge habitat differs from mature forest in that humidity, temperature, openness, light distribution and microclimatic conditions will be altered (Forman, 1995; deMaynadier & Hunter, 1998). The creation of well defined buffer zones around the Pacific

Rim National Park boundaries would be key in preserving critical Marbled Murrelet habitat for the future.

A visual inspection of the maps indicates that Maxent tends to predict suitability further inland relative to the habitat suitability maps. Given that that Marbled Murrelets will not nest in habitat farther than 80km from the coast (Hamer *et al.*, 1995), I suspect that the Maxent models are inaccurate because they are failing to weigh this criterion appropriately. This is likely occurring because the habitat suitability maps gave a value of 0 for any location where any of the niche parameters was classified as unsuitable regardless of the values of the remaining niche parameters (i.e. those localities are predicted to be outside the species' "envelope" of habitable environments). The Maxent models, however, seem to allow for species to inhabit areas where particular environmental parameters are unsuitable to them. The field data results provide a way of assessing which modelling method most accurately predicts suitable Marbled Murrelet habitat. Again, by the criteria set forth by Landis and Koch (1977), there was "fair" agreement between the binary field score and the Maxent model. The habitat suitability model, which was initially predicted to be the less accurate of the two models because of its extremely simple statistical foundation, reflected more closely to field data. This is an interesting result and one that requires further research. Few studies actually use field data to test species distribution models (but see Greaves *et al.*, 2006). Instead, species distribution models are typically tested internally (as in step 2.4.3) with some observations being held back from model construction to allow accuracy assessment (usually using AUC values). It is difficult to question model results when the only accuracy assessments possible for them indicate "excellence" (using the terminology of Swets 1988) and no additional data exist. The mismatch between suitabilities, as observed directly in the field, with estimates from a widely used modelling method (Maxent) emphasizes the need for expanded field testing of species distribution models. Conversely, however, this result could be viewed as encouraging, as qualitative habitat suitability models may perform effectively for rare and poorly-surveyed species.

These models, like all species distribution models, should be interpreted with caution. There were other environmental parameters that could have been considered at the landscape scale that were not considered (ex. canopy complexity and closure, which would have required different remote sensing data sources that could not be obtained for this study). Biological interactions such as predation and competition, which were also excluded from this analysis, can also play a role in species distribution models, particularly when those biotic interactions are very well-defined and represent clear niche dimensions (ex. obligate host plants for a butterfly species: Araujo & Luoto, 2007). Unfortunately, biological interactions are often difficult to quantify and reliable data may not be available, especially in remote regions where field observations required to collect such data are difficult. Although biological interactions are not a particularly important niche parameters for Marbled Murrelets, it is something that should be taken into consideration for other such studies. The variables that were ultimately selected were based on data availability and on the basic habitat and physiological requirements of the Marbled Murrelets. Another potential source of noise with this particular analysis was that field data and satellite data could not be collected at identical times periods (field data are from 2007, Landsat data from 2004 and 2005), which most likely increased disagreements for purposes of the Kappa statistic calculations. A few sites that were sampled during the field campaign had clearly experienced disturbances *since* the satellite observations used in this study were collected, such as blowdowns or clear cuts, which would likely render these areas unsuitable for the Murrelets. The land cover data, however, would obviously not account for changes occurring after its collection. However, these sites were few and did not impede my ability to generate a model (specifically, the habitat suitability model), that strongly agreed with my field sampling.

The results of this analysis closely reflect the conclusions of Greaves *et al.* (2006), one of the few studies to test species distribution models with field data. Their logistic regression model of a threatened bat species in New Zealand had moderate agreement with data collected in the field. One of the key conclusions from Boitani *et al.* (2008), another study to test predictive distribution models with field data, was that empirically derived, statistical models may not be as appropriate as expert-built habitat suitability models when data on species distribution are scarce. Regardless of the modelling techniques used, evaluation of the predictive ability of these models is essential for practical use.

Conclusions

In this study, I found a decline in the amount of suitable habitat for the Marbled Murrelet between 2000 and 2005 due to logging outside park boundaries. Using up-to-date satellite imagery, it will be possible to monitor the status of Marbled Murrelet habitat over time and to direct conservation and mitigation efforts accordingly. Extensive habitat losses extend directly to park boundaries, eliminating habitat for Murrelets across broad areas and pointing to a need to develop buffer zones around the park. This approach could improve the likelihood of conserving Murrelet populations within the park but would also be valuable for regional habitat protection for this species. As Murrelets are already threatened by habitat loss, rapid deforestation of coastal, old growth forests will increase threats to this species'

Simple habitat suitability models may be more accurate than more complex, machine learning modelling techniques such as Maxent. Further work using on-the-ground field data will need to be done to resolve the issues of accuracy with different modelling techniques. These models, however, will continue to be useful for conservation planning and other important ecological applications.

Table 1.1 – Interpretation of kappa values. Adapted from Landis, JR and Koch, GG. (1977).The measurement of observer agreement for categorical data. *Biometrics*, 33, 159-174.

Kappa	Intrepretation
0	No agreement
0 - 0.19	Poor agreement
0.2 - 0.39	Fair agreement
0.4 - 0.59	Moderate agreement
0.6 - 0.79	Substantial agreement
0.8 - 1.0	Almost perfect agreement

Table 1.2 – Confusion matrix, with corresponding Kappa values and 95% confidence intervals, for the comparison between the Maxent models and habitat suitability models for both the 2000 and 2005 time periods.

	2000 models				2005 models		
	0	1	**Total**		0	1	**Total**
0	17347	8148	25495	0	17379	8031	25410
1	6132	4218	10350	1	6171	4264	10435
Total	23479	12366	35845	**Total**	23550	12295	35845

Kappa = 0.0831
95% confidence interval (0.0727, 0.0936)

Kappa = 0.0876
95% confidence interval (0.0771, 0.0981)

Table 1.3 - Confusion matrix, with corresponding Kappa values and 95% confidence intervals, of the comparison between field data scores with those of the 2005 Maxent and habitat suitability models.

	Maxent				Habitat suitability		
	0	1	**Total**		0	1	**Total**
0	20	3	23	0	19	4	23
1	9	9	18	1	3	15	18
Total	29	12	41	**Total**	22	19	41

Kappa = 0.384

95% confidence interval (0.108, 0.659)

Kappa = 0.656

95% confidence interval (0.424, 0.888)

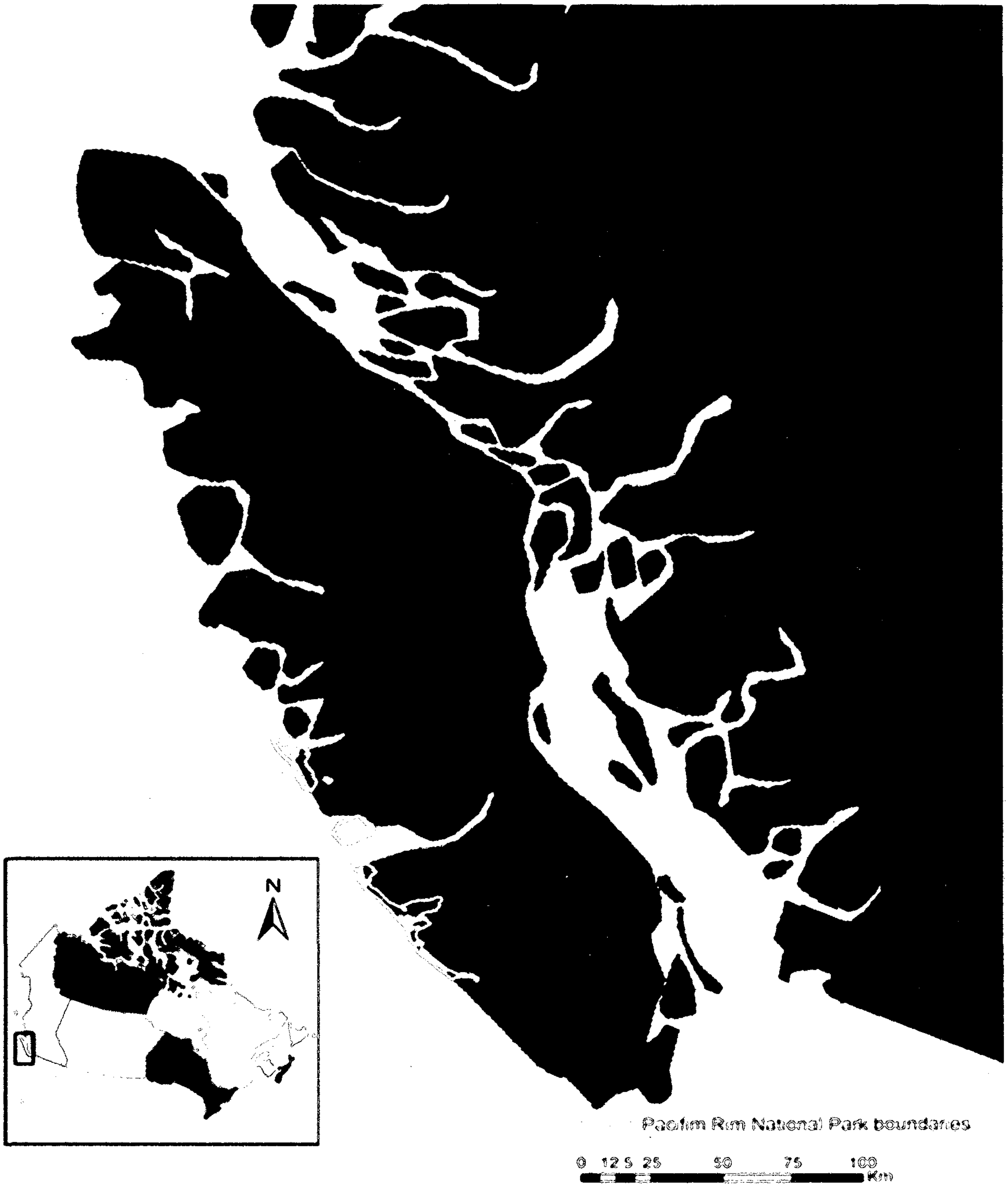

Figure 1.1 – The study region – Pacific Rim National Park, in British Columbia, Canada (48°51'N, 125°19'W).

Figure 1.2 – Location of field sites (red circles) around Tofino, Ucluelet and Bamfield. The underlying map is the land cover classification for the year 2005.

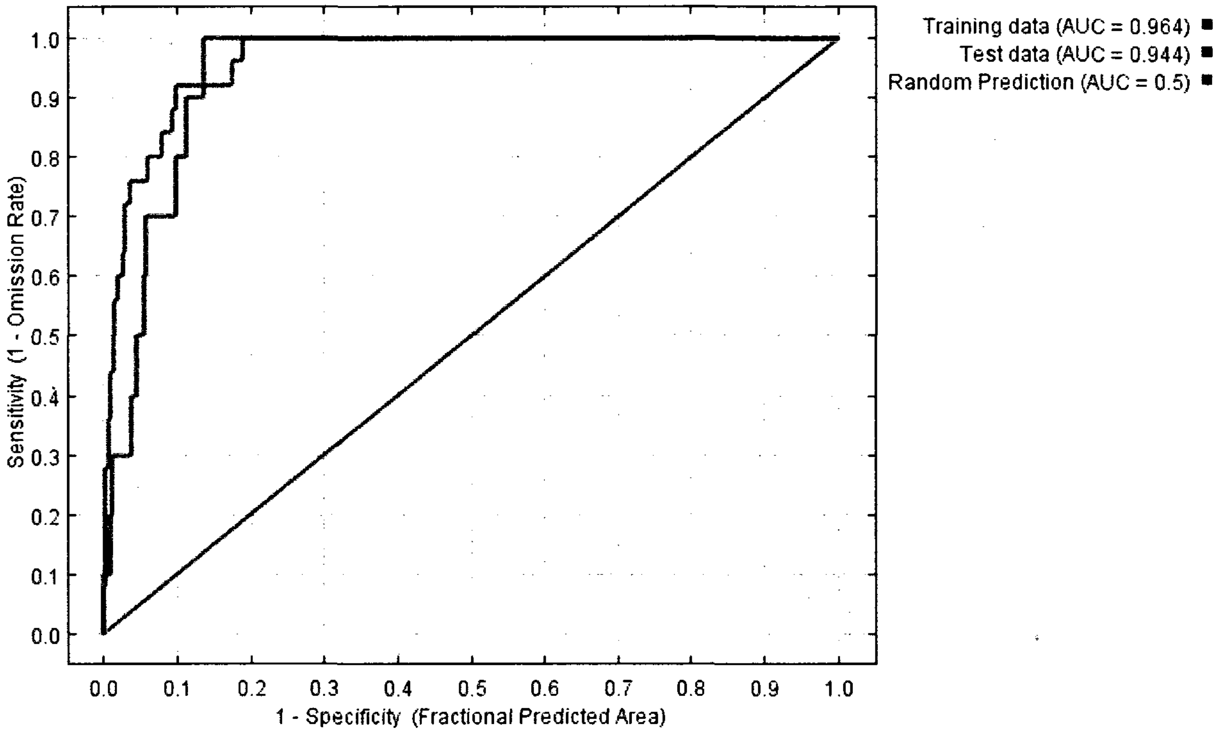

Figure 1.3 – Area under the receiver operator characteristic curve with training and test data values. A random prediction is shown with a green line and represents an AUC value of 0.5.

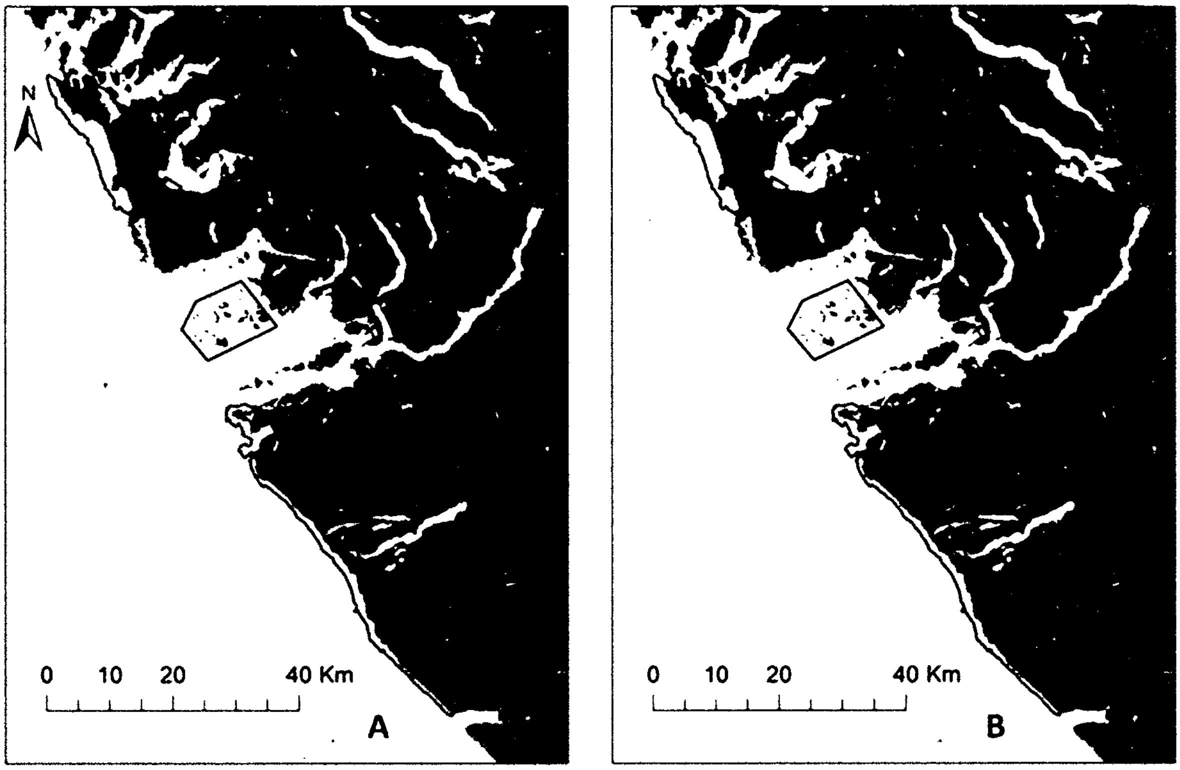

Figure 1.4 – A) Habitat suitability and B) Maxent binary maps for the year 2000. Areas in green are predicted present (suitable habitat) and grey areas are predicted absent (unsuitable habitat). The park boundary is shown in blue.

