ncasi

NATIONAL COUNCIL FOR AIR AND STREAM IMPROVEMENT

THE ROLE OF FOREST MANAGEMENT IN MAINTAINING CONSERVATION VALUES

TECHNICAL BULLETIN NO. 983

APRIL 2011

by

Dr. Darren J.H. Sleep
NCASI
Montreal, Quebec

Kirsten Vice
NCASI
Montreal, Quebec

Dr. T. Bently Wigley
NCASI
Clemson, South Carolina

Acknowledgments

The authors gratefully acknowledge the review and comments provided by Dr. Alan Lucier, Senior Vice President of NCASI.

For more information about this research, contact:

Darren J.H. Sleep, Ph.D.
Senior Forest Ecologist
NCASI
P.O. Box 1036, Station B
Montreal, Quebec H3B 3K5 Canada
(514) 286-9690
dsleep@ncasi.org

Kirsten Vice
Vice President, Canadian Operations
NCASI
P.O. Box 1036, Station B
Montreal, QC H3B 3K5 Canada
(514) 286-9111
kvice@ncasi.org

For information about NCASI publications, contact:

Publications Coordinator
NCASI
P.O. Box 13318
Research Triangle Park, NC 27709-3318
(919) 941-6400
publications@ncasi.org

Cite this report as:

National Council for Air and Stream Improvement, Inc. (NCASI). 2011. *The role of forest management in maintaining conservation values*. Technical Bulletin No. 983. Research Triangle Park, N.C.: National Council for Air and Stream Improvement, Inc.

ncasi

serving the environmental research needs of the forest products industry since 1943

PRESIDENT'S NOTE

The diversity of living things, or "biodiversity," has become one of the prime features of our biosphere that society values. However, quantifying, assessing and describing the goals and values of biodiversity conservation and quantifying human effects on biodiversity can be daunting. Biodiversity, irrespective of its value, is a concept that is difficult to define and challenging to measure.

One method by which biodiversity values can be indirectly measured is through the use of indicators and criteria. Concepts such as "fragmentation", "endemism", and "representation" are useful in that they offer a framework under which effects of human intervention can be measured, and areas can be identified for conservation that are likely to help maintain biodiversity at some scale. To help understand these issues, NCASI participated in a review of criteria and indicators, which subsequently resulted in a 2005 publication by the Yale School of Forestry that reviewed the criteria used by global conservation organizations in the designation of conservation priorities around biodiversity.

This report extends that earlier work. Here, we review these same conservation criteria, but examine them on a scientific basis, where it is appropriate to do so. Some criteria used are directly measurable and quantifiable, and are directly linked with certain aspects of biodiversity. Other criteria tend to be relative, and are more connected with social norms and desires, or public perceptions, rather than with ecological or environmental sciences. Where possible, the report clearly distinguishes the two influences (ecological/environmental or social), and links each criterion based in science with the supporting literature.

Finally, the report links conservation criteria with forest management and aspects of forest research and utilization that are designed to enhance achievement of these criteria and hence aid in the long-term conservation of biodiversity. The report also documents where information is lacking and research gaps exist, in the hopes of raising new questions and research programs to meet these needs. While the report focuses primarily on forest management and its relationship to biodiversity in a Canadian context, the concepts and lessons have broad application wherever sustainable forest management is applied.

Ronald A. Yeske

April 2011

NOTE DU PRÉSIDENT

La diversité des êtres vivants, ou « biodiversité », est devenue l'une des principales caractéristiques de la biosphère que valorise la société. Cependant, la quantification, l'évaluation et la description des objectifs et des valeurs de conservation et la quantification de notre impact sur la biodiversité peuvent s'avérer un défi de taille. La biodiversité, quelle que soit sa valeur, est un concept difficile à définir et à mesurer.

Il est possible de mesurer indirectement les valeurs de la biodiversité en faisant appel à des indicateurs et à des critères. Les concepts tels que « fragmentation », « endémisme » et « représentation » sont utiles, car ils offrent un cadre de travail à l'intérieur duquel il est possible de mesurer les effets de l'intervention humaine et de définir des aires de conservation susceptibles de maintenir la biodiversité à une certaine échelle. Pour mieux comprendre ces questions, NCASI a participé à une revue de critères et d'indicateurs utilisés par les organismes internationaux de conservation pour définir les priorités de conservation reliées à la biodiversité que le *Yale School of Forestry* avait déjà analysés et qui a subséquemment donné lieu à la publication de ces critères et indicateurs en 2005.

La présente étude est un prolongement de ces travaux antérieurs.Nous passons en revue ces mêmes critères de conservation, mais nous les évaluons sur une base scientifique lorsqu'ils se prêtent à une telle évaluation. Certains critères sont directement mesurables et quantifiables et ont un rapport direct avec certains aspects de la biodiversité. D'autres critères sont plutôt relatifs et sont davantage reliés aux normes et aux désirs sociaux ou aux perceptions du public qu'aux connaissances scientifiques en matière d'écologie et d'environnement. Dans la mesure du possible, nous séparons explicitement ces deux influences (écologique/environnementale ou sociale) dans le rapport. Si le critère repose sur des bases scientifiques, nous l'avons relié au document pertinent dans la littérature.

Finalement, nous relions les critères de conservation à l'aménagement forestier et aux aspects de la recherche en foresterie et de l'utilisation des forêts qui permettraient de respecter ces critères et, par conséquent, de contribuer à préserver la biodiversité à long terme. Dans le présent rapport, nous faisons également ressortir les lacunes en matière d'information et de recherche dans l'espoir de soulever de nouvelles questions qui donneront lieu à la mise sur pied de programmes de recherche destinés à combler ces besoins. Bien que cette étude mette l'accent sur le lien qui existe entre l'aménagement forestier et la biodiversité dans un contexte canadien, les concepts et les conclusions de cette étude peuvent s'appliquer plus largement dans tout contexte lié à l'aménagement forestier.

Ronald A. Yeske

Avril 2011

THE ROLE OF FOREST MANAGEMENT IN MAINTAINING CONSERVATION VALUES

TECHNICAL BULLETIN NO. 983
APRIL 2011

ABSTRACT

Biodiversity is a key concept in conservation biology and a prime target for conservation efforts across the globe. In North America forest management planning and operations have been under intense scrutiny to ensure the maintenance and sometimes enhancement of biodiversity in every area of operation. Forest management policy and guidelines and public and private forest certification schemes have been developed to take biodiversity into consideration, and contribute to extensive efforts to plan for the adequate protection of biodiversity. However, biodiversity is difficult to conceptualize, and therefore more difficult to quantify. As a result, numerous criteria and indicators have been developed and selected that if measured and maintained on a landscape, would be expected to conserve biodiversity. Based on a report by the Yale School of Forestry and Environmental Studies, the following 12 criteria were examined:

1. Representation
2. Species richness
3. Species endemism
4. Rarity
5. Significant or outstanding ecological or evolutionary processes
6. Presence of special species or taxa
7. Threatened species
8. Species decline
9. Habitat loss
10. Fragmentation
11. Large intact areas
12. High and low future threat.

The purpose of this report is to link a selection of these criteria to their scientific underpinnings, by examining the published scientific literature that underscores them. The basis for each criterion (ecological/environmental or social) is described, and where notable, uncertainties are noted. Findings suggest that there are at least three categories into which the criteria can be examined (species, landscape, and future threat), relatively few can be quantified in a meaningful way, and most of them are significantly inter-related and confounded. Finally, the report highlights a number of research areas that need to be explored to better link forest management to biodiversity.

KEYWORDS

biodiversity, criteria, endangered species, endemism, forest management, fragmentation, habitat, indicators, intact areas, rarity, representation, threat, values

RELATED NCASI PUBLICATIONS

Technical Bulletin No. 885. (August 2004). *Managing Elements of Biodiversity in Sustainable Forestry Programs: Status and Utility of NatureServe's Information Resources to Forest Managers*.

LE RÔLE DE L'AMÉNAGEMENT FORESTIER DANS LE MAINTIEN DES VALEURS DE CONSERVATION

BULLETIN TECHNIQUE N^{O} 983
APRIL 2011

RÉSUMÉ

La biodiversité est un concept important dans la biologie de la conservation et une priorité dans les efforts de conservation à l'échelle mondiale. En Amérique du Nord, on scrute à la loupe les plans d'aménagement et les activités d'exploitation de la forêt pour s'assurer du maintien et quelquefois de l'enrichissement de la biodiversité dans toutes les activités d'exploitation. On a élaboré des politiques et des lignes directrices sur l'aménagement des forêts et développé des programmes publics et privés de certification forestière pour tenir compte de la biodiversité et contribuer aux nombreux efforts destinés à protéger adéquatement la biodiversité. Par contre, la biodiversité est une notion difficile à conceptualiser et, par conséquent, difficile à quantifier. C'est pourquoi on a choisi et défini des critères et des indicateurs qui devraient normalement préserver la biodiversité si on les mesure et on les respecte dans un paysage donné. Les douze (12) critères ci-dessous, tirés du rapport du *Yale School of Forestry and Environmental Studies*, font l'objet d'une analyse dans le présent rapport:

1. Représentation
2. Diversité des espèces
3. Endémisme des espèces
4. Rareté
5. Processus évolutifs ou écologiques importants ou exceptionnels
6. Présence d'espèces ou taxons uniques
7. Espèces menacées
8. Déclin d'une espèce
9. Perte d'habitat
10. Fragmentation
11. Grandes aires intactes
12. Menace élevée ou faible dans le futur

L'objectif de la présente étude est de relier un certain nombre de ces critères à leurs assises scientifiques en examinant la littérature scientifique publiée à leur sujet. Dans le présent rapport, on y décrit les fondements de chaque critère (écologique/environnemental ou social) et, lorsqu'elles sont importantes, on signale les incertitudes. Les conclusions de cette étude semblent indiquer qu'il existe au moins trois angles sous lesquels on peut examiner ces critères (espèce, paysage, et menace future), qu'il n'est pas possible de quantifier la plupart de ces critères de manière satisfaisante et que la plupart d'entre eux sont très étroitement liés entre eux et se confondent entre eux. Finalement, le rapport fait ressortir un certain nombre d'axes de recherche qu'il faudrait explorer pour mieux relier l'aménagement forestier à la biodiversité.