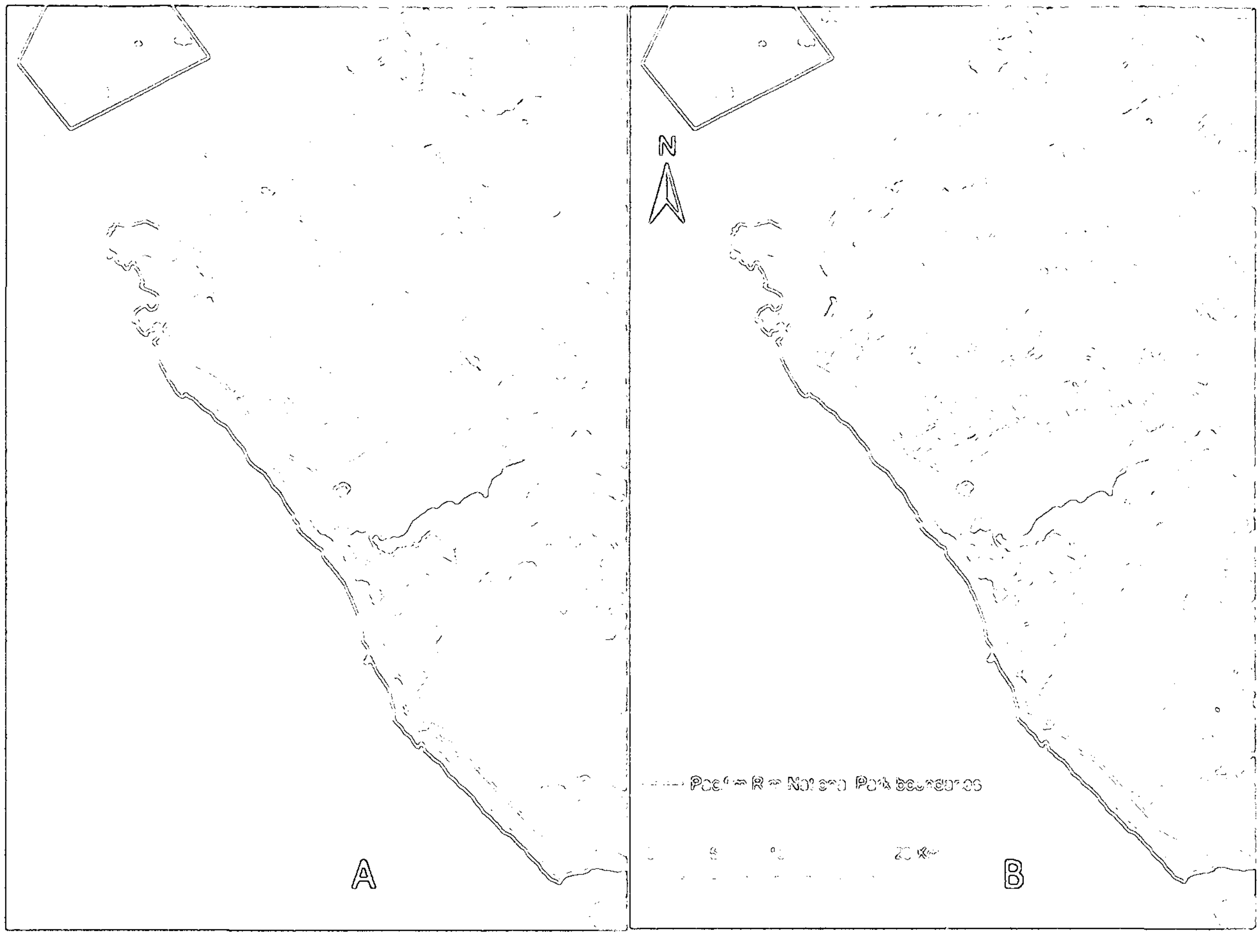

Figure 1.5 – Land cover maps around Bamfield for the years A) 1990 and B) 2005. Light and dark blue patches are barren and fresh cuts, respectively.

Chapter 2

A comparison of habitat loss rates in areas of high and low species at risk richness in Canada

Abstract

It is well established that habitat loss is the primary driver of species endangerment. Endangered species richness in Canada is highest primarily in the southern portion of the country, where agricultural and urban areas are predominant. I looked at the rates of habitat loss during two time periods (1990-2000 and 2000-2005) in areas of both high and low endangered species richness in Canada. Secondarily, I also examined rates of land cover change in an area of southern Ontario, Canada, where species at risk are concentrated, to provide a qualitative view of whether those rates have changed following Canada's inclusion in the Convention on Biological Diversity (in 1992). Habitat loss was higher in areas with higher species at risk richness than in areas with lower richness in both the Mixed Wood Plains and Montane Cordillera ecozones. Also, recent (2000-2005) habitat loss appears to be increasing. This increase extends to southern Ontario since Canada joined the CBD. These data do not indicate that Canadian governments have yet adopted measures to mitigate habitat loss within two of Canada's hotspots of species endangerment.

Introduction

Human activities have precipitated a sixth mass extinction globally and are also degrading ecosystem services upon which human society depends (Vitousek *et al.*, 1997). Extinction rates are expected to increase into the future as climate change impacts accelerate and interact with ongoing land use changes (Thomas *et al.*, 2004; Foley *et al.*, 2005, but see Ladle *et al*, 2004). Among these anthropogenic activities, land-use change is likely to continue to have the largest effect on terrestrial ecosystems in the future (Sala *et al.*, 2000). Land use conversions to agriculture predict species endangerment (Stoate *et al.*, 2001) and are likely the primary cause in Canada (Kerr and Cihlar, 2004). The Mixed Wood Plains ecozone in Southern Ontario is the most diverse region in terms of biodiversity and has more federally-listed species at risk than any other ecozone, but high intensity agricultural activities dominate much of the area (Kerr and Cihlar, 2004). Urbanization presents a second cause of habitat loss for species (Czech and Krausman, 1997; Venter *et al.,* 2006). In Canada, human populations and species richness both increase toward the warmer, southern areas (Figure 2.1). Each year in Canada, new urban centres consume an area roughly the size of Hamilton, Ontario (~ 1,300 km^2) (Oliver, 1999).

In 1992, the Convention on Biological Diversity (CBD), an international treaty designed to conserve biodiversity, was signed and ratified by Canada, legally binding the country to implement its provisions (Convention on Biological Diversity, 2007). It was not until the next year though, that the CBD was put into force. The treaty stresses the use of traditional conservation efforts (e.g. *in-situ* and *ex-situ* conservation) and the sustainable use of biological resources. In 2002, parties to the CBD committed to the 2010 Biodiversity Target, in which they would achieve a "significant reduction of the current rate of biodiversity loss at the global, regional and national level…" (Convention on Biological Diversity, 2007). In 2003, Canada implemented the Species At Risk Act (SARA), which was intended to prevent endangered wildlife from going extinct through habitat and species conservation (Species At Risk Act, 2003). SARA, however, is primarily restricted to federally-owned lands (Species At Risk Act, 2003), which are limited in southern Canada, where the majority of species at risk are located (Deguise and Kerr, 2006). The creation of protected areas (i.e. national parks, nature reserves, etc.) is the most widely recognized conservation method. The effectiveness of these protected areas to reduce extinction rates in Canada's biodiversity hotspots, though, is inhibited by their relatively small size in areas with high endangered species richness (Kerr & Cihlar, 2004; Deguise & Kerr, 2006). Kerr and Cihlar (2004) found that of 11 watersheds with over 25 species at risk in Canada, only about 0.14% of the area (~44km^2) was protected.

Given the conflict of Canada's human population coinciding with its biodiversity, the need to monitor biodiversity hotspots effectively is critical to ensure that the proper precautions are taken to maintain that biodiversity (Duro *et al.*, 2007). Monitoring these areas over time can act as an early warning system, such that regions of concern can be identified and proper mitigation efforts conducted. Monitoring systems that use remote sensing data often use coarse resolution data, which are often ineffective at accurately capturing the key characteristics of the region, including land cover (Cihlar *et al.*, 2003). There are very few monitoring initiatives in Canada that use remote sensing data, with a few exceptions throughout the country.

This study focused on monitoring rates of habitat loss (i.e. natural habitat being converted to human-modified habitat) in areas of both high and low species endangerment richness in Canada over a 15-year interval (1990-2005) using satellite remote sensing imagery. The overall goal is to assess the status of habitat loss rates in both areas of high and low species at risk richness. Ideally, rates of habitat loss should be slowing, especially in areas with many species at risk. To my knowledge, no previous study has either quantified land cover change rates in Canada in areas where biological diversity is high, nor compared those rates with areas where species richness - and numbers of species at risk – are lower. I also conducted a secondary analysis looking at the rate of habitat loss in the greater Toronto area before and after the signing of the CBD treaty in 1992. Although the CBD treaty did not come into force as a binding agreement until 2002, the aspect of measuring rates of habitat loss before and after the implementation date can be help indicate whether Canada has any reasonable prospects of reaching its 2010 target. There has been very little evaluation on Canada's progress towards meetings its CBD goals (Laikre *et al.*, 2008).

Methods

Species distribution data

Species distribution maps were provided by COSEWIC for species that were listed in January 2001 (COSEWIC). There were a total of 243 terrestrial species, including amphibians (N = 13), arthropods (N = 7), birds (N = 46), lichens (N = 4), mammals (N = 34), mosses (N = 1), reptiles (N = 15), and vascular plants (N = 123) (full list of species and COSEWIC status in Appendix A). These maps are based on historical and present-day distribution records and almost certainly overestimate the true distribution of the species. That caveat aside, such maps are informative and are used in studies in biogeography (Hurlbert & White, 2007), evolutionary biology (Bletter *et al.*, 2004), and conservation biology (Dobson *et al.*, 1997). To pinpoint areas of high and low richness for species at risk, their distribution maps were overlaid. Although Southern Canada has the greatest species richness in Canada, its species at risk richness is disproportionately greater (i.e., species are disproportionately endangered in southern Canada) (Kerr & Cihlar, 2004).

Remote sensing data and site selection

I identified two regions in Canada known to have large numbers of species at risk and adjacent areas where there are fewer (Figure 2.2). Sites were ultimately selected based on the availability of remote sensing imagery and the number of endangered species in the area. Five Landsat scenes were obtained from the United States Geological Survey website (http://glovis.usgs.gov) for years circa 1990, 2000, and 2005. Data for 1985 in the greater Toronto area were freely available from Geogratis (http://geogratis.cgdi.gc.ca/). The Landsat Thematic Mapper (TM) sensor is arguably one of the most efficient sensors currently used in ecological studies. It is ideal for this study because the spatial resolution is moderately high at 30m (but not fine enough to limit computing resources), and the revisit time is high enough to detect land cover changes shortly after they occur. I ensured that the Landsat scenes that were obtained for sites of high and low species endangerment were within the same ecozone, to control for the differences in climate, land cover, and soil conditions that typically define an ecozone and the land uses within them. One scene (containing Peterborough, Belleville, and the Abitibi River in Northern Ontario) fell between the boundary of the Boreal Shield and the Mixed Wood Plains ecozones (Figure 2.2). This scene was ultimately clipped in two parts – one part representing the Boreal Shield and the other representing the Mixed Wood Plains. There were thus a total of five Landsat scenes in three different ecozones, including the Mixed Wood Plains, Boreal Shield, and Montane Cordillera for the three time periods (for a total of 15 scenes). All scenes were captured during the summer/early autumn months (June-early October) to reduce the likelihood that phenological differences among scenes would be misinterpreted as land cover change. Each scene's total area was different because cloud cover had to be removed from some images. Additionally, the scene near the Okanagan region was comprised of two separate scenes that had to be mosaicked together to cover the entire study region (Table 2.1).

For the greater Toronto area only, a secondary analysis was carried out looking at rates of land cover change before and after the signing and ratification of the CBD in 1992. This area was chosen because greater data availability made it possible to test whether rates have increased or decreased following the ratification of the CBD. Furthermore, the greater Toronto area contains Carolinian Canada, Canada's most diverse and threatened area (Environment Canada, 2005). If the CBD is to be effective for Canada's biodiversity, this is an area where maximizing species protection is ideal.

The scenes in areas of high species at risk richness that were selected were situated in the greater Toronto area (Mixed Wood Plains ecozone), Okanagan Valley (Montane Cordillera ecozone) of southern British Columbia, and the area around Peterborough and Belleville (portion in the Boreal Shield ecozone). The other portion of the Peterborough and Belleville scene represented the area of low species at risk richness in the Mixed Wood Plains ecozone. Other sites of low species at risk richness were located around Stuart and Babine lake (Montane Cordillera ecozone), and the area around Abitibi River (Boreal Shield ecozone).

Land cover classification and change detection

To process the remote sensing data, I used the Automated Multi temporal Updating through Signature Extension (AMUSE) protocol (Fraser *et al.*, In press). AMUSE produces temporal land cover time series images using updated, remotely sensed satellite imagery. The protocol ensures that proper co-registration (i.e. georeferencing) and radiometric normalization are carried out to maximize the accuracy of the images (Figure 2.3). Radiometric normalization, a method to generate consistency among satellite imagery, was carried out using methods described by Olthof *et al.* (2005). The land cover classifications that I ultimately produced for all the study sites were based on baseline land cover classifications for circa 2005 that were produced for St. Lawrence Islands National Park (SLINP) and the Okanagan Valley using the Federal Geographic Data Committee legend (see Appendix B). The SLINP baseline classification falls within the boundary between the Boreal Shield and the Mixed Wood Plains ecozones, while the Okanagan Valley baseline classification falls in the Montane Cordillera ecozone. Although no field-based data collection in the Okanagan Valley baseline classification image was carried out, a previously completed land cover dataset based on similar techniques north of the Prairies was found to have an accuracy of 91% (Beaubien *et al.*, 2002).

Spectral signatures from baseline SLINP and Okanagan Valley classification images were used to classify the images in the study sites for all time periods. Spectral signatures are formed from the characteristic combination of reflectance values in the three spectral bands (red, near infrared, and short-wave infrared) that consistently characterize particular land covers (Lillesand *et al.* 2004). Some of the study sites, however, had no spatial overlap with the baseline images. The AMUSE protocol was designed to detect land cover changes through time, not space, so an assumption of the study is that the spectral characteristics of the baseline classifications will be the same as those of the study sites. This assumption is used in virtually all land cover classification processes. I am not aware of any reason why this assumption would fail in this study.