MOTS-CLÉS

aires intactes, aménagement forestier, biodiversité, critères, endémisme, espèce menacée, fragmentation, habitat, indicateurs, menace, rareté, représentation, valeurs

AUTRES PUBLICATIONS DE NCASI

Bulletin technique n° 885. (août 2004). *La gestion des éléments de biodiversité dans les programmes de foresterie durable: état de la situation et utilité des ressources de NatureServe pour les gestionnaires de forêts*

CONTENTS

TABLES

FIGURES

THE ROLE OF FOREST MANAGEMENT IN MAINTAINING CONSERVATION VALUES

1.0 INTRODUCTION AND OBJECTIVES

Forest management activities in Canada provide important economic and societal benefits, but also contribute to the conservation of terrestrial and aquatic biodiversity through implementation of strategies and approaches focused at a variety of scales. Provincial forest management guidelines are designed to ensure protection of conservation values such as rare, sensitive, and "at risk" species, riparian habitat, and water quality and quantity. At a broad scale, coarse-filter approaches within harvest and regeneration guidelines are used to emulate natural disturbance, maintain ecological processes, and conserve species diversity across space and time.

Within these approaches, two levels of planning are used to achieve these objectives. Long-term and landscape-scale forest management plans are developed to maintain native cover and habitat types and to conserve genetic diversity. Stand-level harvest and forest renewal plans are subsequently developed to contribute cumulatively to landscape biodiversity objectives by providing variable disturbance size classes, maintaining insular and peninsular forest patches, and maintaining varying amounts of snags and downed woody material within harvest areas. Both levels of planning are developed and implemented within an adaptive management context.

Progress towards conservation objectives is commonly assessed using criteria and indicators that provide insight into the state and trends of various aspects of biological diversity. Government regulatory frameworks and forest certification systems such as the Canadian Standards Association (CSA), the Forest Stewardship Council (FSC), and Sustainable Forestry Initiative® (SFI) promote and evaluate achievement of responsible forest stewardship and the conservation of biological diversity on the managed land base through the assessment of various indicators and criteria. Conservation organizations also use a variety of criteria and indicators within their respective approaches to identify areas for conservation emphasis and often promote or designate such areas as reserves. Gordon, Franco, and Tyrrell (2005) highlighted 12 criteria used by global conservation organizations to direct conservation efforts:

1. Representation
2. Species richness
3. Species endemism
4. Rarity
5. Significant or outstanding ecological or evolutionary processes
6. Presence of special species or taxa
7. Threatened species
8. Species decline
9. Habitat loss
10. Fragmentation
11. Large intact areas
12. High and low future threat

It is the purpose of this report to characterize the scientific basis for each of these criteria, to discuss the relationship between active forest management and each criterion, and therefore the contribution of sustainable forest management as a whole to conservation of biological diversity. We also describe industry research related to the criteria. While social aspects are relevant to the basis of, and management planning for, certain criteria, this report focuses solely on scientific aspects of criteria as they relate to quantifiable measures of biodiversity. The analysis uses the criteria identified by

Gordon, Franco, and Tyrrell (2005) as a foundation; however, other criteria have been incorporated where specific Canadian issues may warrant discussion (e.g., boreal forest).

2.0 BACKGROUND

The criteria to be examined in this report are detailed in Gordon, Franco, and Tyrrell (2005) who compared and contrasted eight conservation planning approaches promoted by five prominent scientifically based conservation environmental non-government organizations for guiding decisions about which areas to prioritize for conservation (Table 2.1). For each approach, the authors discuss organizational or partnership missions, planning principles, conservation targets, scientific criteria, and thresholds. The conservation planning approaches were selected on the basis of the following five criteria.

1. Approach relies primarily on scientifically based criteria.
2. Approach sets conservation priorities.
3. Approach applies at a global scale and identifies "where" to conserve.
4. Approach emphasizes a variety of taxa or, if it focuses on one type of organism, that the taxa serve as an indicator for broader biodiversity.
5. Organizational or partnership representatives were willing to participate and share details about their conservation priority setting processes at the level needed for this project.

Table 2.1 Organizations and Conservation Planning Approaches Characterized by Gordon, Franco, and Tyrrell (2005)

Organization or Partnership	Conservation Approaches Studied[a]
Alliance for Zero Extinction	• AZE Sites (epicenters of imminent (AZE) extinction
BirdLife International	• Endemic Bird Areas
	• Important Bird Areas
Conservation International	• Biodiversity Hotspots
	• High Biodiversity Wilderness Areas
Wildlife Conservation Society	• Range-wide Priority Setting
	• Last of the Wild
World Wildlife Fund	• Global 200

[a] In addition to the eight global approaches indicated above, cursory review is given to six regional approaches utilized by the African Wildlife Foundation, Conservation International, Ducks Unlimited Canada, The Nature Conservancy (TNC), the World Wildlife Fund, and the Wildlife Conservation Society.

Many of the criteria identified by Gordon, Franco, and Tyrrell (2005) are considered by conservation programs and can be addressed through management at a variety of spatial scales, from global to local. Given that some of the programs that employ the criteria are global but target local sites, this wide range of spatial scales is to be expected. However, for the purposes of this report, forest management's maximum spatial scale is considered to be the landscape. In Canada, this generally manifests itself in the large forested areas available for management by organizations, subject to provincial licensing agreements. Each province refers to these areas using slightly different designations [e.g., Forest Management Areas (FMAs) in Ontario; Timber Supply Areas (TSAs) in Alberta and BC; unité d'aménagement forestier (UAFs) in Quebec; Crown Timber Licenses (CTLs) in New Brunswick]. In each province, these areas also vary in size, but generally represent several hundred square kilometers (km^2) each. Such large management areas allow for development of long-term sustainable management plans and conservation of numerous biological values, while maintaining a sustainable fibre supply and suitably long rotation ages. Larger spatial scales beyond those of individual allocated forest management areas are considered by provincial governments, and are managed through provincial government policy.

Why Biodiversity?

The primary objective of the development and use of these (and other) conservation criteria has overwhelmingly been the preservation and promotion of biodiversity. As pointed out by Bunnell and Dunsworth (2009), it has been a challenge to define both the term "biodiversity" as well as what is meant by sustaining it, making it difficult to envision the desired management outcomes. They point out that "biodiversity" is an abstract concept that is usually treated as a thing. West (1994) described it as a "concept cluster".

Biodiversity has been variously defined, with over 90 published definitions by 1998 (Bunnell 1998; DeLong 1996). Most definitions are suitably broad, owing to the breadth and variety of biodiversity, depending on the ecological scale and level within the biological hierarchy examined (Cairns and Lackey 1992). For example, McNeely (1988) defined biodiversity as "the degree of nature's variety", whereas Hughes and Noss (1992) defined it as "the variety of life and its processes". Generally, biodiversity is recognized at four primary scales: 1) genetic diversity within a species; 2) species diversity in a given area; 3) ecosystem diversity, accounting for the diversity of ecological processes and habitats in a region; and 4) landscape diversity, or the spatial heterogeneity within a larger region (Noss 1989; Norse et. al. 1986; OTA 1987). In Canada, as in much of the world, public interest has held that biodiversity should be protected, with both national and international legislation and treaties aimed at protecting and reducing biodiversity loss [e.g., the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), 1979; the Canadian Species At Risk Act; (SARA), 2002].

When discussing the concept and importance of biodiversity, it is important to understand that the concept of biodiversity embraces both scientific and human values (Cairns and Lackey 1992). The benefits of conserving or enhancing biodiversity are thought to be various, but can be grouped into two broad categories: intrinsic values and anthropocentric values. The intrinsic value of biodiversity suggests that biodiversity must be preserved irrespective of its present or future use to benefit humanity, and rests on species' (and therefore biodiversity's) inherent right to exist (Ehrenfeld 1978). Taylor (1986) suggested that species have moral value of their own, independent of human use. Anthropocentric value refers to those values that are of direct or indirect benefit to humans. Benefits may be commercial and economic, or ecological in nature. For example, many plants are the source of essential chemicals for pharmaceuticals (Farnsworth 1988) and important sources of genetic variability for crops (Spears 1988). More directly, many species of plants and animals are used directly as sources of fibre (e.g., wood, cotton, wool) either through domestication or wild collection. Natural biota also supports a variety of ecological processes that are essential to humans, such as the purification of air and water, and the stabilization of soils. The public and the scientific community, therefore, place a high value on the preservation of ecosystems and the species within them.

Environmental/Ecological vs. Social Indicators

Criteria and indicators are used in conservation planning to monitor progress towards goals or targets related to achievement of conservation objectives. Often, a mix of science and societal values drives selection of biodiversity conservation goals (Czech and Krausman 2001) necessitating a mix of objectives as well as criteria/indicators (Tear et al. 2005; Margules and Pressey 2000). Thus, a conservation planning effort may include criteria/indicators that emphasize science and/or societal values (Johnson 1995) as well as criteria/indicators that are quantitative or qualitative (Sarkar et al. 2006).

In this analysis, we distinguish between environmental/ecological indicators and social indicators. We define an ecological indicator as any expression of the environment that provides quantitative information on one or more ecological resources; it is frequently based on discrete pieces of information that reflect the status of large systems (Hunsaker and Carpenter 1990). In contrast, we define societal indicators as being more qualitative, reflecting values that society or a segment of society places on various aspects of the environment. It is an ongoing challenge to integrate ecological indicators with social goals for resource management (Dale and Beyeler 2001; Schiller et al. 2001).

As noted by Lackey (2006), the importance of definitions cannot be overstated, as they can lead to unintended implications for policy. Many of the criteria discussed in this report by definition contain aspects that are societal and/or aesthetic in nature, and/or aspects that are of a scientific nature. Having more of one aspect than the other does not mean a particular criterion is less important, only that it contains aspects that are difficult to quantify empirically. Therefore, this report focuses on the scientific and measurable aspects of indicators rather than their social and aesthetic value.

Conservation organizations apply the criteria identified by Gordon, Franco, and Tyrrell (2005) primarily in the selection, planning, and effectiveness monitoring of protected areas. Protected areas (IUCN Categories I-IV) are generally managed for biodiversity values, with a management paradigm of nature protection (Duinker et al. 2010). Because biodiversity is difficult to quantify, criteria such as those examined here are used as surrogates for biodiversity. In theory, if these criteria can be maintained, biodiversity will be maintained, making the protected area in question an effective tool for conservation.

In contrast, for industrially managed forests, the dominant value may be timber production and its associated economic and societal benefits, with the management paradigm being sustainable forest management (Duinker et al. 2010). However, biodiversity conservation is also considered necessary for the maintenance of both the biological capacity of the landscape (e.g., ongoing production of timber, maintenance of water quality and quantity), and corporate social responsibility, as managers assess the effects of their activities on the environment, consumers, employees, communities, stakeholders, and all other members of the public sphere (IUCN Category VI). Therefore, if the criteria discussed in this report are effective surrogates of biodiversity, they may also be applied to forest management areas.

3.0 ANALYSIS OF CRITERIA

3.1 Species Richness

The basic concept of species richness is the number of species in a given area (Gordon, Franco, and Tyrrell 2005; Cairns and Lackey 1992), as well as the turnover of species between areas. The basis for this criterion is environmental/ecological in nature. “Biodiversity Hotspots”, or areas with very high species richness, are sometimes targeted as areas in significant need of protection (Reid 1998; Myers et al. 2000; Rohner and Krebs 1998).

Accurately and effectively assessing species richness presents various challenges. Depending on the species group in question, measuring total species richness over a large spatial scale (e.g., landscape or ecozone) requires substantial survey effort to reach a reasonable measure of precision (de Thoisy, Brosse, and Dubois 2008). There are a number of solutions to this difficulty, each with their own strengths and weaknesses.

One common approach is the use of surrogates or indices of species richness. Surrogate species or indicators may represent a range of biodiversity elements, and should have a number of attributes (see Noss 1989). In the case of species richness, a surrogate may be a species that is generally well detected and whose presence indicates the co-occurrence of many other species, and therefore high species richness [i.e., an "umbrella" species; Caro and O'Doherty (1999)]. Measuring the diversity of higher taxonomic orders (e.g., family richness) in an area has shown promise among several carefully selected groups [e.g., bats in Northern and Central America, British ferns and butterflies; Williams and Gaston (1994)]. However, some authors have questioned the effectiveness of surrogates for protecting conservation values, arguing that no single species seems to be more effective than any other (Andelman and Fagan 2000). Pearson and Carroll (1998) suggested that the use of such a bioindicator (e.g., richness of butterflies) was useful when combined with precipitation data, but that the usefulness of any particular indicator varied between continents.

Surrogates have also been proposed for use at various levels of biological organization, such as the regional landscape, community, ecosystem, population, species and genetic levels (Franklin 2003; Noss 1989). Because scale has a significant effect on the appropriate surrogate or indicator to use, and different aspects of biodiversity are manifested at different scales, having multiple surrogates or indicators for each scale is reasonable.

Another approach is the use of species accumulation curves (Rosenzweig 1995). It has long been known that if you sample a larger area, you will detect more species (Williams 1943). Therefore, it can be assumed that by collecting data on the number of newly observed species as samples accumulate, an extrapolation of the resulting curve to its asymptote will approximate species richness for the total area. This works remarkably well for systems in which the species are well known and readily located and identified, ensuring that it is possible to estimate the number of species present [e.g., breeding birds in Britain; Colwell and Coddington (1994)]. However, where this is not the case, there are a number of modifications to the approach that must be used to account for imperfect detection rates or cryptic species (e.g., Ugland, Gray, and Ellingsen 2003; Dorazio et al. 2006). Preston (1962a, 1962b) noted that such curves differ between various systems, such as small samples of single biotas, islands of one archipelago, or areas with differing evolutionary histories.

The most common means of assessing the contribution of managed areas to conserving species richness has often been to benchmark a managed area against a mature, unmanaged area (e.g., Arcese and Sinclair 1997; Sinclair 1998; Sinclair, Mduma, and Arcese 2002) of similar habitat and seral-stage compositions. There are several limitations to this approach, but most centre on the fundamental differences between harvest and natural disturbance, as well as the time since disturbance. Most studies examining the effects of natural versus harvest disturbance have found that many species of small mammals and cavity-nesting birds may breed in areas immediately after harvest and natural disturbance (NCASI 2009c), and vertebrate and plant communities tend to converge by about 30 years post-disturbance (Drapeau et al. 2000; Hobson and Schieck 1999; Simon, Schwab, and Otto 2002; NCASI 2006). Therefore, to adequately assess species richness on a managed landscape, comparisons to unmanaged landscapes (naturally disturbed) must account for time since disturbance (e.g., using a time-series approach) and habitat composition of each landscape by incorporating the full range of seral stages provided by management.

3.2 Species Endemism

Endemic species, or species of restricted geographic range and that often exist nowhere else, are sometimes used as an indicator of high-priority areas that should be targeted for conservation efforts (Gordon, Franco, and Tyrrell 2005). The basis for this criterion is environmental/ecological in nature. The prioritization of areas with many endemic species is logical in that the loss of such areas would likely lead to global extinctions of species.

However, such species-level approaches have been questioned, as many endemic species are restricted to very specific biotopes, and therefore may not occur in areas of high species richness (Thomas and Mallory 1985). Indeed, Prendergast et al. (1993), examining species richness patterns in the United Kingdom, suggested that areas with high species richness do not coincide with areas with high endemism, which could create conflicting conservation objectives. In contrast, Kerr (1996) found that species richness and endemism do coincide, as long as the analysis remains at one taxonomic level (e.g., mammals) and is conducted at a large scale (e.g., continent-wide). Similarly, Jetz, Rahbek, and Colwell (2004) found that areas of high species richness tended to be high in endemics, and suggested such regions may represent areas of special evolutionary history as centres of diversification. However, more recently, Orme et al. (2005) found that, at a global scale, richness hotspots, endemic hotspots, and threatened species hotspots coincided relatively infrequently (only 32 times were all three coincident).

For plants, the majority of endemic species are located at low latitudes (closer to the equator) with some pockets of endemics at mid-latitudinal ranges with semi-arid and seasonal climactic patterns (Cowling and Samways 1995). Many areas with endemic species, particularly in northern latitudes like Canada, are thought to be glacial refugia, or places where species survived during the glaciations of the Pleistocene epoch (2.6 million years to 12,000 years before present). These centres have been identified as centres of isolation, recolonization, and evolution. Argus (1976) and IUCN (1997) [as cited in Cannings et al. (2005)] have identified 12 such areas in Canada (Table 3.1).

Table 3.1 12 Areas of Glacial Refugia Identified in Canada (from Cannings et al. 2005)

Area	Jurisdiction
Ellesmere Island	Nunavut
Baffin Island	Nunavut
Central Arctic Islands	Yukon
Central Yukon	Yukon
MacKenzie Mountains	Northwest Territories, Yukon
Lake Athabaska dunes	Saskatchewan
Queen Charlotte Islands (Gwaii Haanas)	British Columbia
British Columbia (serpentine areas)	British Columbia
Western Newfoundland	Newfoundland and Labrador
Torngat Mountains (serpentine areas)	Québec, Newfoundland and Labrador
Rocky Mountains	Alberta, British Columbia
Gulf of St. Lawrence	Québec

Given the extent to which Canada was glaciated, there are relatively few (68) endemic species in Canada, 46 (67%) of which are vascular plants (see Appendix A for a full list). Areas with multiple endemic species are well known (Figure 3.1).