Change vector analysis, which measures the difference in reflectance values between two time periods, was used to pinpoint areas of change (Figure 2.4). This method can describe the magnitude and direction of change between the two time periods (Johnson and Kasischke, 1998). Areas where land covers differ between time periods (e.g. forest to field or field to built-up area) show large changes in their spectral signatures compared with areas of no land cover change that retain consistent or nearly consistent spectral characteristics. A subjective threshold was applied to create a binary image representing areas of change and no change. This resulting change mask created can be sensitive to small changes in threshold, which can be minimized by knowledge of known land cover changes during the time period (Fraser *et al*, In press).To make the estimates of overall change more conservative, and to minimize the sensitivity of the subjective threshold, the classification was constrained by a simple rule: land cover change labelling occurs for only those pixels that have are identified as change by the threshold. For example, if the land cover at a location is classified as having not changed between the years 1990 and 2000 because differences in that location's spectral signature were too small to be significant, that area retained the land cover label derived for the initial 1990 classification. This method improves the accuracy of the classification but emphasizes the capacity of the remote sensing imagery to detect land cover differences that are sufficiently large to indicate a substantial ecological trend (Fraser *et al.*, In press). There is the potential for natural habitat loss to occur in some of these areas (e.g. forest fires, windfall, etc.) but these losses are typically balanced by natural renewal through succession over the long term (Sinclair *et al*, 1995).

Measuring rates of habitat loss

Habitat loss was defined as a change from natural land cover to human-modified land cover. It was calculated by constructing binary classification maps using an aggregated system in which all original classes would be classified as either natural or human-modified (Figure 2.6). When calculating change in land cover between two time periods, only those changes from natural to human-modified habitat (ex. forest to agriculture, or urban) were considered (Kerr & Deguise, 2004) as they are the changes that relate to loss of habitat for a majority of species (Bennett, 1990; Kouki & Vaananen, 2000; Pellet *et al.*, 2004). Water was excluded from this aggregated classification scheme since it will, for the most part, remain constant throughout time. By taking the difference between binary maps from the two time periods, four possible patterns can be detected: natural habitat remaining natural, human-modified habitat remaining human-modified, natural habitat becoming human-modified, and human-modified habitat reverting to natural. Only conversions where changes in habitat occurred were considered as areas that remained natural and human-modified do not affect the rates of habitat loss. The land cover conversion rate was then calculated by dividing the relative area of a particular conversion (either natural to human-modified or *vice-versa*) by the number of years in the time period. Rates of both conversion types (human-modified to natural habitat, and vice-versa) were summed together with a negative overall rate reflecting habitat loss, and a positive rate reflecting a gain in natural habitat. Rates were qualitatively compared between areas of low and high species at risk richness. All geographic data were processed using ArcInfo Grid 9.0 (Environmental Systems Research Institute, 2006) and Geomatica 9.0 (PCI Geomatics, 2003).

Land cover classification validation

Validation of any land cover classification is crucial in ensuring accurate results that can be analyzed in a meaningful way. One hundred random points were generated for each 2005 Landsat classified image and compared to very high resolution imagery provided from Google Earth (Figure 2.5), which utilizes data from several imaging sources including the Quickbird and SPOT sensors. The spatial resolution around all study sites was very high, with some areas (mainly in Southern Ontario) having a resolution of less than 1m. The Landsat classification scheme, which initially had 16 classes for Ontario and 20 for the Okanagan Valley, was aggregated down to 6 classes. Fewer classes improves accuracy of change detection and facilitated accuracy assessment using supplementary imagery available through Google Earth. Accuracy assessments were based on imagery from 2005, which most closely matches the time period of imagery available through Google Earth. The results were quantified simply by calculating the proportion of sites correctly classified, a method commonly used in land cover classification accuracy assessments (Trodd, 1995).

Results

In two of the ecozones (Montane Cordillera and Mixed Wood Plains), sites of high species at risk richness had higher rates of habitat loss than sites with lower species at risk richness. The Boreal Shield ecozone, however, was an exception with sites of low endangered species richness had higher rates of habitat loss. Conversion rates to human-modified habitat increased in the 2000-2005 period compared to rates in the 1990-2000 time period in four of the six study sites. In the site of high species at risk richness in the Mixed Wood Plains, for example, conversion rates to human-modified habitat nearly quadrupled in the 2000-2005 time period. Again, the Boreal Shield ecozone was an exception with rates of habitat loss decreasing in the second time period in sites of high species at risk richness. This decrease was relatively small though, with a net gain of approximately 0.3 km^2 of natural habitat per year. Of the six sites studied, five experienced an overall loss of habitat over the 15 year period (the Mixed Wood Plains site of low species at risk richness was the only exception, with an increase in natural habitat of about 2.7 km^2 each year) (Table 2.2). Rates of habitat loss also increased during the 2000-2005 time period in four of the six study sites.

Land cover conversion rates to human-modified habitat in the greater Toronto area exceeded those to natural habitat in the 1985-1990 period (Table 2.3). In the 1990-2000 time period (during which time the CBD was signed), conversion rates to human-modified habitat decreased by one-third and conversion rates to natural habitat exceeded those of human-modified habitat. Rates to human-modified again increased in the 2000-2005 period. Rates of habitat conversion were higher in the last time period than during the first.

Accuracy of the maps varied from moderate (56%) to very high (89%) and fell very close to the recommended accuracy level of 85% (Thomlinson *et al.*, 1999). Much of that error (44%) of the greater Toronto area scene is caused by agriculture land being classified incorrectly as other land cover types. Agricultural landscapes are known to have highly varied spectral profiles for a variety of reasons, including different crop types and phenology (Guindon *et al.*, 2004). It is worth noting that of the 44% error, 18% of it was confusion between agriculture and urban land. For the purposes of this study, those land cover types were both treated as human-modified land and thus, unsuitable habitat for species, so the impact of the error is less than error rates indicate.

Discussion

For the areas of Canada investigated in Canada, rates of habitat loss are generally greater in areas with high species at risk richness, and these rates seem to be increasing over time. These are areas in which human-modified habitat was more predominant to begin with, so there should be less remaining natural habitat to be modified. Intensification of agricultural practices and urbanization have caused extensive habitat loss in these areas. In certain parts of southwest Ontario, namely Carolinian Canada, over 90% of the original forests have been lost (Environment Canada, 2005) and my results show a decrease of about 7.5 km^2 of natural habitat each year in that region. Most of the remaining forests in this region were already too small and isolated to accommodate some threatened species in the area including the Acadian Flycatcher and Hooded Warblers (Whittam & McCracken, 1999; Environment Canada, 2005), so the continued loss of habitat may be detrimental to other species. The Okanagan valley is experiencing similar losses of natural habitat. The area experienced a decline of approximately 378 km^2 of natural habitat over a fifteen year period based on the results of this study. Rapid habitat loss in the form of urbanization is occurring there (Brewer *et al.*, 2001) with population growth rates being highest in the south, where the number of species at risk is greatest (BC Stats, 2006). This region is also experiencing an increase in the extent of agriculture. Grape vineyards, which are typically monocultures, are one of the most profitable crops in the area and have experienced an increase in geographical extent of over 600% between 2004 and 2006, with a strong expansion predicted to continue (BC Wine Institute, 2007). The increased extent of agriculture coupled with the irregular terrain and use of irrigation in the Okanagan region has made it necessary for the extensive use of fertilizers and pesticides to increase agricultural yields (Brewer *et al.*, 2001). The use of pesticides in agricultural areas is often associated with species declines for a variety of different taxa (Davidson, 2004; Relyea, 2005; Gibbs *et al*, 2008), potentially further impacting the species at risk in this region beyond the effects of habitat loss itself.

Conversion of apparently natural land covers to human-dominated lands continues in southern Ontario. Habitat loss rates fluctuate considerably but there is no indication that conversion rates have yet declined, even though the CBD has been in force since 1993. Observations here are qualitative and do not rely on comprehensive remote sensing data. Had more extensive data been accessible, it would have been possible to approximate a BACI (Before-After/Control-Impact) experimental design, albeit from a macroecological standpoint, to evaluate the effect of CBD legislation upon habitat conversion rates. There are many ways that such a pseudo-experiment could be constructed, perhaps including data from other countries that did not ratify the Convention on Biological Diversity and that might, consequently, serve as an imperfect control against which Canadian observations could be compared. Such data are not accessible, nor can constant change detection methodology be applied to regions beyond Canadian borders, where the method used here remains untested and different land covers may exist.

This study was limited by the number of freely accessible Landsat scenes and by the technical difficulty of mosaicking very large areas for change detection. This is a problem that is difficult to avoid in studies of this scale. Given that Landsat data were only easily accessible for years circa 1990, 2000, and 2005 (except for the 1985 scene around the greater Toronto area), the two time periods studied (1990-2000 and 2000-2005) were essentially the only options available to study. Once all data in the archives are made available, scenes dating back to the 1970's can be used in a more extensive study, allowing for not only a larger sample size, but a longer time frame to observe changes in land cover. The inclusion of more data would also make change detection less sensitive to error because there will be small radiometric differences that would require larger changes to be observed before a statistically significant difference could be reliably labelled.

Based on the current distribution of protected areas in Canada, the hotspots of species endangerment in this study are hardly protected at all. Only about 0.3% of the Mixed Wood Plains ecozone is protected, with a mean reserve size of about $4km^2$ (Kerr & Cihlar, 2004). Although a larger proportion of the Montane Cordillera ecozone is protected (about 16%), few protected areas have been established in the southern portion of this ecozone, where species endangerment is most pronounced (Figure 2.7). This area is especially vulnerable because natural habitat is decreasing at a rate of about $33km^2$ per year. There is hope, however, as plans are currently underway by the British Columbia and federal governments to conduct a joint feasibility analysis to investigate the possibility of a new national park in the South Okanagan and Similkameen Valleys, with a decision of whether to go through with the plan to be made by the end of 2008 or early 2009 (Parks Canada, 2008). Such a park has the potential to protect habitat for more than 250 species at risk (South Okanagan Naturalist's Club & Wilderness Committee, 2006), and to protect a variety of ecosystem types, including alpine tundra, grasslands, and "pocket desert".

The recovery potential of many species in southern Canada is extremely limited given the lack of protected areas in this area (Kerr & Cihlar, 2004). Warman *et al.* (2004) found that sites of irreplaceability, a measure of the likelihood that a species will be represented in an area, were highest in southern Canada and that protection is more warranted in these areas. Under SARA, habitat protection on federal lands is possible, but there is very little such land in areas where species at risk are concentrated (Kerr & Cihlar, 2004). Knowing this, it would be useful to consider the impacts that changes in climate and land cover will have on species (Kharouba *et al.*, In press; Algar *et al.*, 2009). Species range shifts are already known to be occurring with changes in climate for a variety of taxa (Parmesan & Yohe, 1999; Thomas & Lennon, 1999; Hickling *et al.*, 2006; White & Kerr, 2006), so the strategic placement of new parks further north in Canada is a possibility for effective conservation.

My results reveal that increasing habitat loss continues to threaten some of Canada's most biologically fragile and diverse areas. In fact, Canada has rates of species endangerment that are comparable with other countries in the Americas with more widespread habitat loss because habitat loss in Canada is most prominent in areas with high numbers of endangered species (Kerr & Deguise, 2004). These rates of habitat loss will undoubtedly decrease as the amount of human-modified habitat increases but they should, in my opinion, decrease for the purposes of biological conservation and not because of the diminishing availability of natural habitat. Although parks are the foundation of biological conservation, they cannot be solely relied on to conserve biodiversity (Margules & Pressey, 2000). Expanding community-based conservation to complement protected areas will also prove to be potentially useful (Sinclair & Byrom, 2006; Kathleen Duncan, M.Sc. thesis) in conserving species at risk in Canada.

Conclusions

Habitat loss continues to increase in some areas of Canada, especially areas with high species at risk richness. To achieve Canada's 2010 Biodiversity Target, the country will likely have to rely heavily on community-based conservation, as the likelihood of the creation of a new large park (at least in southern Ontario) is very doubtful. Although the data available in this study to test the effectiveness of the CBD are far from comprehensive, these results provide a perspective intended to be constructive. As new remote sensing data become available, further monitoring of rates of habitat loss can continue so that areas of concern can be pinpointed and mitigation efforts taken accordingly.

Table 2.1 - The five selected sites with their corresponding location, range of acquisition dates, path/row, area, and the ecozone(s) within that scene. The maximum number of endangered species in each scene determined sites of low and high species at risk richness.

Location	Date range	Path/Row	Ecozone(s)	Scene area (km^2)	Maximum number of endangered species
Peterborough, Belleville Φ	June 15-Sep. 5	015/028	Boreal Shield/Mixed Wood Plains	10,895 / 10,177	24
Abitibi River and surroundings†	July 8-Sep. 6	019/026	Boreal Shield	25,207	6
Greater Toronto area§	June 12-Sep. 20	017/030 & 018/030	Mixed Wood Plains	4,571	43
Okanagan Valley region§	July 30-Sep. 3	045/025 & 045/026	Montane Cordillera	53,965	23
Stuart and Babine Lake†	Aug. 2-Oct. 2	050/022	Montane Cordillera	29,060	6

Φ – Site was divided in two to represent both ecozones. One half was the site of low endangered species richness for the Mixed Wood Plains, while the other half was the site of high endangered species richness for the Boreal Shield.

† – A low species at risk richness site

§ – A high species at risk richness site

Table 2.2 - Conversion rates (% of scene area/year) of the two conversions for all ecozones between the two time periods in sites of high and low species at risk richness. The rate was calculated by dividing the proportion of land being converted with respect to the total area of land by the number of years in that time period. The overall rate of changes in habitat summed across all ecozones for all time periods is -0.197, a loss of natural habitat.

	Mixed Wood Plains				Montane Cordillera				Boreal Shield			
Conversion type	**Low SAR richness**		**High SAR richness**		**Low SAR richness**		**High SAR richness**		**Low SAR richness**		**High SAR richness**	
	1990-2000	2000-2005	1990-2000	2000-2005	1990-2000	2000-2005	1990-2000	2000-2005	1990-2000	2000-2005	1990-2000	2000-2005
HM to natural†	0.0973	0.0745	0.0989	0.124	0.00117	0.0132	0.0339	0.0430	0.00753	0.00790	0.00157	0.00835
Natural to HM§	0.0404	0.105	0.0525	0.187	0.0252	0.00403	0.0764	0.105	0.0364	0.0639	0.00671	0.00564
Overall rate	0.0569	-0.0305	0.0464	-0.0630	- 0.0240	0.00917	- 0.0425	-0.0620	- 0.0289	-0.0560	- 0.00514	0.00271

†- Human-modified to natural habitat

§- Natural to human-modified habitat

Table 2.3 – Rate of conversion (% of scene area/year) to human-modified and natural habitat in the greater Toronto area during the four time periods. The overall rate of changes in habitat summed across all ecozones for all time periods is -0.0954, a loss of natural habitat.

Conversion type	1985-1990	1990-2000	2000-2005	1985-2005
HM to natural†	0.0829	0.0989	0.124	0.0746
Natural to HM§	0.152	0.0525	0.187	0.0843
Overall rate	-0.0691	0.0464	-0.0630	-0.00970

† - Human-modified to natural habitat

§ - Natural to human-modified habitat

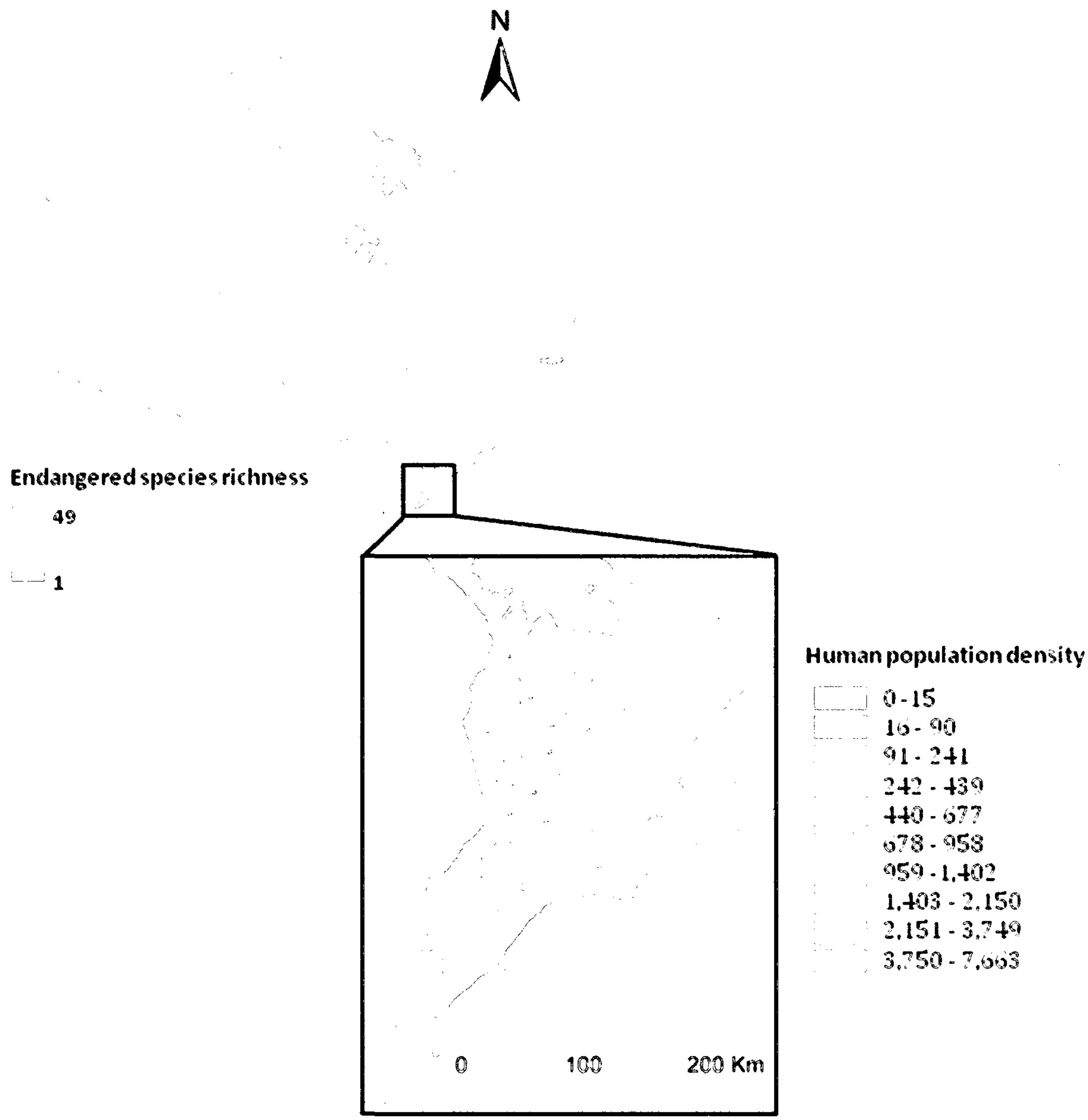

Figure 2.1 – Endangered species richness in Canada. Inset showing human population density in Southern Ontario – the area of Canada with the highest concentrations of species at risk. Human population density unit is individuals/km^2.

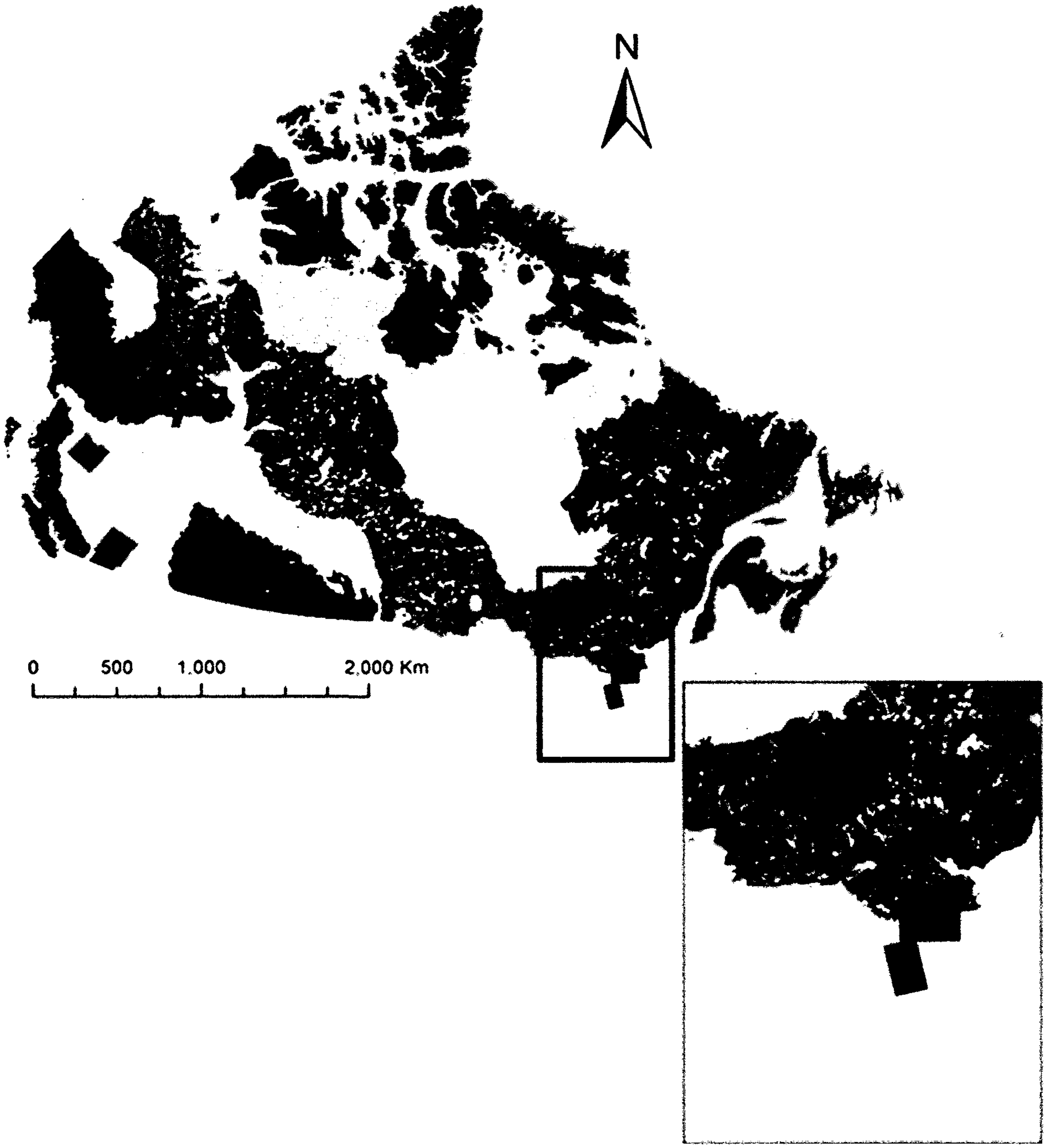

Figure 2.2 – Location of the five Landsat scenes distributed throughout 3 ecozones in Canada. Detail is shown around the Mixed Wood Plains (green) – Boreal Shield (blue) boundary, where one scene was divided to represent both ecozones.

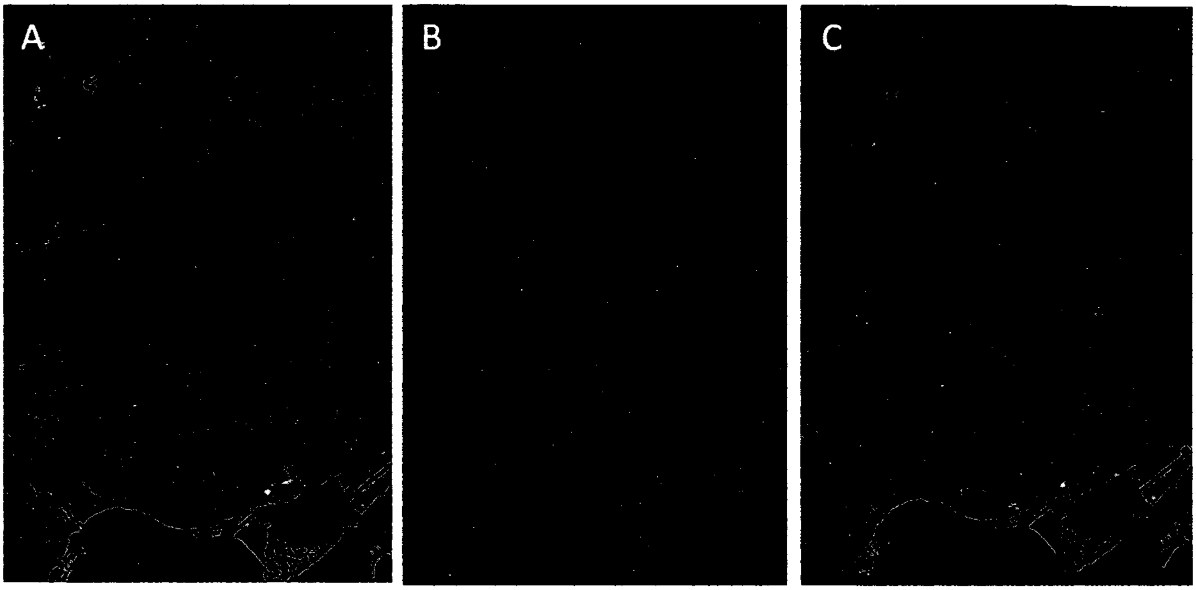

Figure 2.3 – Example of radiometric normalization in Southern Ontario with A) the 1990 baseline image, B) the 2005 un-normalized image and C) the 2005 normalized image.

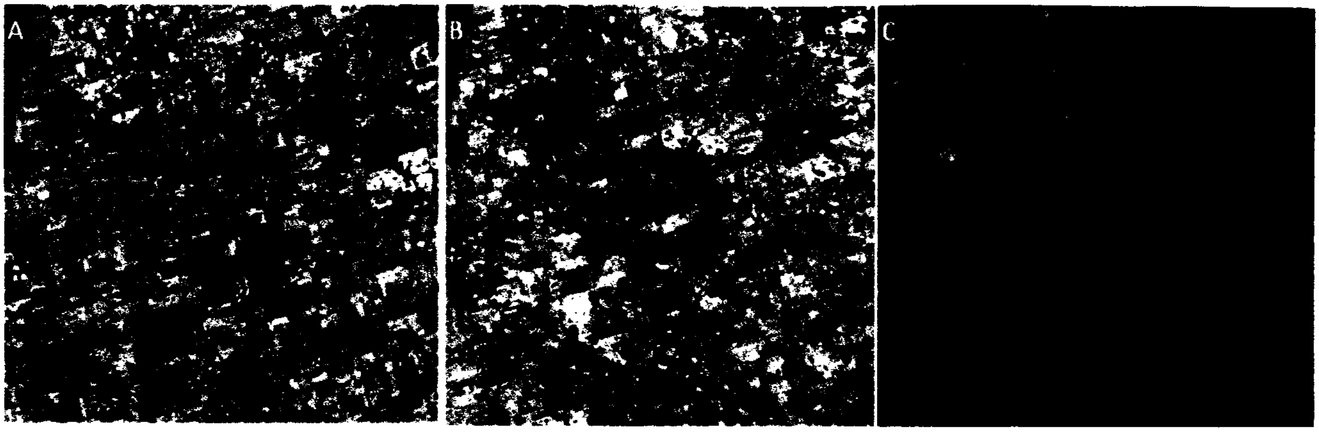

Figure 2.4 – Change vector analysis in Southern Ontario. The reflectance values from the A) 1985 image were subtracted from those of the B) 2005 image to produce an image of C) change. Dark colours in C) represent areas of little change while brighter colours represent areas with more change.

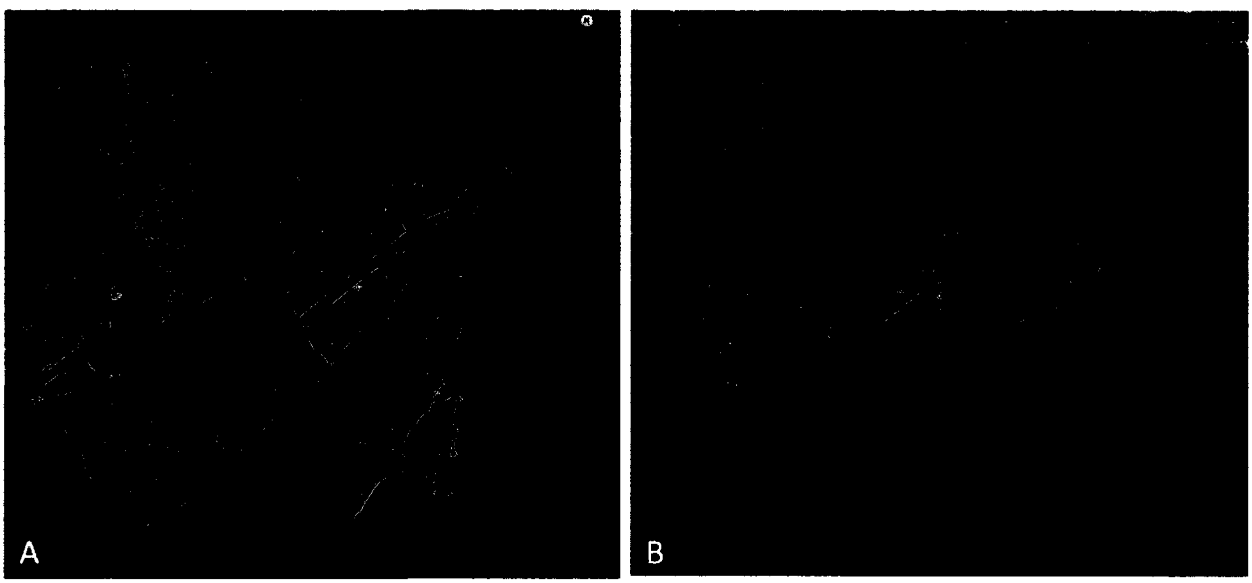

Figure 2.5 – Validation was assessed by comparing A) Google Earth imagery to that of the B) Landsat classified image for the year circa 2005.