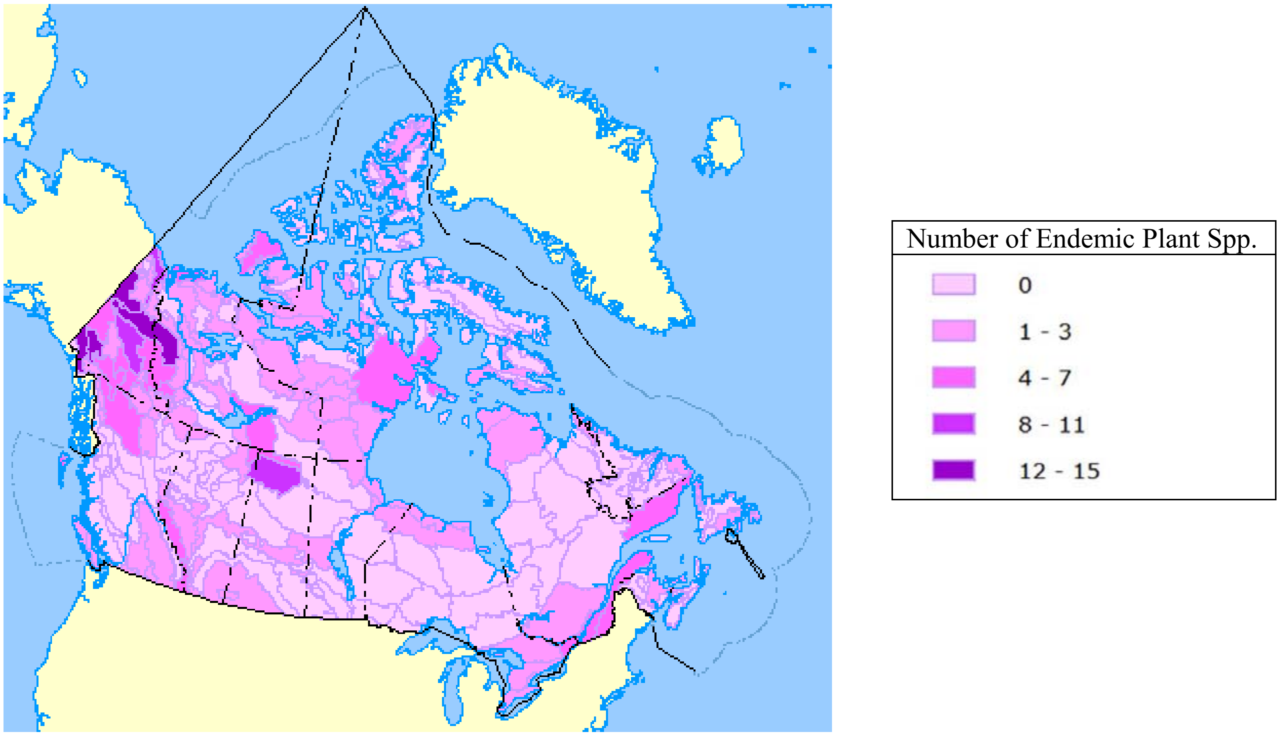

Figure 3.1 Map of the Endemic Plant Diversity in Canada (from Haber 1994)
[Dark areas indicate areas of high plant species endemism, suggesting glacial refugia.]

Forest management operations in Canada occur primarily in areas with low species endemism (most harvesting occurs in the Boreal Shield (0 plants), Pacific Maritime (1-3 plants) and Atlantic Maritime (1-3 plants) and Montane Cordillera (0 plants)). As a result, avoidance of high endemism areas by forest managers in Canada is not generally an issue. Furthermore, range extent is an important factor considered by organizations such as IUCN and NatureServe when assigning global conservation status to species (e.g., Master et al. 2009), and species with very restricted ranges are typically ranked as imperiled or critically imperiled or listed as threatened or endangered. Thus, companies operating under the auspices of sustainable forestry certification programs are committed to protecting occurrences of such species.

3.3 Rarity

Closely tied to endemic species are rare, scarce, or uncommon species. The basis for this criterion is environmental/ecological in nature. It logically suggests that in order to maintain biodiversity, species that are rare on the landscape, and therefore may be limited genetically or prone to extinction by catastrophic events, should be protected (Gordon, Franco, and Tyrrell 2005). A number of characteristics of rarity have been correlated with a higher probability of extinction, some of which may be synergistic (Davies, Margules, and Lawrence 2004). "Rarity Hotspots" are areas with above-normal numbers of rare species and fall under the broader conservation tool of "Biodiversity Hotspots" (Section 3.2).

Although rarity is a key factor considered when assessing extinction risk (Master et al. 2009), rarity does not inherently make a given species of high conservation value, and may be more related to the natural history, evolutionary pathway, and species ecology. Rabinowitz (1981) and Rabinowitz, Cairns, and Dillon (1986) suggested that there are at least seven different types of rarity, each with its own degree of conservation concern (Table 3.2). As pointed out by Rodrigues and Gaston (2002), species that appear to be rare (such as those on the edge of their range and more abundant at the core of their range, but see Sagarin and Gaines 2002) may be of lower conservation value than globally rare or endemic species. Further, a species may be rare naturally, having evolved in such a way that rarity is a necessary or consequential feature of its ecology, rather than rare owing to the effects of anthropogenic activity. Bunnell, Campbell, and Squires (2005) suggested that one or both of these factors were likely at play for the majority of rare and endangered species in British Columbia.

Table 3.2 Seven Types of Rarity (Rabinowitz 1981; Rabinowitz, Cairns, and Dillon 1986)

		Geographic Range			
		Large		Small	
Population Size	Somewhere Large[a]	Common.	Locally abundant over a large range in a specific habitat.	Locally abundant in several habitats, but restricted geographically.	Locally abundant in specific habitat, but restricted.
	Everywhere Small[a]	Constantly sparse over a large range and several habitats.	Constantly sparse in a specific habitat, but over a large range.	Constantly sparse and geographically restricted in several habitats.	Constantly sparse and restricted to a specific habitat.

[a] "Somewhere Large" and "Everywhere Small" refers to the global distribution of population sizes of a particular species. While on average, a species may be distributed in scattered small populations, the existence of one very large population (e.g., wildebeests on the Serengeti) changes the profile of a species' conservation status.

In addition to various types of rarity, "rarity" as a concept suffers from a number of confounding factors, making management based on this criterion particularly challenging. Data paucity and limited survey effort often result in *de facto* limited information about the occurrences of any species, but especially rare species. The only mitigation of data paucity is increased search effort, but depending on the nature of the species in question, sufficient searching may not be economically viable for truly rare species. This conundrum can lead to a number of challenges in designating conservation areas for rare species. Areas may be insufficient to maintain the species in question, habitat conserved may not reflect the true ecological needs of the species, or conservation measures may be overly broad and unwarranted. Further, conservation areas based on historic distributions may not be appropriate, as landscapes or habitat may have changed dramatically and restoration may not be technically feasible.

The global state of a species may also offset the use of rarity as an indicator of conservation importance. Locally abundant species may in fact be in decline globally, and should therefore have priority, or a focus on rare species may ignore species that are globally concentrated within a political jurisdiction, as suggested by Bunnell, Campbell, and Squires (2005). They suggest that the need for conservation efforts should be assigned to species based three primary criteria: 1) significant world populations or local ranges, 2) population trends, and 3) species vulnerabilities and threats. Similarly, NatureServe, its member Conservation Data Centres, and their collaborators, assess extinction risk of species and ecosystems using factors organized into three broad categories: rarity, trends, and threats (Master et al. 2009).

Because many species are in fact rare, "rarity" in a conservation sense may be better defined based on a species' population relative to the minimum number of individuals that are required for a population to persist over a suitably long time (i.e., minimum viable population). This definition may better relate population size to risk of extinction and therefore conservation efforts.

However, estimating the likelihood of population persistence (or extinction probability) over long-term planning horizons can be technically challenging. Fieberg and Ellner (2000) showed that precise estimates of extinction probability over a time horizon of t years requires between $5t$ and $10t$ years of data. In contrast, data sets for many rare or threatened species are likely to be highly limited in terms of data quantity or be of relatively poor quality. Population Viability Analysis (PVA) models tend to be very sensitive to their assumptions and parameter estimates, giving them very wide confidence intervals (Ellner et al. 2002), which can be estimated only by repeatedly re-sampling real data (Sokal and Rohlf 1981). Such broad confidence intervals reduce their usefulness in making conservation decisions. Nonetheless, solutions to these limitations have been proposed, such as producing a prediction interval over the entire time horizon of interest (rather than an arbitrarily selected horizon; Sæther et al. 2000), or calculating extinction probabilities over a range of parameter estimates using Bayesian methods (Ludwig 1996). Beissinger and Westphal (1998) suggest using PVA to evaluate relative extinction rate, using simple models, and that projections be made over relatively short time scales. Others argue that PVA models should continue to be used as just one factor in the decision-making context, and results of any PVA should be interpreted with caution (Brook et al. 2002).

Jurisdictionally rare species may be better managed through the incorporation of data on global abundance and trends, as well as stewardship responsibility based on the percent of species range within the jurisdiction, to ensure they are prioritized correctly to maximize the net benefit of conservation efforts.

3.4 Presence of Special Species or Taxa

Presence of special species or taxa may be used as a criterion for conservation for a variety of reasons. While some conservation organizations are or were originally species- or taxon-based (e.g., Ducks Unlimited), others may target specific species attributes (e.g., Wildlife Conservation Society's wide-ranging species). Alternatively, some species may be targeted for the simple reason that coarse-scale management is thought to be insufficient to protect them, and therefore a species-centric or fine-scale approach to management must be used. Further, special species may be targeted due to their assumed ecological functions, such as umbrella (conservation of one species equates to the conservation of many) or indicator (the loss of a particular species signifies degradation of the environment or habitat). Finally, "flagship" species may be targeted owing primarily to their appeal to the general public and ability to rally support for conservation concerns. The basis for this criterion is therefore both environmental/ecological and social in nature.

As previously mentioned (Section 3.2), the scientific basis for single-species approaches has at times has been found lacking. While the costs and difficulty associated with broad-scale biodiversity monitoring have been well documented, the option of surrogates for biodiversity such as single or groups of species offers the apparent panacea of effectively measuring or monitoring biodiversity through the monitoring of one or a few focal species, and providing broad-scale habitat management that protects all biodiversity values through the management of such umbrella species. However, the biodiversity value of single-species approaches has been questioned by a number of authors

(Andelman and Fagan 2000; Caro and O'Doherty 1999; Lambeck 1997, 2002; Lindenmayer et al. 2002) and it is unclear if such approaches accomplish the intended goal.

It is also appropriate to note that many species may be targeted as "special species" for purely moralistic or aesthetic reasons (Gordon, Franco, and Tyrrell 2005). For example, beyond the potential ecological function or links to biodiversity, birds may be the focus of conservation efforts because of the personal preference of the bird-watching community. Such socially driven species prioritization is an important component of conservation management, but one that is beyond the scope of this report.

3.5 Threatened Species

A threatened species is a plant or animal that is at risk of becoming extinct, and the presence of a threatened species is a common criterion for the designation of conservation efforts. The basis for this criterion is environmental/ecological in nature. Various organizations categorize threats to species using a variety of standards, but the most commonly cited international ranking is the International Union for Conservation of Nature's (IUCN) Red List (Figure 3.2; www.iucnredlist.org). The IUCN has seven levels of threat, from extinct to least concern. Categories also exist for poorly understood species (data deficient) and those species not examined by the IUCN (not evaluated).

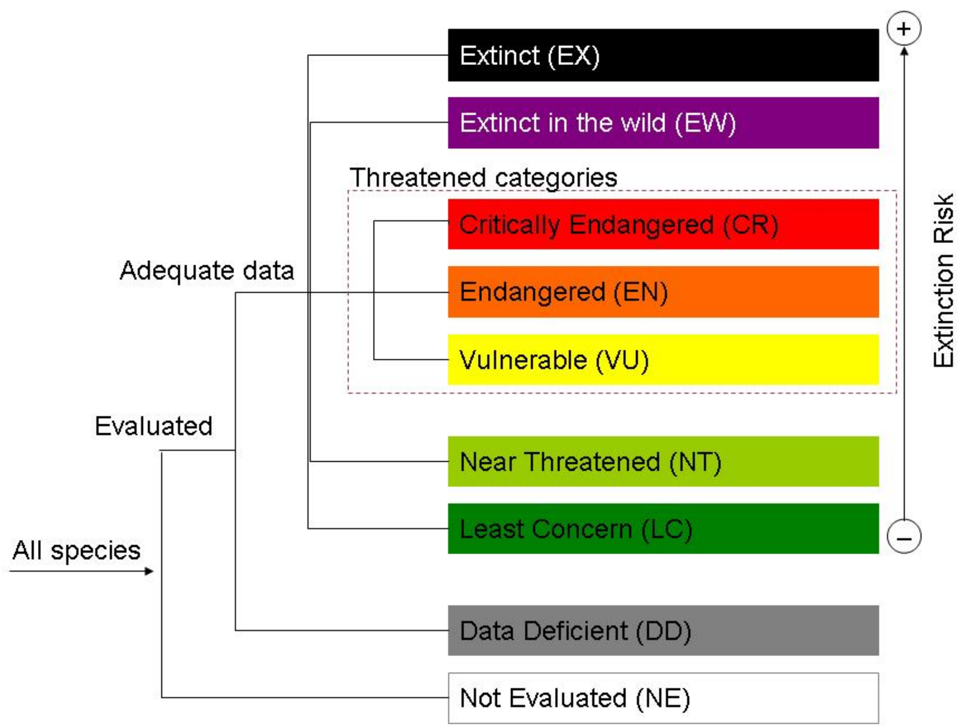

Figure 3.2 Structure of the IUCN's Category System (reproduced from Vié, Hilton-Taylor, and Stuart 2009)

Canada's Committee on the Status of Endangered Wildlife in Canada (COSEWIC) is responsible for assessing and ranking all terrestrial, aquatic and marine species in Canada. COSEWIC's assessment criteria are based on IUCN criteria, with some differences. The COSEWIC ranking system informs the decision process authorized by Canada's Species at Risk Act (SARA 2002), which incorporates COSEWIC's scientific assessment with input from various other government departments and organizations (Figure 3.3).

Various provincial jurisdictions may also have assessment programs (e.g., Committee on the Status of Species at Risk in Ontario, COSSARO) that may or may not agree with the national body. Despite having common grounding in IUCN guidelines, inconsistencies between provincial and national designations may result from a number of causes, including regional species population dynamics differing from the national level (e.g., Sleep, Drever, and Szuba 2009), "jurisdictional rarity" (Bunnell, Campbell and Squires 2004), internal inconsistencies in the application of listing criteria (Luckey and Crawford 2009; Mooers et al. 2007), differing assessment periods, and differing amounts or sources of assessment data, to name a few.

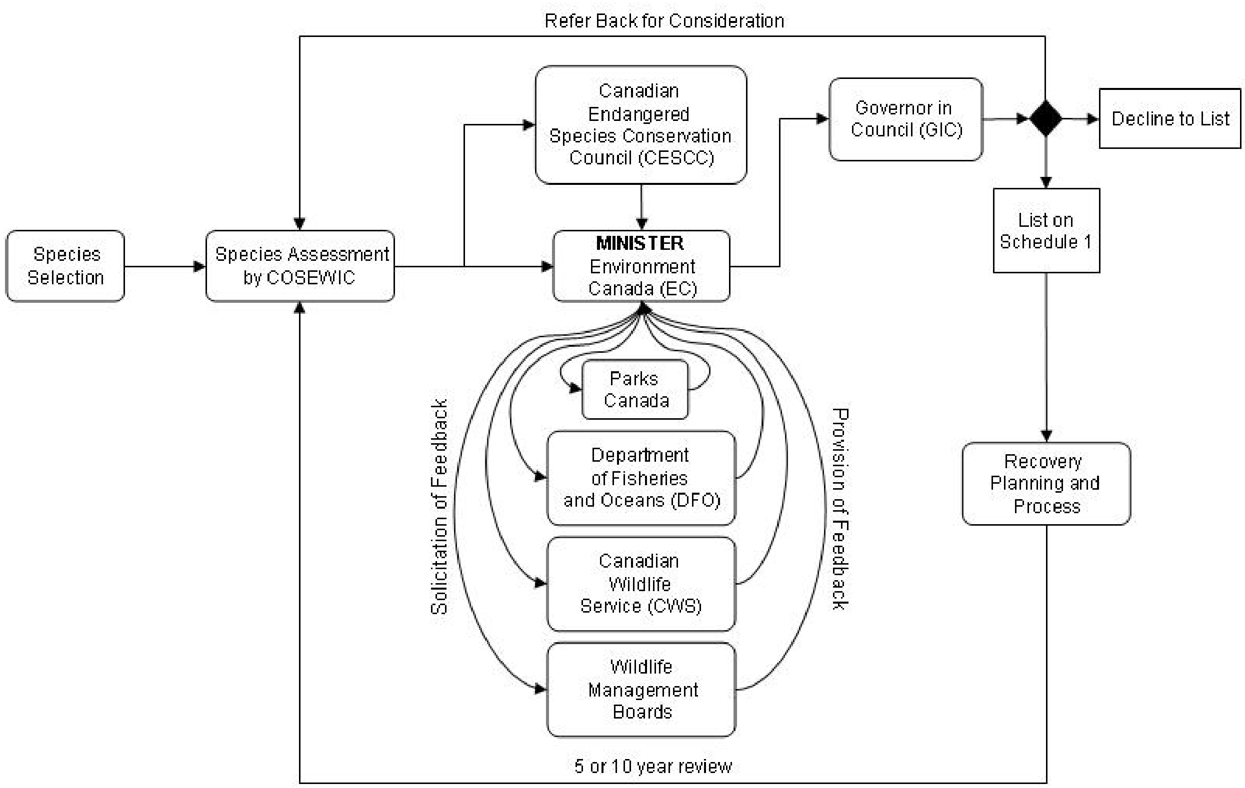

Figure 3.3 Schematic Diagram of the SARA Listing and Recovery Planning Process for Threatened and Endangered Species in Canada

National or provincial recognition of a species as threatened or endangered automatically invokes legal protection above and beyond standard protections from federal (e.g., the Canadian Migratory Bird Convention Act 1994) or provincial (e.g., the Ontario Fish and Wildlife Conservation Act 1997) statutes. Such legal protection, either under federal or provincial oversight, includes prohibitions against killing, harming, harassing, or disturbing individuals, their residences, or their critical habitats. Further, listing begins a legally required process of planning for species recovery through development of recovery strategies and subsequent action plans. Recovery efforts are directed by legislation to involve identification of critical habitat and increased monitoring of populations.

3.6 Species in Decline

Species decline is less frequently cited as a conservation criterion, likely because it is related to presence of threatened species which have sometimes undergone historic decline (Gordon, Franco, and Tyrrell 2005). The basis for this criterion is environmental/ecological in nature. The cause of a species' decline is often unrelated to its status as a criterion for conservation.