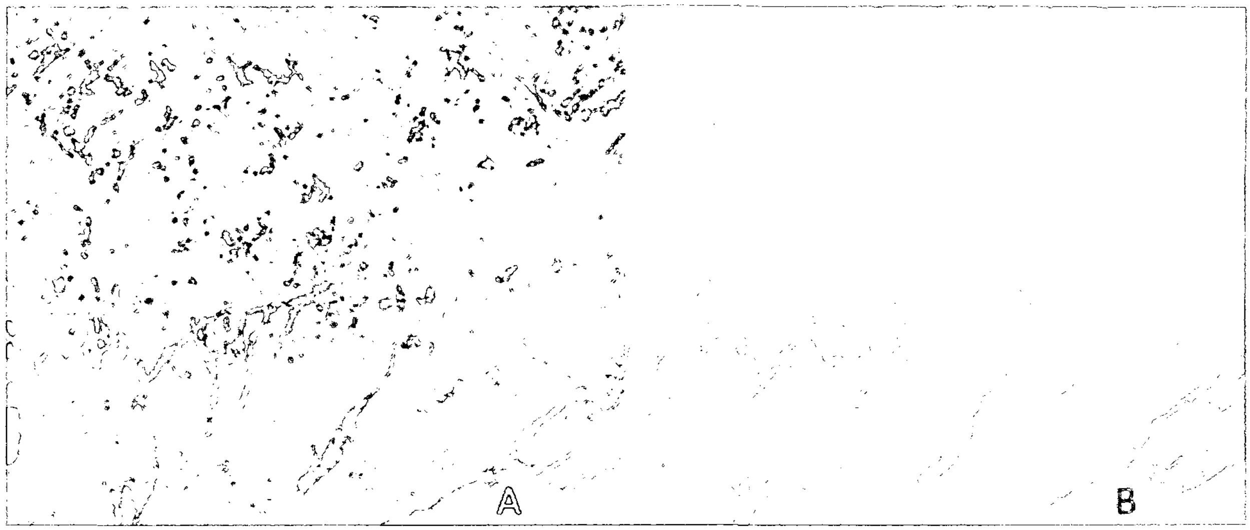

Figure 2.6 – Example of the A) original FGDC classified image for the Boreal Shield/Mixed Wood Plains scene and B) the binary map representing natural (green) and human-modified (dark blue) habitat.

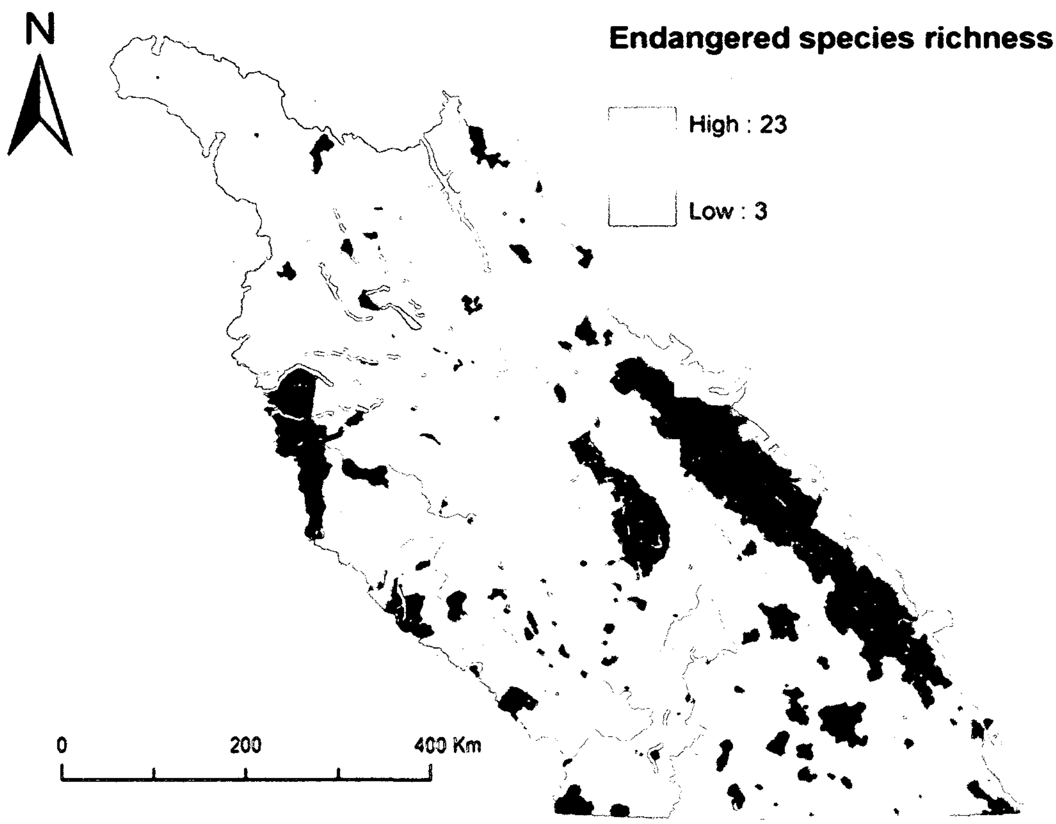

Figure 2.7 – Endangered species richness in the Montane Cordillera ecozone. Areas in black represent protected areas in the region, which cover about 16% of the ecozone.

References

Alberta Biodiversity Monitoring Institute. (2008). Program Overview.

Algar, A. C., Kharouba, H.M., Young, E.R. & J. T. Kerr. (2009). Predicting the future of species diversity: macroecological theory, climate change, and direct tests of alternate forecasting methods. *Ecography,* 32, 1, 22-33.

Asante, K.O., R.D. Macuacua, G.A. Artan, R.W. Lietzow, & J.P. Verdin. (2007). Developing a flood monitoring system from remotely sensed data for the Limpopo basin. *IEEE Transactions on Geoscience and Remote Sensing*, 45, 1709-1714.

Araujo, M.B., & M. Luoto. (2007). The importance of biotic interactions for modelling species distributions under climate change. *Global Ecology and Biogeography,* 16, 743-753.

Batisse, M. (1997). Biosphere reserves—a challenge for biodiversity conservation and regional development. *Environment,* 39, 7–33.

BC Wine Institute. (2007). 2007 Annual Report. Available online at: http://www.winebc.com/publications.php

BC Stats. (2006). BC Stats : Census of Canada. Available online at : http://www.bcstats.gov.bc.ca/census.asp

Beaubien, J., Cihlar, J., & R. Latifovic. (2002). Land cover classification of LANDSAT ETM+ scene 02 August 2000 P52/R10. Satellite information for Land cover of Canada (SILC) Project, Natural Resources Canada, Canada Centre for Remote Sensing, Ottawa, Ontario.

Bennett, A.F. (1990). Land use, forest fragmentation and the mammalian fauna at Naringal, southwestern Victoria. *Australian Wildlife Research.* 17, 325–347.

Bletter, J., Janovec, J., Brosi, B. & D.C. Daly. (2004). A digital base map for studying the Neotropical flora. *Taxon,* 53, 469–477.

Boitani, L., Sinibaldi, I., Corsi, F., De Biase, A., d'Inzillo Carranza, I., Ravagli, M., Reggiani, G., Rondinini, C. & P. Trapanese. (2008). Distribution of medium- to large-sized African mammals based on habitat suitability models. *Biodiversity and Conservation*, 17, 605-621.

Bradley, R.W., Cooke, F., Lougheed, L.W., & W.S. Boyd. (2004) Inferring breeding success through radiotelemetry in the Marbled Murrelet. *Journal of Wildlife Management*, 68, 318–331.

Brewer, R., Cohen, S., Embley, E., Hamilton, S., Julian, M., Kulkani, T., Taylor, B., Tansey, J., VanWynsberghe, R., & P. Whitfield. (2001). Water Management & Climate Change in the Okanagan Basin. S. Cohen & T. Kulkarni (Editors).

Buermann, W., Saatchi, S., Smith, T.B., Zutta, B.R., Chaves, J.A., Mila, B., & C.H. Graham. (2008). Predicting species distributions across the Amazonian and Andean regions using remote sensing data. *Journal of Biogeography*. 35, 1160-1176.

Burger AE, & V. Bahn. (2004). Inland habitat associations of Marbled Murrelets on southwest Vancouver Island, British Columbia. *Journal of Field Ornithology,* 75, 3-66.

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

Cihlar, J., B. Guindon, J. Beaubien, R. Latifovic, D. Peddle, M. Wulder, R. Fernandes, and J. T. Kerr. (2003). From need to product: a methodology for completing a land cover map of Canada with Landsat data. *Canadian Journal of Remote Sensing,* 29, 171-186.

Cohen, J. (1960) A coefficient of agreement for nominal scales. *Educational and Psychological Measurement,* 20, 37–46.

Convention on Biological Diversity. (2007). Available online at: http://www.cbd.int

Cramer, J. S. (2003). *Logit models: from economics and other fields*. Cambridge University Press, U.K.

Czech, B., & P. R. Krausman. (1997). Distribution and causation of species endangerment in the United States. *Science,* 277, 1116–1117.

Davidson, C. (2004). Declining downwind: Amphibian population declines in California and historical pesticide use. *Ecological Applications*, 14, 1892-1902.

DeGange, A.R. (1996). A conservation assessment for the Marbled Murrelet in Southeast

Alaska. USDA Forest Service General Technical Report. PNW-GTR-388. Pacific Northwest Research Station, Portland

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

deMaynadier, P.G., & M.L. Hunter. (1998). Effects of silvicultural edges on the distribution and abundance of amphibians in Maine. *Conservation Biology,* 12, 340–352.

Dobson, A. P., J. P. Rodriguez, W. M. Roberts, & D. S. Wilcove. (1997). Geographic distribution of endangered species in the United States. *Science,* 275, 550–553.

Duncan, K. (2007). M.Sc. thesis title: Species at risk in Canada: Assessing protected areas conservation and community-based conservation solutions.

Duro, D.C., Coops, N.C., Wulder, M.A, & T. Han. (2007). Development of a large area biodiversity system driven by remote sensing. *Progress in Physical Geography*, 31, 235-260.

Ecological Monitoring and Assessment Network. (2007). The Ecological Monitoring and Assessment Network (EMAN).
Available online at: http://beta.eman-rese.ca/eman/program/about.html

Elith, J., Graham, C. H., Anderson, R. P., Dudik, M., Ferrier, S., Guisan, A., Hijmans, R. J., Huettmann, F., Leathwick, J. R., Lehmann, A., Li, J., Lohmann, L. G., Loiselle, B. A., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., Overton, J. M., Peterson, A. T., Phillips, S. J., Richardson, K. Scachetti-Pereira R.,Schapire, R. E., Soberon, J., Williams, S., Wisz, M. S., & N.E. Zimmermann. (2006). Novel methods improve prediction of species' distributions from occurrence data. *Ecography,* 29, 129-151.

Environment Canada. (2005). "Conserve Ontario's Carolinian Forest: Preserve Endangered Songbirds". Available online at:
http://www.on.ec.gc.ca/wildlife/factsheets/fs_songbirds- e.html.

Environmental Systems Research Institute. (2006). Arc/INFO. Version 9.0. ESRI Limited, Redlands, California, USA.

Fielding, A.H., & J.F. Bell. (1997) A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation*, 24, 38-49.

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., & P.K. Snyder. (2005). Global consequences of land use. *Science*, 309, 570-574.

Forman, R.T.T. (1995). *Land Mosaics: The Ecology of Landscapes and Regions.* Cambridge University Press, Cambridge, UK.

Fraser, R., Olthof, I., Pouliot, D., Wang, S., Prevost, S., Leblanc, S., Zhang, Y., & A. Clouston. (2007). Government Related Initiatives Program CCRS Progress Report 06/07. Using satellite remote sensing technology to monitor and assess ecosystem integrity and climate change in Canada's National Parks.

Fraser, R.H., Olthof, I., & Pouliot, D. In press. Monitoring land cover change and ecological integrity in Canada's national parks. *Remote Sensing of Environment.* DOI: 10.1016/j.rse.2008.06.019.

Garman, S.L., Swanson, F.J., & T.A. Spies. (1999). Past, present, and future landscape patterns in the Douglas-fir region of the Pacific Northwest. In: Forest Fragmentation: Implications for Wildlife Management. Rochelle J.A., Lehmann L.A. & J. Wisniewski (Editors). Brill Academic Publishers, Leiden.

Gibbs, K.E., Mackey, R.L. & D.J. Currie. (2008). Human land use, agriculture, pesticides and losses of imperiled species. *Diversity and Distribution*. Published online: Nov. 20, 2008.

Gottschalk, T. K., Huettmann, F., & M. Ehlers. (2005). Thirty years of analysing and modelling avian habitat relationships using satellite imagery data: A review. *International Journal of Remote Sensing,* 26, 2631–2656.

Greaves, G.J., Mathieu, R., & P.J. Seddon. (2006). Predictive modelling and ground validation of the spatial distribution of the New Zealand long-tailed bat (*Chalinolobus tuberculatus*). *Biological Conservation,* 132, 211-221.

Guindon, B., Zhang, Y., & C. Dillabaugh. (2004). Landsat urban mapping based on a combined spectral-spatial methodology. *Remote Sensing of Environment*, 92, 218-232.

Guisan, A., & N.E. Zimmermann. (2000). Predictive habitat distribution models in ecology.

Ecological Modelling, 135, 147-186.

Hamer, T.E., Cooper, B.A., & C.J. Ralph. (1995). Use of Radar to Study the Movements of Marbled Murrelets at Inland Sites. *Northwestern Naturalist*, 76, 73-78.

Harte, J., Ostling, A., Green, J.L., & A. Kinzig. (2004). Biodiversity conservation: Climate change and extinction risk. *Nature,* 430.

Hickling, R., D. B. Roy, J. K. Hill, R. Fox, and C. D. Thomas. (2006). The distribution of a wide range of taxonomic groups are expanding polewards. *Global Change Biology*, 12, 450-455.

Hijmans, R. J., & C.H. Graham. (2006). The ability of climate envelope models to predict the effect of climate change on species distributions. *Global Change Biology,* 12, 2235-2486.

Huggel, C., J. Caplan-Auerbach, C.F. Waythomas, & R.L. Wessels. (2007). Monitoring and modelling ice-rock avalanches from ice-capped volcanoes: A case study of frequent large avalanches on Iliamna Volcano, Alaska. *Journal of Volcanology and Geothermal Research*, 168, 114-136.

Hurlbert, A.H., & E.P. White. (2007). Ecological correlates of geographical range occupancy in North American bird. *Global Ecology and Biogeography*, 16, 764-773.

IUCN. (2007). "2007 IUCN Red List". Available online at: http://www.iucnredlist.org/search/details.php/2989/all.

Jobaggy, E.G., Sala, O.E., & J.M. Paruelo. (2002). Patterns and controls of primary production in the Patagonian steppe: a remote sensing approach. *Ecology,* 83, 307-319.

Jodice, P.G.R., & M.W. Collopy. (2000). Activity patterns of Marbled Murrelets (*Brachyramphus marmoratus*) in Douglas-fir old-growth forests of the Oregon coast range. *The Condor*, 102, 275-285.

Johnson, R.D., & E.S. Kasischke. (1998). Change vector analysis: a technique for the multispectral monitoring of land cover and condition. *International Journal of Remote Sensing*, 19, 411-426.

Jongman, R.H.B. (1995). Nature conservation planning in Europe: developing ecological networks. *Landscape and Urban Planning*, 32,169–183.

Kerr, J.T., & M. Ostrovsky. (2003). From space to species: ecological applications for remote sensing. *Trends in Ecology and Evolution*, 18, 299-304.

Kerr, J. T., & J. Cihlar. (2004). Patterns and causes of species endangerment in Canada. *Ecological Applications*, 14, 743-753.

Kerr, J.T. & I. Deguise. (2004). Habitat loss and the limits to endangered species recovery. *Ecology Letters*, 7, 1163-1169.

Kerr, J. T., Kharouba, H.M., & D. J. Currie. (2007). The macroecological contribution to global change solutions. *Science,* 316, 1581-1584.

Kharouba, H. M, A. C. Algar, & J. T. Kerr. In press. Historically calibrated predictions of butterfly species' range shift using global change as a pseudo-experiment. *Ecology*. DOI:

Koleff, P. (2004). CONABIO: National Commission for Knowledge and Use of Biodiversity, Invasive Species Programme, Mexico. Report prepared for the Experts Meeting Towards the Implementation of a Global Invasive Species Information Network, 6-8 April, 2004. Baltimore, Maryland, USA.

Kouki, J. & A. Vaananen. (2000). Impoverishment of resident old-growth forest bird assemblages along an isolation gradient of protected areas in eastern Finland. *Ornis Fennica*. 77, 145–154.

Ladle, R. J., P. Jepson, M. B. Araujo, and R. J. Whittaker. (2004). Dangers of crying wolf over risk of extinctions. *Nature*, 428,799.

Laikre, L., Bengt-Gunnar, J., Ihse, M., Marissink, M., Dock Gustavsson, A.M., Ebenhard, T., Hasberg, L., Stal, P.O., von Walter, S. & P. Wrammer. (2008). Wanted: Scientists in the CBD Process. *Conservation Biology*, 22, 814-815.

Lancia, R.A., Douglas, S.D., Adams, D.A., & D.W. Hazel. (1982). Validating habitat quality assessment: an example. Transactions of the 47th North American Wildlife Conference, 96-110.

Landis, J.R. & G.G. Koch. (1977).The measurement of observer agreement for categorical data. *Biometrics*, 33, 159-174.

Li, Q, R. Fraser, J. Jin, A.A. Abuelgasim, I. Csiszar, P. Gong, R. Pu, & W. Hao. (2003).

Evaluation of algorithms for fire detection and mapping across North America from satellite. *Journal of Geophysical Research*, 108, 4076, doi:10.1029/2001JD001377.

Lillesand, T.M., Kiefer, R.W., & J.W. Chipman. (2004). *Remote Sensing and Image Interpretation – Fifth Edition*. John Wiley & Sons, Inc. Hoboken, New Jersey, U.S.A.

Liu, C.R., Berry, P.M., Dawson, T.P., & R.G. Pearson. (2005) Selecting thresholds of occurrence in the prediction of species distributions. *Ecography*, 28, 385-393.

Lobo, J.M., Jimenez-Valverde, A., & R. Real. (2008). AUC: a misleading measure of the performance of predictive distribution models. *Global Ecology and Biogeography*, 17, 145-151.

Loiselle, B.A., Howell, C.A., Graham, C.H., Goerck, J.M., Brooks, T., Smith, K.G., & P.H. Williams. (2003). Avoiding pitfalls of using species distribution models in conservation planning. *Conservation Biology*, 17, 1591–1600.

Mackay, B.G., & D.B. Lindenmayer. (2001). Towards a hierarchical framework for modelling the spatial distribution of animals. *Journal of Biogeography*, 28, 1147-1166.

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

Matlack, G. & J. Litvaitis. (1999). Forest edges. In: *Maintaining Biodiversity in Forest Ecosystems*. Hunter, Jr., M.L., editor. Cambridge University Press, Cambridge, USA.

Menendez, R., Megias, A. G., Hill, J. K., Braschler, B., Willis, S. G., Collingham, Y., Fox, R., Roy, D. B., & C.D. Thomas. (2006). Species richness changes lag behind climate change. *Proceedings of the Royal Society Biological Sciences,* 273, 1465-1470.

Meyer, C.B., & S.L. Miller. (2002). Use of fragmented landscapes by Marbled Murrelets for nesting in southern Oregon. *Conservation Biology*, 16, 755-766.

Meyer, C.B., Miller, S.L., & C.J. Ralph. (2002) Multi-scale landscape and seascape patterns associated with Marbled Murrelet nesting areas on the US west coast. *Landscape Ecology*. 17, 95-115.

Nativi, S., Mazzetti, P., Saarenmaa, H., Kerr, J., Kharouba, H., O Tuama, E & S.J. Singh

Khalsa. (2007). Predicting the impacts of climate change on biodiversity – a GEOSS scenario. In: "The Full Picture, A publication for GEO Ministerial Summit, 'Earth Observation for Sustainable Growth and Development' Cape Town, 30 November, 2007".

Nelson, S.K. (1997). Marbled Murrelet (*Brachyramphus marmoratus*). *The Birds of North America*. Poole A., & F. Gill (Editors). Academy of Natural Sciences and American Ornithologists' Union, Philadelphia, PA, and Washington, DC, U.S.A.

Oliver, S. (1999). Building for Tomorrow: Rural Residential Appraisals. Presentation to the Appraisal Institute of Canada. June 18 and 19, 1999. Brock University: St. Catherines, Ontario.

Olthof, I., Pouliot, D., Fernandes, R. & Latifovic, R. (2005). Landsat-7 ETM+ radiometric normalization comparison for northern mapping applications. *Remote Sensing of Environment*, 95, 388-398.

Overton, I.C. (2005). Modelling floodplain inundation on a regulated river: Integrating GIS, remote sensing and hydrological models. *River Research and Applications*, 21, 991-1001.

Parfitt, B. (2008). Restoring the public good on private forestlands. A resource economics project report. Canadian Center for Policy Alternatives.

Parks Canada. "South Okanagan–Lower Similkameen National Park Reserve Feasibility Study". Available online at: http://www.pc.gc.ca/progs/np-pn/cnpn-cnnp/os-os/index_e.asp.

Parmesan, C., & G. Yohe. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *Nature*, 421, 37-43.

PCI Geomatics. (2003). GeomaticaVersion 9.0. Richmond Hill, Ontario, Canada.

Pearce, J., & D. Lindenmayer. (1998). Bioclimatic analysis to enhance reintroduction biology of the endangered helmeted honeyeater (*Lichenostomus melanops cassidix*) in southeastern Australia. *Restoration Ecology*, 6, 238–243.

Pearson, R. G., & T.P. Dawson. (2003). Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? *Global Ecology and Biogeography*, 12, 361-371.

Pellet, J., Guisan, A. & N. Perrin. (2004). A concentric analysis of the impact of urbanization on the threatened European tree frog in an agricultural landscape. *Conservation Biology*. 18, 1599-1606.

Peterson, A.T. (2003). Predicting the geography of species' invasions via ecological niche modelling. *The Quarterly Review of Biology*, 78, 419-433.

Peterson, A.T., Paper, M., & J. Soberón. (2008). Rethinking receiver operating characteristic analysis applications in ecological niche modelling. *Ecological Modelling*, 213, 63-72.

Phillips, S.J., Anderson, R.P., & R.E. Schapire. (2006) Maximum entropy modelling of species geographic distributions. *Ecological Modelling*, 190, 231-259.

Piatt, J.F. & N.L. Naslund. (1995) Abundance, distribution and population status of Marbled Murrelets in Alaska. In: *Ecology and Conservation of the Marbled Murrelet.*C.J. Ralph, G.L Hunt, J.F. Piatt, & M. Raphael, editors. USDA Forest Service General Technical Report. PSW-GTR-152. Pacific Southwest Research Station, Albany.

Raphael, M.G., Evans Mack, D., & B.A. Cooper. (2002). Landscape-scale relationships between abundance of Marbled Murrelets and distribution of nesting habitat. *The Condor*, 104, 331-342.

Raxworthy, C.J., Martinez-Meyer, E., Horning, N., Nussbaum, R.A., Schneider, G.E. Ortega-Huerta, M.A., & A.T. Peterson. (2003). Predicting distributions of known and unknown reptile species in Madagascar. *Nature*, 426, 837-841.

Relyea, R. (2005). The impact of insecticides and herbicides on the biodiversity and productivity of aquatic communities. *Ecological Applications*, 15, 618-627.

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

Sanderson, E., M. Jaiteh, M. A. Levy, K. H. Redford, A. Wannebo, & G. Woolmer. (2002). The human footprint and the last of the wild. *Bioscience*, 52,891-904.

SAS Institute. 2003. SAS system for windows. SAS Institute Inc. Cary, NC.

Schadt, S, Revilla, E, Wiegand, T, Knauer, F, Kaczensky, P, Breitenmoser, U, Bufka, L, Cerveny, J, Koubek, P, Huber, T, Stanisa, C., & L. Trepl. (2002). Assessing the suitability of central European landscapes for the reintroduction of Eurasian lynx. *The Journal of Applied Ecology,* 39, 189-203.

Schamberger, M.L., & L.J. O'Neil. (1986). Concepts and constraints of habitat-model testing. In: Wildlife 2000: Modelling Habitat Relationships of Terrestrial Vertebrates. Verner, J., Morrison, M.L., & C.J. Ralph (Editors). University of Wisconsin Press, Madison,WI, U.S.A.

Scott, J.M., & J. Schipper. (2006). Gap analysis: a spatial tool for conservation planning. In: *Principles of Conservation Biology* (3rd ed.). Sunderland, MA: Sinauer.

Shafer, C.L. (1999). US National park buffer zones: historical, scientific, social and legal aspects. *Environmental Management*, 23, 49–73.

Sinclair, A.R.E., Hik, D.S., Schmitz, O.J., & G.G.E. Scudder. (1995). Biodiversity and the need for habitat renewal. *Ecological Applications*, 5, 579-587.

Sinclair, A.R.E. & A.E. Byrom. (2006). Understanding ecosystem dynamics for conservation of biota. *Journal of Animal Ecology*, 75, 64-79.

South Okanagan Naturalists' Club & Wilderness Committee. (2006). Proposed South Okanagan Similkameen Nature Park Reserve. Volume 25, No. 04.

Species at Risk Act. 2003. Government of Canada, Minster of Public Works and Services, Ottawa.

Stoate, C., Boatman, N. D., Borralho, R.J., Carvalho, C. R., de Snoo, G.R., & P. Eden. (2001). Ecological impacts of arable intensification in Europe. *Journal of Environmental Management*, 63, 337-365.

Sutton, T., de Giovanni, R., & M.F de Siqueira. (2007). Introducing openModeller – A fundamental niche modelling framework. *OSGeo Journal,* 1, 1-6.

Swets, J.A. (1988). Measuring the Accuracy of Diagnostic Systems. *Science*, 240, 1285.

Thomas, C.D., & J.J. Lennon. (1999). Birds extend their ranges northwards. *Nature*, 399, 213.

Thomas, C. D., Cameron, A., Green, R. E., Bakkenes, M., Beaumont, L. J., Collingham, Y. C., Erasmus, B. F. N., Ferreira de Siqueira, M., Grainger, A., Hannah, L., Hughes, L., Huntley, B., van Jaarsveld, A. S., Midgley, G. F., Miles, L., Ortega-Huerta, M. A., Peterson, A. T., Phillips, O. L., & S.E. Williams. (2004). Extinction risk from climate change. *Nature,* 427,145-148.

Thomlinson, J. R., Bolstad, P. V., & W.B. Cohen. (1999). Coordinating methodologies for scaling landcover classifications from site-specific to global: steps toward validating global map products. *Remote Sensing of Environment*, 70, 16– 28.

Thuiller, W, Lavorel, S, Araujo, M.B, Sykes, M.T, & I.C. Prentice. (2005). Climate change threats to plant diversity in Europe. *Proceeding of the National Academy of Science,* 102, 8245-8250.

Trodd, N. M. (1995). Uncertainty in land cover mapping for modelling land cover change. In: *Proceedings of RSS95 remote sensing in action* (pp. 1138– 1145). Nottingham: Remote Sensing Society.

Tuttle, E. M., Jensen, R. R., Formica, V. A., & R.A. Gonser. (2006). Using remote sensing image texture to study habitat use patterns: A case study using the polymorphic white-throated sparrow (*Zonotrichia albicollis*). *Global Ecology and Biogeography,* 15, 349–357.

UNEP. (2002). United Nations Environmental Programme 2002: Report of the Sixth Meeting on the Convention on Biological Diversity.

Venier, L. A., J. Pearce, J. E. McKee, D. W. McKenney, & G. J. Niemi. (2004). Climate and satellite-derived land cover for predicting breeding bird distribution in the Great Lakes Basin. *Journal of Biogeography*, 31, 315-331.

Venter, O., N.N. Brodeur, L. Nemiroff, B. Belland, I.J. Dolinsek and J.W.A. Grant. (2006). Threats to endangered species in Canada. *Bioscience,* 56, 903-910.

Vitousek, P.M., Mooney, H.A., Lubchenco, J., & J.M. Melillo. (1997). Human domination of Earth's ecosystems. *Science,* 277, 494-499.

Warren, M. S., Hill, J.K., Thomas, J. A., Asher, J., Fox, R., Huntley, B., Roy, D.B., Telfer, M.G., Jeffcoate, S., Harding, P., Willis, S.G., Greatorex-Davies, J.N., Moss, D., & C.D Thomas. (2001). Rapid responses of British butterflies to opposing forces of climate and habitat change. *Nature,* 414, 65-69.

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

Whittam, B., & J. McCracken. (1999). Productivity and habitat selection of hooded warblers in southern Ontario. Internal Report Port Rowan, Ontario: Bird Studies Canada. Pp. 47.

Willis, K. J., M. B. Araujo, K. D. Bennett, B. Figueroa-Rangel, C. A. Froyd, & N. Myers. (2007). How can a knowledge of the past help to conserve the future? Biodiversity conservation and the relevance of long-term ecological studies. *Philosophical Transactions of the Royal Society of London*, 362, 175-186.

Zharikov, Y., Lank, D.B., & F. Cooke. (2007). Influence of landscape pattern on breeding distribution and success in a threatened Alcid, the Marbled Murrelet: model transferability and management implications. *Journal of Applied Ecology*, 44, 748-759

Appendix A - List of all COSEWIC-listed species and their corresponding COSEWIC status (n = 243) used to create the species richness map of Canada.