Given its potential overlap with some other criteria (e.g., species rarity, threatened species, endemic species), presence of species in decline alone may be sufficient as an indicator of areas of high conservation priority. For declining species that are not yet rare, designated as threatened, or endemic, determining the cause of the decline could contribute to the value of species decline as an indicator. Alternatively, commonness *per se* does not preclude a species from being considered a conservation concern if it is in decline. For example, the Passenger Pigeon (*Ectopistes migratorius*) was conservatively estimated to have a North American population between 3 to 5 billion individuals circa 1831 (Schorger 1955), but went officially extinct in 1914.

The definition of species decline is fundamentally straightforward. Anything that causes a population's mortality rate to exceed its reproductive replacement rate over a sustained period will result in population decline. However, determining which factor(s) is/are ultimately responsible for the proximate factor(s) that shift the equation from growth to decline are often exceedingly complicated. The most frequently cited cause of species decline is loss of habitat or living space. Wilcove et al. (1998) reported that "habitat degradation or loss" was cited as a cause of imperilment for 88.7% of species listed as threatened or endangered in the US. The authors reported that the causes of decline were based on expert opinion (e.g., perceptions of USFWS employees), and may or may not have been based on empirical evidence. Further, the authors noted that it was unclear how many species were actually declining, and how many were simply rare but stable.

In a similar analysis of Canadian threatened and endangered species, Venter et al. (2006) concluded that habitat loss and degradation was a factor in 93% of species listed, irrespective of its importance in causing the species decline. The authors did not include threats that were deemed theoretical, but were unable to determine if the threats were historical or current. Further, Venter et al. (2006) determined that agriculture (46%) and urbanization (44%) were the most prevalent causes of endangerment in Canada. Gibbs, Mackey, and Currie (2009) reported a statistical association between the proportion of an area in intensive agricultural use and the number of imperiled species in Canada. They argued that habitat conversion *per se* may be a less important factor than how the converted land is subsequently used.

Rosenzweig (1995) grouped the causes of species declines to extinction into two primary categories: "accidents" and "population interactions". Species may disappear accidentally due to stochastic events that result in either the death of entire populations (e.g., owing to catastrophic events such as volcanic eruptions, uncharacteristic fires) or the loss of a species niche (e.g., through the extinction of a predator's only prey species). Species interactions like predation or competition can also result in species extinction, at least in experimental settings [e.g., in manipulated tidal pools (Paine 1966), or mites in the lab (Huffaker 1958)]. However, before accidents or species interactions can result in global extinctions of a species, a species must be in some way rare (Section 3.3), and for a common species to become rare, it must first undergo population decline.

3.7 Representation

Representation is the inclusion or retention of all ecological elements of interest (e.g., ecotypes, ecosystems, habitat types, species, etc.) in landscape planning efforts (Olson and Dinerstein 1998; Gordon, Franco, and Tyrrell 2005). It is essentially a "coarse filter" approach (Armstrong et al. 2003) whereby users of this approach assume that maintaining a sufficient area in all ecosystems or habitat types will translate to maintenance of all species and to full biodiversity conservation (Sierra, Campos, and Chamberlin 2002). Olson and Dinerstein (1998) described representation as an integration of maintaining species diversity and preserving distinct ecosystems and ecological processes. Others have suggested that reserves will fulfill their role when they have been designed to be as representative as possible (Pressey et al. 1993). However, representation tends to require higher amounts of area conservation than other approaches such as species richness (Margules, Nicholls, and Pressey 1988) and representation may not be obtainable under arbitrary minimum percentage set-aside systems [e.g., 10 or 12%, McNeely and Miller (1984)] as suggested by Sierra, Campos, and Chamberlin (2002) and WCED (1987). Haufler, Mehl, and Roloff (1996) describe application of a coarse-filter approach on industry lands to identify adequate ecological representation.

The primary keys to conserving biodiversity through representation of biodiversity pattern (of taxa and land classes) are inventories and conservation bias avoidance. Systematic inventories of both existing conservation areas and of the overall taxa and land classes to be conserved are needed prior to or during management in order to determine the extent to which conservation targets have already been met, and what future efforts will be needed to achieve the remainder (Jennings 2000; Margules and Pressey 2000). Conservation bias refers to the placement of conservation efforts in an "*ad hoc* reservation" (Pressey 1994) system that does not meet conservation goals, but instead tends to be biased towards other values such as remote, rugged, or scenic terrain. Such conservation efforts are not incorrect *per se*, but do not necessarily contribute to the conservation of ecological or biodiversity values (Wiersma and Nudds 2009). The basis for this criterion can be therefore both environmental/ecological and societal in nature.

Species representation can pose a more significant challenge to forest management, particularly for species that tend to operate at larger spatial scales. Until the advent of broad-scale radio telemetry studies in the 1960s (e.g., Mech 1967), it was difficult understand how species used a given landscape. Species with specific habitat attributes (e.g., bald eagle nests, black bear dens) are readily managed through feature identification and the creation of "areas of concern" (AOC) around them (e.g., OMNR 1987). In contrast, species that use cryptic features (e.g., American marten, *Martes americana*) such as den snags (Potvin, Bélanger, and Lowell 2000), or species that are thought to be associated with general habitat features (e.g., woodland caribou, *Rangifer tarandus*) such as late-seral stage conifer for some portion of the year (Fortin et al. 2008), pose a greater challenge for forest management.

3.8 Significant or Outstanding Ecological or Evolutionary Processes

The presence of significant or outstanding ecological or evolutionary processes is a common criterion for conservation efforts. Defined broadly, this may include presence of migratory pathways, significant breeding sites, unique species assemblages, or evolutionary radiation (Gordon, Franco, and Tyrrell 2005). In general, such processes are defined and delineated at the local level, and are protected as the long-term source of either species abundance or richness. The basis for this criterion is primarily environmental/ecological in nature, with some element of societal value judgment used to determine what constitutes an "outstanding" process.

One aspect of forest management that fits this criterion is the maintenance of water quality and quantity. Water purification and supply is an essential ecosystem service provided by natural areas, particularly in forested landscapes. Nonetheless, growing stresses are currently placed upon this natural resource by industrial demands, urban and rural development, changes in land use practices, and in the future, by possible long-term alterations in precipitation patterns and hydrologic processes (Environment Canada 2004). As a result, protection of water quality or quantity as a significant ecological process may be an effective criterion for designating areas of conservation concern. However, the maintenance of ecosystem services as a reason for preserving biodiversity tends to overlook the fact that some ecosystem services may or may not be affected by species decline, and therefore identifying the role of some rare or endangered species in maintaining ecosystem services may be hard to determine (Ridder 2008).

Forests, forest management, and the forest industry play a unique role in the maintenance of water quantity and quality as well as hydrological processes (NCASI 2009a, 2009b, 2010). Forests act to process precipitation into high quality surface waters, and in North America, most surface waters are derived from forested areas. Managing forests for timber production can affect the partitioning of water between runoff, groundwater recharge, and evapotranspiration, and can affect the quality of streamflow (NCASI 2010). Timber harvesting can decrease evapotranspiration, resulting in increased streamflow and groundwater recharge. These increases decrease with the gradual regrowth of the forest, and effects are mitigated by managing harvest schedules to limit the areas of forest watershed harvested at any one time and by incorporating riparian management approaches.

Although managing forests for timber production can negatively affect water quality, contemporary best management practices (BMPs), as widely used in Canada and the US, reduce or eliminate impacts of harvest on surface waters when properly implemented (Williams et al. 2000). Natural disturbance events (e.g., wildfires, tropical storms, insect outbreaks) can have effects on water quality greatly exceeding those of forest management for timber production (McBroom et al. 2003; Ice and Schoenholtz 2003). Alternative land uses (e.g., agriculture, urban) typically have much greater negative effects on water quality with respect to loads of sediment, pesticides, nutrients, and other pollutants to streams than do properly managed forest operations. Compared to these other land uses, forests are also best for managing peak flows (Poor and McDonnell 2007). These observations, coupled with the knowledge that most fresh water originates from forested lands, suggest that maintenance of timberland, in lieu of conversion to non-forest land uses, is essential to the continued availability of high-quality fresh water supplies.

Forest management practices have also been used in many areas to maintain or mimic natural processes, such as natural disturbance events (e.g., fire and wind-throw). While forest management will never completely replace such processes (i.e., fire is a chemical process whereas harvesting is a mechanical process), practices can be tailored to leave natural patterns and residuals behind (e.g., green-tree patches, natural-contoured edges, significant amounts of snags and course woody debris). In time, wildlife tend to respond similarly between fire and harvested areas (NCASI 2006), and practices such as structural retention practices that mimic natural processes are thought to either maintain or “lifeboat” old-seral stage species until harvested areas recover enough to support them (NCASI 2009c).

3.9 Habitat Loss

Habitat loss is the permanent conversion of a desired habitat type to a less desirable type—one that is less suitable, or even wholly hostile for a target species or group of species (Gordon, Franco, and Tyrrell 2005). The basis for this criterion is environmental/ecological in nature. Habitat loss is most evident where forest is converted to another land use (e.g., agriculture, urban development), or where habitat becomes permanently hostile or unsuitable for a given species (e.g., flooding as a result of hydroelectric development). In contrast, anthropogenic activities (such as harvesting) and natural disturbances (e.g., wildfire and blowdown) are considered forest habitat change (rather than loss) as long as they do not result in conversion of forest to other land uses (e.g., urbanization, agriculture).