Species Name	Common Name	Taxon	COSEWIC Status
Taxidea taxus jacksoni	American Badger jacksoni subspecies	Mammals	Endangered
Taxidea taxus jeffersonii	American Badger jeffersonii subspecies	Mammals	Endangered
Martes americana atrata	Newfoundland Marten	Mammals	Endangered
Cynomys ludovicianus	Black-tailed Prairie Dog	Mammals	Special Concern
Scalopus aquaticus	Eastern Mole	Mammals	Special Concern
Canis lupus lycaon	Eastern Wolf	Mammals	Special Concern
Mustela erminea haidarum	Ermine haidarum subspecies	Mammals	Threatened
Myotis thysanodes	Fringed Bat	Mammals	Data Deficient
Sorex gaspensis	Gaspé Shrew	Mammals	Special Concern
Urocyon cinereoargenteus	Grey Fox	Mammals	Threatened
Ursus arctos	Grizzly Bear (Northwestern population)	Mammals	Special Concern
Myotis keenii	Keen's Long-eared Bat	Mammals	Data Deficient
Aplodontia rufa	Mountain Beaver	Mammals	Special Concern
Sylvilagus nuttallii nuttallii	Nuttall's Cottontail nuttallii subspecies	Mammals	Special Concern
Dipodomys ordii	Ord's Kangaroo Rat	Mammals	Special Concern
Sorex bendirii	Pacific Water Shrew	Mammals	Threatened
Antrozous pallidus	Pallid Bat	Mammals	Threatened
Rangifer tarandus pearyi	Peary Caribou	Mammals	Endangered
Rangifer tarandus pearyi	Peary Caribou	Mammals	Endangered
Rangifer tarandus pearyi	Peary Caribou	Mammals	Endangered
Ursus maritimus	Polar Bear	Mammals	Special Concern
Glaucomys volans	Southern Flying Squirrel	Mammals	Special Concern
Euderma maculatum	Spotted Bat	Mammals	Special Concern
Vulpes velox	Swift Fox	Mammals	Endangered
Scapanus townsendii	Townsend's Mole	Mammals	Endangered
Marmota vancouverensis	Vancouver Island Marmot	Mammals	Endangered
Reithrodontomys megalotis megalotis	Western Harvest Mouse megalotis subspecies	Mammals	Special Concern
Gulo gulo	Wolverine (Eastern population)	Mammals	Endangered
Gulo gulo	Wolverine (Western population)	Mammals	Special Concern

Bison bison athabascae	Wood Bison	Mammals	Threatened
Rangifer tarandus caribou	Woodland Caribou (Atlantic-Gaspésie population)	Mammals	Endangered
Rangifer tarandus caribou	Woodland Caribou (Boreal population)	Mammals	Threatened
Rangifer tarandus caribou	Woodland Caribou (Southern Mountain population)	Mammals	Threatened
Microtus pinetorum	Woodland Vole	Mammals	Special Concern
Empidonax virescens	Acadian Flycatcher	Birds	Endangered
Falco peregrinus anatum	Peregrine Falcon anatum subspecies	Birds	Threatened
Synthliboramphus antiquus	Ancient Murrelet	Birds	Special Concern
Tyto alba	Barn Owl (Western population)	Birds	Special Concern
Bucephala islandica	Barrow's Goldeneye (Eastern population)	Birds	Special Concern
Catharus bicknelli	Bicknell's Thrush	Birds	Special Concern
Athene cunicularia	Burrowing Owl	Birds	Endangered
Dendroica cerulea	Cerulean Warbler	Birds	Special Concern
Lanius ludovicianus migrans	Loggerhead Shrike migrans subspecies	Birds	Endangered
Icteria virens virens	Yellow-breasted Chat virens subspecies	Birds	Special Concern
Buteo regalis	Ferruginous Hawk	Birds	Special Concern
Otus flammeolus	Flammulated Owl	Birds	Special Concern
Histrionicus histrionicus	Harlequin Duck (Eastern population)	Birds	Special Concern
Ammodramus henslowii	Henslow's Sparrow	Birds	Endangered
Wilsonia citrina	Hooded Warbler	Birds	Threatened
Passerculus sandwichensis princeps	Savannah Sparrow princeps subspecies	Birds	Special Concern
Pagophila eburnea	Ivory Gull	Birds	Special Concern
Rallus elegans	King Rail	Birds	Endangered
Ixobrychus exilis	Least Bittern	Birds	Threatened
Melanerpes lewis	Lewis's Woodpecker	Birds	Special Concern
Numenius americanus	Long-billed Curlew	Birds	Special Concern
Seiurus motacilla	Louisiana Waterthrush	Birds	Special Concern
Brachyramphus marmoratus	Marbled Murrelet	Birds	Threatened
Charadrius montanus	Mountain Plover	Birds	Endangered
Colinus virginianus	Northern Bobwhite	Birds	Endangered
Strix occidentalis caurina	Spotted Owl caurina subspecies	Birds	Endangered
Ardea herodias fannini	Great Blue Heron fannini subspecies	Birds	Special Concern
Falco peregrinus pealei	Peregrine Falcon pealei subspecies	Birds	Special Concern
Charadrius melodus circumcinctus	Piping Plover circumcinctus subspecies	Birds	Endangered
Charadrius melodus melodus	Piping Plover melodus subspecies	Birds	Endangered

Lanius ludovicianus excubitorides	Loggerhead Shrike excubitorides subspecies	Birds	Threatened
Protonotaria citrea	Prothonotary Warbler	Birds	Endangered
Accipiter gentilis laingi	Northern Goshawk laingi subspecies	Birds	Threatened
Melanerpes erythrocephalus	Red-headed Woodpecker	Birds	Special Concern
Buteo lineatus	Red-shouldered Hawk	Birds	Special Concern
Sterna dougallii	Roseate Tern	Birds	Endangered
Rhodostethia rosea	Ross's Gull	Birds	Threatened
Centrocercus urophasianus urophasianus	Greater Sage-Grouse urophasianus subspecies	Birds	Endangered
Oreoscoptes montanus	Sage Thrasher	Birds	Endangered
Asio flammeus	Short-eared Owl	Birds	Special Concern
Anthus spragueii	Sprague's Pipit	Birds	Threatened
Falco peregrinus tundrius	Peregrine Falcon tundrius subspecies	Birds	Special Concern
Icteria virens auricollis	Yellow-breasted Chat auricollis (British Columbia population)	Birds	Endangered
Picoides albolarvatus	White-headed Woodpecker	Birds	Endangered
Grus americana	Whooping Crane	Birds	Endangered
Coturnicops noveboracensis	Yellow Rail	Birds	Special Concern
Desmognathus ochrophaeus	Allegheny Mountain Dusky Salamander	Amphibians	Threatened
Plethodon idahoensis	Coeur d'Alene Salamander	Amphibians	Special Concern
Bufo fowleri	Fowler's Toad	Amphibians	Threatened
Spea intermontana	Great Basin Spadefoot	Amphibians	Threatened
Bufo cognatus	Great Plains Toad	Amphibians	Special Concern
Ambystoma jeffersonianum	Jefferson Salamander	Amphibians	Threatened
Rana pipiens	Northern Leopard Frog (Southern Mountain population)	Amphibians	Endangered
Rana pipiens	Northern Leopard Frog (Western Boreal/Prairie populations)	Amphibians	Special Concern
Rana aurora	Red-legged Frog	Amphibians	Special Concern
Rana pretiosa	Oregon Spotted Frog	Amphibians	Endangered
Dicamptodon tenebrosus	Coastal Giant Salamander	Amphibians	Threatened
Ascaphus montanus	Rocky Mountain Tailed Frog	Amphibians	Endangered
Ambystoma texanum	Small-mouthed Salamander	Amphibians	Endangered
Satyrium behrii columbia	Behr's (Columbia) Hairstreak	Arthropods	Threatened
Euphyes vestris	Dun Skipper (Western population)	Arthropods	Threatened
Plebejus saepiolus insulanus	Island Blue	Arthropods	Endangered
Coenonympha tullia nipis	Maritime Ringlet	Arthropods	Endangered
Danaus plexippus	Monarch	Arthropods	Special Concern
Euphydryas editha taylori	Taylor's Checkerspot	Arthropods	Endangered

Limenitis weidemeyerii	Weidemeyer's Admiral	Arthropods	Special Concern
Castanea dentata	American Chestnut	Vascular Plants	Endangered
Frasera caroliniensis	American Columbo	Vascular Plants	Special Concern
Panax quinquefolius	American Ginseng	Vascular Plants	Endangered
Asplenium scolopendrium	American Hart's-tongue Fern	Vascular Plants	Special Concern
Justicia americana	American Water-willow	Vascular Plants	Threatened
Symphyotrichum anticostense	Anticosti Aster	Vascular Plants	Threatened
Armeria maritima ssp. interior	Athabasca Thrift	Vascular Plants	Special Concern
Trichophorum planifolium	Bashful Bulrush	Vascular Plants	Endangered
Symphyotrichum subulatum	Bathurst Aster (Bathurst population)	Vascular Plants	Special Concern
Triphysaria versicolor ssp. versicolor	Bearded Owl-clover	Vascular Plants	Endangered
Sanicula arctopoides	Bear's-foot Sanicle	Vascular Plants	Endangered
Viola pedata	Bird's-foot Violet	Vascular Plants	Endangered
Fraxinus quadrangulata	Blue Ash	Vascular Plants	Special Concern
Buchnera americana	Bluehearts	Vascular Plants	Endangered
Woodsia obtusa	Blunt-lobed Woodsia	Vascular Plants	Endangered
Isoëtes bolanderi	Bolander's Quillwort	Vascular Plants	Special Concern
Bartonia paniculata ssp. paniculata	Branched Bartonia	Vascular Plants	Threatened
Phegopteris hexagonoptera	Broad Beech Fern	Vascular Plants	Special Concern
Buchloë dactyloides	Buffalograss	Vascular Plants	Threatened
Rosa setigera	Climbing Prairie Rose	Vascular Plants	Special Concern
Dryopteris arguta	Coastal Wood Fern	Vascular Plants	Special Concern
Aletris farinosa	Colicroot	Vascular Plants	Threatened
Ptelea trifoliata	Common Hoptree	Vascular Plants	Threatened
Symphyotrichum prenanthoides	Crooked-stem Aster	Vascular Plants	Threatened
Nephroma occultum	Cryptic Paw	Lichens	Special Concern
Magnolia acuminata	Cucumber Tree	Vascular Plants	Endangered
Vaccinium stamineum	Deerberry	Vascular Plants	Threatened
Balsamorhiza deltoidea	Deltoid Balsamroot	Vascular Plants	Endangered
Liatris spicata	Dense Blazing Star	Vascular Plants	Threatened
Trillium flexipes	Drooping Trillium	Vascular Plants	Endangered
Celtis tenuifolia	Dwarf Hackberry	Vascular Plants	Threatened
Lilaeopsis chinensis	Eastern Lilaeopsis	Vascular Plants	Special Concern
Geum peckii	Eastern Mountain Avens	Vascular Plants	Endangered
Platanthera leucophaea	Eastern Prairie Fringed-orchid	Vascular Plants	Endangered

Opuntia humifusa	Eastern Prickly Pear Cactus	Vascular Plants	Endangered
Isoëtes engelmannii	Engelmann's Quillwort	Vascular Plants	Endangered
Carex lupuliformis	False Hop Sedge	Vascular Plants	Endangered
Enemion biternatum	False Rue-anemone	Vascular Plants	Special Concern
Salix silicicola	Felt-leaf Willow	Vascular Plants	Special Concern
Braya fernaldii	Fernald's Braya	Vascular Plants	Threatened
Astragalus robbinsii var. fernaldii	Fernald's Milk-vetch	Vascular Plants	Special Concern
Tanacetum huronense var. floccosum	Floccose Tansy	Vascular Plants	Special Concern
Pedicularis furbishiae	Furbish's Lousewort	Vascular Plants	Endangered
Agalinis gattingeri	Gattinger's Agalinis	Vascular Plants	Endangered
Epipactis gigantea	Giant Helleborine	Vascular Plants	Special Concern
Lophiola aurea	Golden Crest	Vascular Plants	Threatened
Castilleja levisecta	Golden Paintbrush	Vascular Plants	Endangered
Hydrastis canadensis	Goldenseal	Vascular Plants	Threatened
Arisaema dracontium	Green Dragon	Vascular Plants	Special Concern
Symphyotrichum laurentianum	Gulf of St. Lawrence Aster	Vascular Plants	Threatened
Dalea villosa var. villosa	Hairy Prairie-clover	Vascular Plants	Threatened
Oxytropis lagopus	Hare-footed Locoweed	Vascular Plants	Special Concern
Plantago cordata	Heart-leaved Plantain	Vascular Plants	Endangered
Potamogeton hillii	Hill's Pondweed	Vascular Plants	Special Concern
Pycnanthemum incanum	Hoary Mountain-mint	Vascular Plants	Endangered
Eleocharis equisetoides	Horsetail Spike-rush	Vascular Plants	Endangered
Carex juniperorum	Juniper Sedge	Vascular Plants	Endangered
Gymnocladus dioicus	Kentucky Coffee-tree	Vascular Plants	Threatened
Isotria verticillata	Large Whorled Pogonia	Vascular Plants	Endangered
Achillea millefolium var. megacephalum	Large-headed Woolly Yarrow	Vascular Plants	Special Concern
Braya longii	Long's Braya	Vascular Plants	Endangered
Scirpus longii	Long's Bulrush	Vascular Plants	Special Concern
Calochortus lyallii	Lyall's Mariposa Lily	Vascular Plants	Threatened
Deschampsia mackenzieana	Mackenzie Hairgrass	Vascular Plants	Special Concern
Limnanthes macounii	Macoun's Meadowfoam	Vascular Plants	Threatened
Azolla mexicana	Mexican Mosquito-fern	Vascular Plants	Threatened
Juncus caesariensis	New Jersey Rush	Vascular Plants	Special Concern
Triphora trianthophora	Nodding Pogonia	Vascular Plants	Endangered
Pseudocyphellaria rainierensis	Oldgrowth Specklebelly	Lichens	Special Concern

Cephalanthera austiniae	Phantom Orchid	Vascular Plants	Threatened
Coreopsis rosea	Pink Coreopsis	Vascular Plants	Endangered
Polygala incarnata	Pink Milkwort	Vascular Plants	Endangered
Cirsium pitcheri	Pitcher's Thistle	Vascular Plants	Endangered
Sabatia kennedyana	Plymouth Gentian	Vascular Plants	Threatened
Lupinus lepidus var. lepidus	Prairie Lupine	Vascular Plants	Endangered
Erigeron philadelphicus var. provancheri	Provancher's Fleabane	Vascular Plants	Special Concern
Liparis liliifolia	Purple Twayblade	Vascular Plants	Endangered
Morus rubra	Red Mulberry	Vascular Plants	Endangered
Lachnanthes caroliana	Redroot	Vascular Plants	Threatened
Solidago riddellii	Riddell's Goldenrod	Vascular Plants	Special Concern
Bartramia stricta	Rigid Apple Moss	Mosses	Endangered
Smilax rotundifolia	Round-leaved Greenbrier (Great Lakes Plains population)	Vascular Plants	Threatened
Salix brachycarpa var. psammophila	Sand-dune Short-capsuled Willow	Vascular Plants	Special Concern
Ammannia robusta	Scarlet Ammannia	Vascular Plants	Endangered
Corydalis scouleri	Scouler's Corydalis	Vascular Plants	Threatened
Lotus formosissimus	Seaside Birds-foot Lotus	Vascular Plants	Endangered
Hypogymnia heterophylla	Seaside Bone	Lichens	Special Concern
Heterodermia sitchensis	Seaside Centipede	Lichens	Endangered
Solidago speciosa	Showy Goldenrod	Vascular Plants	Endangered
Quercus shumardii	Shumard Oak	Vascular Plants	Special Concern
Agalinis skinneriana	Skinner's Agalinis	Vascular Plants	Endangered
Lespedeza virginica	Slender Bush-clover	Vascular Plants	Endangered
Halimolobos virgata	Slender Mouse-ear-cress	Vascular Plants	Threatened
Cypripedium candidum	Small White Lady's-slipper	Vascular Plants	Endangered
Isotria medeoloides	Small Whorled Pogonia	Vascular Plants	Endangered
Lipocarpha micrantha	Small-flowered Lipocarpha	Vascular Plants	Endangered
Tripterocalyx micranthus	Small-flowered Sand-verbena	Vascular Plants	Endangered
Chenopodium subglabrum	Smooth Goosefoot	Vascular Plants	Special Concern
Yucca glauca	Soapweed	Vascular Plants	Threatened
Adiantum capillus-veneris	Southern Maidenhair Fern	Vascular Plants	Endangered
Chimaphila maculata	Spotted Wintergreen	Vascular Plants	Endangered
Hibiscus moscheutos	Swamp Rose-mallow	Vascular Plants	Special Concern
Clethra alnifolia	Sweet Pepperbush	Vascular Plants	Special Concern
Actaea elata	Tall Bugbane	Vascular Plants	Endangered