In Canada, the most recent data indicates that the amount of land area in forest cover is increasing (9,700 ha of afforestation in 2005; NRCAN 2009). Under this calculation, forests that are harvested are not included as habitat loss as they are offset by area successfully regenerated over the same time period (NRCAN 2009). Forest habitat types may change in extent if forested areas are changed from one type to another, owing to natural disturbance, successional pathways, or silvicultural practices. For example, under natural conditions, shade-intolerant species such as trembling aspen (*Populus tremuloides*) tend to recolonize when mature conifer forest cover is removed, and shade-tolerant conifer species do not successfully establish until a full canopy is re-established. Depending on the species in question, changes from one forest type to another could be deemed habitat loss or gain even though the area of forest has not changed.

As discussed in Section 3.6, habitat loss is considered a significant cause of species declines. Research to date in Canada suggests that the primary ultimate causes of habitat alteration in Canada stem from agricultural and urbanization activities, accounting for 46% and 44% of presumed threats to species, respectively (Venter et al. 2006). Habitat loss is sometimes confounded with other indicators of conservation concern such as fragmentation.

3.10 Fragmentation

While there is no universally accepted definition of fragmentation (NCASI 2008), Gordon, Franco, and Tyrrell (2005) define it as “the disruption of extensive habitats into isolated and small patches.” The basis for this criterion is environmental/ecological in nature. The scientific underpinnings of fragmentation come from Island Biogeographic Theory (IBT), first proposed by MacArthur and Wilson (1963, 1967). IBT states that the number of species found on an island is primarily a function of its size and distance from shore, all else being equal. IBT is well supported for true islands as well as permanently isolated terrestrial habitats (e.g., caves and mountain tops; Watson 2002). However, support for its transference to fragmented landscapes in which patches of target habitat (e.g., mature forest) are dispersed in a matrix of different, and perhaps less suitable habitat (e.g., younger forest), is less clear (Bunnell 1999b).

Fragmentation has two components: habitat loss and habitat isolation. The concepts of habitat loss, isolation, and fragmentation are often confused (Zaviezo et al. 2006). Should fragmentation be the result of some permanent land use change (e.g., roads or agriculture), fragmentation effects could be largely due to permanent habitat loss as forest area is lost. There is also the potential for the new land use to have detrimental effects on the areas of remaining habitat or to be inhospitable for species as they move across it (Gibbs, Mackey, and Currie 2009). In contrast, forest management usually alters habitat temporarily rather than permanently, and the intermittent matrix varies in suitability for forest-dwellings species. Nonetheless, the confusion surrounding the term “fragmentation” has led a number of authors to question its value as an ecological term and to argue for clarity in future studies (Bunnell 1999a; Haila 2002; Fahrig 2003; Lindenmayer and Fischer 2006).

Some of the complications around fragmentation stem from its inherent species-specific effects. For example, a permanent road built across a forested landscape would on the surface appear to constitute fragmentation. It absolutely constitutes forest habitat lost, as the road bed will not be forested again as long as it is maintained. However, fragmentation effects of the road will vary by species. Plants or animals with limited dispersal ability may be unable to colonize the far side of the road. Songbirds (depending on the species) may or may not establish territories that encompass the road. Some larger mammals will have no problems crossing the road (although collisions may be a problem), whereas other large mammals (e.g., caribou) may in fact avoid the road (Dyer et al. 2002). In the case of a transportation corridor such as a road, the level of habitat fragmentation for some species may be in part dictated by the level of traffic using the road.

Evidence is ambiguous as to the relationship between the effect of habitat fragmentation and a species' traits (e.g., ecology, behaviour). Some authors have suggested that certain species traits such as mobility, population density, and habitat specificity likely make them more susceptible to negative effects of fragmentation (MacNally and Bennett 1997). In a subsequent study however, they found that species mobility and habitat specificity were not predictors of species susceptibility to fragmentation, whereas population density was, but it was a negative predictor, contrary to theory (MacNally, Bennett, and Horrocks 2000). They suggested that the failure of these traits to predict susceptibility to fragmentation may have been related to secondary effects of fragmentation (e.g., changes to habitat quality and non-target species populations) that also affected target species. Similarly, Blanchet et al. (2010) reported species-specific effects of fragmentation on fish species in forested watershed, but found that susceptibility was related to fish size, which related to mobility and effective population size.

3.11 Large Intact Areas

Intactness is a term that is closely related to fragmentation and habitat loss, but is more holistic in the sense that it captures all disturbance that is anthropogenic in nature. Although specific criteria for defining an area as intact may vary among conservation organizations (Table 3.3), they are based on a suitably large area (e.g., >50,000 ha that is at least 10 km wide, Lee et al. 2003) being almost entirely unaffected by human disturbance, as determined through the use of satellite imagery, ancillary data, and expert consultation. The basis for this criterion is primarily societal in nature.

Table 3.3 Comparison of Different Assessments of the World's Intact Forest Area (within the forest zone) (Potapov et al. 2008b)

Type of Intact Area (definition)	Minimum Size km^2	Main Data Source	Percent Deemed Intact
Wilderness areas (McCloskey and Spalding 1989)	4000	Jet navigation charts	21.1
Frontier Forests (Bryant et al. 1997)	Inconsistent, but >500	Expert information	21.8
Areas with minimal (0-10) human influence derived from Human Footprint data set (Sanderson et al. 2002)	500	Coarse-scale global maps and low spatial resolution remote sensing data analysis results	38.0
Intact forest landscapes (Potapov et al. 2008a)	500	High spatial resolution satellite images	23.5

By definition, "intact" refers to something that is untouched, especially by anything that harms or diminishes (Merriam-Webster OnLine 2010). However, "intact forests" have been defined as "…forests unaffected by habitat fragmentation and other large-scale human activities over time periods long enough to allow natural forest structure, composition and functions to be determined primarily by naturally occurring ecological processes (e.g., natural regeneration, natural levels of disturbance, including fire and flooding, scale, frequency and intensity of natural disturbance regimes, and gap dynamics)" (Wye Report 2002, 8). The Canadian boreal forest, for example, has many areas that are deemed "intact" if one defines intact as "…an unbroken expanse of natural ecosystems within the zone of current forest extent, showing no signs of significant human activity" (Potapov et al. 2008b).

In the scientific literature, however, there are other definitions for the terms "intact" or "intactness", when used in reference to landscapes or forests. For example, some authors have used the term "intact" to refer to forests that are not subjected to treatment, such as full or partial canopy removal (Hughes and Fahey 1991) and numerous authors use "intact" to refer to forests that do not have canopy gaps created by natural disturbances (Wilczynski and Pickett 1993; Boman and Casper 1995; Schnitzer and Carson 2001). In contrast, others defined "intact forests" as "…forests with the full complement of natural, physical, and biotic interactions in place" (Naidu and Delucia 1999, 610) irrespective of human involvement. Trejo and Dirzo (2000, 135) used a regionally specific definition, suggesting that "intact forests are characterized by the abundance of trees 8±12 m in height (ca 100 trees per 0.1 ha), abundant lianas and a continuous layer of native shrubs". None of these definitions agree with each other, and each one is measurable only within the context of the individual or closely allied study. Finally, some authors use the term "intact", but do not define it (e.g., Hamilton et al. 2004). When analyses are performed to quantify levels of intactness, the definition used, the data source, and the treatment of the data often have a significant impact on the amount and location of areas defined as "intact". Thus, while the criterion of "intact forest" is based in part on ecological/environmental factors (e.g., due to fragmentation), there is a strong social element to the definition.

Because forests are characterized as "intact" only if they are free of human influence, any human-caused disturbance will have an immediate impact on the extent of areas designated as intact. This is because of the sometimes unstated assumption that in all cases of human use (e.g., temporary disturbance, roads, etc.) the environment is harmed or diminished (e.g., fragmented). However, depending on the type of disturbance and species being considered, such disturbances may or may not increase fragmentation on a landscape. The boreal forest is a naturally heterogeneous area, with frequent natural disturbances that routinely break up previously contiguous habitat. Similarly, forest management typically results in habitat heterogeneity rather than habitat loss and fragmentation, and the effects of forest management are generally indistinguishable from natural disturbance after 30 years have passed (NCASI 2006).

3.12 High and Low Future Threat

The assessment of future threat is a criteria used by a number of organizations to identify areas of conservation priority (Gordon, Franco, and Tyrrell 2005). Salafsky et al. (2008, 898) defined "threats" as "proximate (human) activities or processes that have caused, are causing, or may cause the destruction, degradation, and/or impairment of biodiversity and natural processes". However, the link between perceived future threat and biodiversity is not always clear. Common ways to assess future threat are the level of human population density in a given area, the amount of area protected by IUCN protected area Categories I-IV and the numbers of endemic and threatened species present. Each of these indicators has various assumptions that may or may not be based on empirical evidence.

For example, a number of studies have found a strong correlation between human population density and the number of threatened or endangered species present in an area (McKee et al. 2003; Kirkland and Ostfeld 1999; Thompson and Jones 1999). "Biodiversity hotspots" tend to not only have high species richness, but are also associated with high human populations that are perceived to reduce biodiversity, therefore making them a target for conservation efforts. This results in a clear relationship between human population and high biodiversity (Cincotta, Wisnewski, and Engelman 2000), but confounds the two issues within the "hotspot" definition (Myers et al. 2000). However, current human population growth rates do not correlate with existing biodiversity threats (McKee et al. 2003).