Psilocarphus elatior	Tall Woolly-heads (Pacific population)	Vascular Plants	Endangered
Drosera filiformis	Thread-leaved Sundew	Vascular Plants	Endangered
Cryptantha minima	Tiny Cryptanthe	Vascular Plants	Endangered
Rotala ramosior	Toothcup	Vascular Plants	Endangered
Eleocharis tuberculosa	Tubercled Spike-rush	Vascular Plants	Threatened
Arnoglossum plantagineum	Tuberous Indian-plantain	Vascular Plants	Special Concern
Salix turnorii	Turnor's Willow	Vascular Plants	Special Concern
Polemonium vanbruntiae	Van Brunt's Jacob's-ladder	Vascular Plants	Threatened
Gentianopsis procera ssp. macounii var. victorinii	Victorin's Gentian	Vascular Plants	Threatened
Cicuta maculata var. victorinii	Victorin's Water-hemlock	Vascular Plants	Special Concern
Tephrosia virginiana	Virginia Goat's-rue	Vascular Plants	Endangered
Hydrocotyle umbellata	Water-pennywort	Vascular Plants	Threatened
Ranunculus alismaefolius var. alismaefolius	Water-plantain Buttercup	Vascular Plants	Endangered
Iris missouriensis	Western Blue-flag	Vascular Plants	Threatened
Platanthera praeclara	Western Prairie Fringed-orchid	Vascular Plants	Endangered
Symphyotrichum sericeum	Western Silvery Aster	Vascular Plants	Threatened
Tradescantia occidentalis	Western Spiderwort	Vascular Plants	Threatened
Gentiana alba	White Prairie Gentian	Vascular Plants	Endangered
Eurybia divaricata	White Wood Aster	Vascular Plants	Threatened
Sericocarpus rigidus	White-top Aster	Vascular Plants	Threatened
Camassia scilloides	Wild Hyacinth	Vascular Plants	Threatened
Symphyotrichum praealtum	Willowleaf Aster	Vascular Plants	Threatened
Stylophorum diphyllum	Wood-poppy	Vascular Plants	Endangered
Viola praemorsa ssp. *praemorsa*	Yellow Montane Violet	Vascular Plants	Threatened
Elaphe obsoleta	Eastern Ratsnake	Reptiles	Threatened
Coluber constrictor foxii	Blue Racer	Reptiles	Endangered
Thamnophis butleri	Butler's Gartersnake	Reptiles	Threatened
Elaphe gloydi	Eastern Foxsnake	Reptiles	Threatened
Heterodon platirhinos	Eastern Hog-nosed Snake	Reptiles	Threatened
Sistrurus catenatus	Massasauga rattlesnake	Reptiles	Threatened
Phrynosoma hernandesi	Greater Short-horned Lizard	Reptiles	Special Concern
Coluber constrictor flaviventris	Eastern Yellow-bellied Racer	Reptiles	Threatened
Eumeces fasciatus	Five-lined Skink	Reptiles	Special Concern
Hypsiglena torquata	Nightsnake	Reptiles	Endangered
Eumeces septentrionalis	Prairie Skink	Reptiles	Endangered

Regina septemvittata	Queen Snake	Reptiles	Threatened
Contia tenuis	Sharp-tailed Snake	Reptiles	Endangered
Clemmys guttata	Spotted Turtle	Reptiles	Endangered
Glyptemys insculpta	Wood Turtle	Reptiles	Special Concern

Appendix B – The Federal Geographic Data Committee legend for land classes in Canada.

<u>Tree Dominated</u>
Land dominated by vegetation with a tree (woody plants with a height exceeding approximately 5 metres in most cases) crown density (percentage of the surface covered by projected tree crown perimeters) greater than 25%.

I. **Closed tree canopy** (crown cover >60%)
Trees with their crowns generally overlapping.

A. **Evergreen forest** (>75% of the total tree cover)

8. Temperate or subpolar needle-leaved evergreen closed tree canopy

/1/ Mature to old tree canopy (> ~ 60 y)
Softwood forest primarily found in the southern boreal forest and mountainous regions of western Canada. This class occasionally includes stands with fewer than 75% conifers, with the higher proportion of bodies of water (lakes) spectrally counteracting the higher percentage of hardwoods and more open areas. Three subgroups of spectral clusters with different prevailing canopy characteristics could be distinguished in VGT data:

<u>Irregular</u>
After disturbances in the proximity of more regular high crown density coniferous forest classes. This class is normally adjacent to class /1/ as described above, but with a lower density.

<u>Balsam Fir dominated</u>
Dominant species are different from class [1] above. In Quebec this condition occurs adjacent to mixed forest; balsam fir (as opposed to black spruce) is the dominant species but broadleaf trees also occur more frequently. This class also often includes a younger forest canopy and is regularly adjacent to class /2/.

<u>Conifers with broadleaf</u>
Often younger stands of high to medium density, with a broadleaf component; occasionally broadleaf component can be high shrubs.

/2/ Young tree canopy (~ 30-40 y)
This class contains concentrations of young softwood forest. It occurs primarily in the western coastal region.

B. **Deciduous closed tree canopy** (>75% of the total tree cover)

2. Cold-deciduous closed tree canopy

/3/ Deciduous canopy

Concentrated occurrence of deciduous broadleaf forest, generally with high crown density. In eastern Canada, this class is especially composed of tolerant hardwood species (maple, yellow birch, etc.).

Deciduous with conifers
An additional category that could be distinguished; due to the coarse resolution, the majority of the hardwood forest established elsewhere in Canada contains some conifers.

C. **Mixed evergreen-deciduous closed tree canopy** (25-75% evergreen or deciduous)

3. Mixed needle-leaved evergreen - cold-deciduous closed tree canopy
Land occupied by forest containing 25-75% evergreen needleleaf or deciduous broadleaf trees (determined as the percentage of the number of the trees present, not as tree crown density). Due to the low resolution of the data, pixels may contain a mosaic of needleleaf and broadleaf cover types.

Mixed coniferous (50-75% coniferous trees)
Mixed forest with the proportion of evergreen needleleaf trees above approximately 50% (as % of all trees present).

/4/ Mature to old tree canopy (> ~ 60 y)
This class is found primarily in the southern half of the boreal forest where conifers are more abundant in mixed forest. Occasionally it may contain a higher proportion of needleleaf trees (>75% of the tree population) in younger canopies because in these cases, higher reflectance of the young needleleaf trees compensates for the higher reflectance of broadleaf trees in older stands.

/5/ Young tree canopy (~ 30-40 y)
This class occurs primarily in the western coastal region (especially Vancouver island). Few patches elsewhere indicate young tree canopies.

Mixed deciduous forest (25-50% coniferous trees)
/6/ Mixed deciduous forest
This mixed class is found primarily in the southern half of the boreal forest, between classes /1/ and /3/; the proportion of needleleaf trees may be higher in young stands (higher reflectance of the young needleleaf trees compensates for the higher reflectance of broadleaf trees in older stands).

II. **Open tree canopy** (crown cover 25-60%)
Open stands of trees with crowns not usually touching.

A. **Evergreen open tree canopy** (>75% of the total tree cover)

4. Temperate or subpolar needle-leaved evergreen open tree canopy

Medium crown density (40-60%)
Evergreen needleleaf forest with crown density of the needleleaf species approximately 40-60%. In the VGT and similar coarse resolution data, the pixels may include a mosaic of denser and thinner tree cover.

/7/ Moss-shrub understory
Primarily found in the central part of the boreal region; mosses and shrubs dominate in the understory.

/8/ Lichen-shrub understory
Lichens (with shrubs) are commonly present in the understory; this class occurs particularly in central-northern Quebec.

Low crown density (25-40%)
Evergreen forest with crown density of the needleleaf species approximately 25-40%. Due to the coarse resolution of the VGT data, pixels may contain a mosaic of higher and lower tree cover, including openings such as cutovers or others.

/9/ Shrub-moss understory
As class /7/, this class is primarily found in the central portion of the boreal region. Commonly located on poorer sites with a proportion of broadleaf vegetation (mostly shrubs), it occasionally includes areas of post-fire mixed young cover. Two subgroups of spectral clusters with different cover characteristics could be distinguished in the VGT data:

/10/ Lichen (rock) understory
Particularly common in the northern boreal forest of Quebec (taiga), this class is characterized by the presence of lichen in the understory.

B. **Deciduous open tree canopy** (>75% of the total tree cover)

2. Cold-deciduous open tree canopy

/11/ Medium to low density broadleaf cover
/12/ Low regenerating to young broadleaf cover

C. **Mixed evergreen-deciduous open tree canopy** (25-75% evergreen or deciduous)

3. Mixed needle-leaved evergreen - cold-deciduous open tree canopy

Mixed coniferous (50-75% coniferous trees)

/13/ Medium to low density
This class often occurs as regenerating cover after disturbances.

Mixed deciduous (25-50% coniferous trees)

/14/ Medium to low density
This class exists in the southern boreal forest where it tends to be physically adjacent to /13/, but has a lower fraction of conifers. It contains mixed stands with a more irregular canopy structure owing to fairly old disturbances.

/15/ Low regenerating to young mixed cover
This class contains mostly a mosaic of mixed canopies and openings after disturbance.

Shrub Dominated

III. **Shrubland**

B. **Deciduous shrubland** (>75% of the total cover)

2. Cold-deciduous shrubland

/16/ High-low shrub dominated

Herb Dominated

V. **Herbaceous vegetation**

A. **Perennial graminoid vegetation**

5. Temperate or subpolar grassland

/17/ Grassland, prairie region
Areas covered with herbaceous vegetation with less than 10% tree or shrub cover. In this VGT land cover map, this class is limited to the prairie region (south of the boreal forest).

/18/ Herb-shrub-bare cover, mostly after disturbances
Variable herb-shrub-bare cover, mostly occupying old burns in the central part of the boreal region.

/19/ Shrubs-herb-lichen-bare
Open forest areas in the northern boreal (taiga) region where lichen has appreciable effect on spectral reflectance.

/20/ Wetlands
This class consists primarily of large wetlands (ex., peat bogs). Occasionally, the class also contains parts of formerly disturbed areas where the present ground cover is almost identical spectrally (shrubs and herbaceous vegetation).

6. Temperate or subpolar grassland with a sparse tree layer

/21/ Shrub-herb-lichen cover
Treed (coniferous trees with density <25 %) barren land in which lichen exerts some effect on the reflectance; this class occurs mainly in the northern boreal region.

/22/ Herb-shrub cover
Treed (coniferous trees with density <25 %) barren land of the northern boreal region in which shrubs and herbs form the dominant cover.

Tundra only:
Barren land occurring north of the treeline, but also in mountainous areas.

9. Polar grassland

/23/ Herb-shrub
Herbaceous vegetation and shrubs dominate; this class is close to class /24/.

10. Polar grassland with sparse shrub layer

/24/ Shrub-herb-lichen-bare
Shrubs and herbaceous vegetation are more abundant in favourable sites, such as along certain valleys.

/25/ Shrub-herb-lichen-bare, water bodies
A mixture of cover types (close to class /24/), with increased proportion of small water bodies.

11. Polar grassland with a sparse dwarf-shrub layer

/26/ Lichen-shrub-herb, bare soil or rock outcrops
This class, which is particularly widespread in the tundra, is typically made up of shrubs, lichen, herbaceous vegetation, bare soil and rock outcrops.

/27/ Lichen-shrub-herb, bare soil or rock outcrops, water bodies
This class is close to class /26/ but contains numerous water bodies.

/28/ Low vegetation cover (bare soil, rock outcrops)
Barrens dominated by rock outcrops and bare soil, particularly abundant at high altitudes in mountainous regions (ex., in the Rockies).

/29/ Low vegetation cover, with snow

D. **Annual graminoid or forb vegetation**

Mosaic land
Land containing a mixture of cropland, forest, shrubland, grassland or built-up areas in which no one component comprises more than about 70% (by area) of the landscape.

/30/ Woodland-cropland
Mosaic land in which tree cover (mostly broadleaf species) and shrubs are more prevalent than cropland.

/31/ Cropland-woodland
Mosaic land in which cropland is more prevalent than forest cover (mostly broadleaf species).

2. Temperate or subpolar annual grasslands or forb vegetation

b. Annual row-crop forbs and grasses
Areas dominated by herbaceous crops (usually annual crops) that may also contain a small percentage (under 10%) of trees and shrubs.

/32/ High biomass
Farmland dominated by high-biomass crops owing to the type of crop (ex., corn) or the climate (adequate precipitation).

/33/ Medium biomass
Farmland dominated by medium-biomass crops due to the type of crop or climate (subhumid). This class is mainly present in the Prairie region.

/34/ Low biomass
Farmland dominated by low-biomass crops owing to the type of crop (ex., grains) or the climate (semi-arid region). This class is present in the Prairie region.

Nonvascular Dominated

B. **Lichen vegetation**

1. Temperate or subpolar lichen vegetation

These classes may also be present as small patches in montane areas.

/35/ Lichen barren

/36/ Lichen-shrub-herb-bare ground

/37/ Lichen-shrub-herb cover

Open forest areas (density of coniferous trees <25 %) where lichen is particularly abundant. This class is especially common in northern Quebec and in the Northwest Territories.

Vegetation Not Dominant

VII. **Sparse vegetation** (cover 1-10%)

A. **Consolidated rock sparse vegetation**

2. Sparsely vegetated pavement

/38/ Rock outcrops, low vegetation cover

C. **Unconsolidated material sparse vegetation**

4. Sparsely vegetated soil flats

The following classes are characterized by sparse vegetation cover after disturbance (mainly fire burns). The vegetation cover characteristics vary with the age and the intensity of the burn. Since some low productivity sites are also covered with sparse low vegetation, spectral confusion may occur at coarse (VGT) or fine resolution, but the classes are labelled separately in this classification to preserve the burn patterns as an important environmental feature.

/39/ Recent burns

Areas burned in recent (< ~ 5) years. The sparse vegetation present is distributed according to site quality and the intensity of the fire. Standing dead trees are often present.

/40/ Mostly bare disturbed areas (e. x., cutovers)

Recent forest cutovers are generally included in this class; urban and built-up areas can also be spectrally similar.

/41/ Low vegetation cover

Naturally low productivity sites or sites after disturbances (mainly burns) that occurred within the last ~10 years.

VIII. **Non-Vegetated** (cover <1%)

/42/ Urban and built-up

This is a land use category. It can be inferred from fine resolution data such as TM. In coarse resolution images, surface cover is a mixture of green vegetation and bare ground-type materials (asphalt, concrete, soil) and it is not always possible to unambiguously separate all built-up areas. In coarse resolution data it spectrally resembles vegetated areas after disturbance or northern cover types

/43/ Water bodies

Area covered with water.

/44/ Mixes of water and land
This class primarily contains certain elements of the image (pixels) partly covered with water, as well as a few areas recently ravaged by fire.

/45/ Snow/ice
Area permanently covered with ice or snow.

/46/ Clouds

Appendix C – List of all acronyms used in the thesis.

AI	Applicability index
AMUSE	Automated temporal Updating through Signature Extension
ASTER	Advanced spaceborne thermal emission and reflection radiometer
AUC	Area under the curve
CBD	Convention on Biological Diversity
COSEWIC	Committee on the Status of Endangered Wildlife in Canada
DEM	Digital elevation model
FGDC	Federal Geographic Data Committee
GIS	Geographic information system
GPE	Greater park ecosystem
HRG	High resolution geometric
HRV	High resolution visible
Maxent	Maximum Entropy
OSNAP	Operational Species Niche Assessment Plan
SARA	Species At Risk Act
SLINP	St. Lawrence Islands National Park
SPOT	Satellite Pour l'Observation de la Terre
TM	Thematic Mapper