Protected areas are thought to be one of the primary means to maintain biodiversity. Stricter IUCN designations (I-IV; World Conservation Union 1994) are thought to provide the highest levels of biodiversity conservation, and therefore a protected status is often assumed to result in low future threat to an area. Some parks have been found to adequately protect biodiversity in the medium to long term (e.g., tropical biodiversity, Bruner et al. 2001). However, where conservation efforts are inadequate, protected area designation will do little to maintain local biodiversity (e.g., Liu et al. 2001).

At the global scale, a gap analysis by Rodrigues et al. (2004) suggested that the global protected areas system in 2003 (11.5% of the planet's land surface) represented approximately 88% of approximately 12,633 terrestrial vertebrate species examined. However, presence alone of a species within the boundaries of a protected area is insufficient to ensure protection (Newark 1996), and some authors have suggested that protected areas be designated based on the probability of persistence of species and/or habitats of concern, and the potential vulnerabilities of the same, rather than on presence of species of concern (Gaston, Pressey, and Margules 2002).

Endemic species (see Section 3.2) are often not well represented within protected area networks designed to include areas of high species richness (Rodrigues et al. 2004). As a result, Rodrigues et al. suggested that the extent of protected areas in a region may not be a useful indicator of the sufficiency of protection. They further suggest that uniform percentages of area protected cannot be used to distinguish between regions sufficiently protected and those that require additional protection.

In spite of all the quantifiable estimates of protected area effectiveness, biodiversity and endemism hotspots, and human population density, the criterion of future threat remains primarily a societal value. Human population densities and trends in many locations, most notably in remote locations, can change unrelated to the biodiversity of the local area. Further, changes in the world markets for commodities can move an area from a low to a high future threat or vice versa simply because it becomes more or less economically profitable to use a resource. Finally, biodiversity levels and numbers of endemic species can decline due to natural events [e.g., wildfire destroyed the last remaining Heath Hen, *Tympanuchus cupido cupido* habitat in 1928 (Gross 1931)] changing the long-term biodiversity value of and future threat to a given area.

3.13 Inter-Relatedness and Overlap of Indicators

It is important to note that there is significant overlap amongst the criteria discussed in this report, such that information from one criterion is sometimes incorporated into another. For example, areas of high species richness tend to have the largest numbers of both rare and endemic species, leading researchers to speculate that these areas may be sites of significant ecological or evolutionary processes (Jetz, Rahbek, and Colwell 2004). Also, by virtue of higher species richness, higher numbers of specialist species are likely to be present. Species that are both rare and isolated (such as endemics) are those that are more likely to suffer declines, particularly as habitats undergo fragmentation (Davies, Margules, and Lawrence 2000). Habitat loss and fragmentation are concepts that are impossible to separate (NCASI 2008), and are the *de facto* inverse of large intact areas, all three of which therefore (depending on their state or trajectory) contribute to a high future threat or low future threat for a species (Hoekstra et al. 2005). Rare, declining, and endemic species are often listed as threatened or designated as special in some way or another. It is therefore reasonable to suggest that the only independent criterion of the group is habitat representation. Because conservation planning criteria are inter-related, when planning and executing forest management and when meeting various guidelines and certifications, the forest industry does not and cannot consider and manage for these indicators individually. Rather, management must be designed to meet them simultaneously.

There are a number of ways that these criteria can be categorized. The inter-relatedness among criteria facilitates grouping conservation criteria into categories (Table 3.4) that describe various aspects of species present in an area (Species Presence or Trend), condition of the landscape (Landscape State or Trend), and perceived future of an area (Future Threat). Further, it is worth noting that landscape state/trend criteria tend to be coarse-filter (focusing on broad scale attributes and functions like landscape composition and water filtration) whereas species presence criteria tend to be fine-filter (focusing on species that may or may not be captured by coarse-filter approaches; Noss 1987; Hunter 1991).

Table 3.4 Criteria Examined by Gordon, Franco, and Tyrrell (2005) Grouped According to Interrelatedness

Species Presence/Trend (fine filter)	Landscape State/Trend (coarse filter)	Future Threat
Species Richness	Representation	High Future Threat
Species Endemism	Significant or Outstanding Ecological or Evolutionary Processes	Low Future Threat
Rarity	Habitat Loss	
Special Species or Taxa	Fragmentation	
Threatened Species	Large Intact Areas	
Species Decline		

4.0 FOREST MANAGEMENT'S CONTRIBUTION TO ACHIEVING CONSERVATION CRITERIA

4.1 Species Presence Criteria (Fine Filter)

Conservation of biodiversity values can only be considered at the whole-landscape scale. Collectively, about 90% of the earth's terrestrial land base is found within the "semi-natural" matrix where landscapes are managed for a combination of commodity production and ecological conservation (Chape et al. 1993). It is in this context that forest management, particularly when compared to many other resource-based industries, can be used to promote and maintain biological diversity at the landscape scale. In addition to the various planning and assessment efforts undertaken during forest management, many operators have developed and tested a range of management practices to either maintain or promote biodiversity on managed forests (e.g., the maintenance of forest structural features; see NCASI 2009c). Many of these practices are specifically tailored to promote species presence, either through the provision of ecological niche-space for certain groups of species (e.g., interior forest patches, standings dead wood), or through the maintenance of core habitat for specific species of conservation concern (e.g., Northern Spotted Owls, *Strix occidentalis caurina*).

Forest management planning in Canada generally requires an assessment of species presence and richness on a landscape prior to harvesting, and both provincial guidelines and forest certification standards require the provision of sufficient habitat for all native species (Principle 4, SFI 2010; Principal 6, FSC 1996; CCFM Criterion 1, CSA 2003). Further, many companies conduct effectiveness monitoring to ensure that practices are meeting various biodiversity objectives, and where they are failing, adjustments can be made in an adaptive management framework (Houde, Bunnell, and Leech 2005). Species richness can be managed (increased or decreased) using a variety of silvicultural techniques. For example, recent meta-analyses documented a short-term positive response in richness by many taxonomic groups to thinning (Verschuyl et al. 2011) and a negative response to removal of snags (Riffell et al. 2011).

In Canada, endemic species are maintained on managed landscapes through the maintenance of habitat types and conditions associated with occurrences of rare communities. Many endemics are associated with inoperable sites that already receive special consideration. Thus, endemic species can often be accommodated without significant impacts on more intensively managed parts of the landscape. Should individual species begin to decline beyond that which would be expected naturally, management efforts can be tailored to maintain or promote such species. As a result of these requirements, the forest industry has contributed substantial resources to designing, implementing, and evaluating forest management guidelines for the preservation of species of concern (e.g., American marten, woodland caribou, *Rangifer tarandus*). However, few truly endemic species are found on managed landscapes in Canada (see Section 3.2).

Within managed forests, rare, threatened, and declining species may be accounted for and their populations managed in a variety of ways. If rare species are considered at risk by national or provincial standards, forest management adheres to species-specific forest management guidelines designed to conserve or recover them. Guidelines may provide direction on designating areas of concern around specific geographic or ecological features (e.g., nests for Bald Eagles; OMNR 1987), or may direct managers to plan long-term landscape-level activities that provide conditions suitable for large-scale species (e.g., woodland caribou; Racey et al. 1999). Managing for other criteria, such as representation, may also contribute to maintenance of rare species by maintaining all habitat types on a landscape. Because forest managers often consult with local communities and aboriginal groups, special and significant species to those communities are often directly incorporated into management plans.

Other landscape-level metrics may also be considered during industrial forest management, such as connectivity, allowing for the dispersal of rare species between preferred habitats. Finally, forest certification may require the maintenance of specific habitats that are known to be associated with rare species, such as High Conservation Value Forests (HCVFs: FSC 1996) and Forests with Exceptional Conservation Value (FECVs: SFI 2010).

4.2 Landscape State Criteria (Coarse Filter)

Area representation in managed landscapes is generally easily attained, provided that pre-management conditions are known or obtainable. With the exception of areas where forest plantations (i.e., intensive forestry systems, as defined in the “Triad” approach; Seymour and Hunter 1992) are used, most forest management plans in Canada are designed to maintain the proportion and total surface area of stand types found on the original pre-harvest landscape, unless restoration of some historic condition is sought. Further, management plans often seek to maintain the proportion and total area of age class structure as would be found on the same landscape given a regime of natural disturbance, including disturbance by fire, blowdown, and insect outbreak (e.g., Perera and Buse 2004). Efforts to maintain representation may also include replanting of trees grown from cones gathered from the same area either pre- or post-harvest (i.e., genetic representation). Such efforts not only help maintain representative species composition of a given area, but maintain genetic stock that may not propagate in the absence of fire (e.g., in the case of pyrophilic species such as Jack pine, *Pinus banksiana*).

Significant or outstanding ecological or evolutionary processes are often dealt with on a case by case basis, using such tools as the concept of HCVFs (FSC 1996) or FECVs (SFI 2010). The HCVF and FECV concepts shares common characteristics with the "significant or outstanding ecological or evolutionary processes" criterion (and aspects of several others) described in this report. Principle 9 of FSC states that "management activities in high conservation value forests shall maintain or enhance the attributes which define such forests. Decisions regarding high conservation value forests shall always be considered in the context of a precautionary approach". Similarly, the SFI Standard's Objective 4 extends its biodiversity requirements to FECVs and requires that Program Participants have "plans to locate and protect known sites associated with viable occurrences of *critically imperiled* and *imperiled* species and communities", whereby *critically imperiled* and *imperiled* species are defined by NatureServe. As such, the HCVF and FECV concepts are designed to help conserve a number of the conservation values identified by the criterion (e.g., special species or taxa, outstanding ecological processes) discussed in this report.

However, identifying and delineating significant areas or HCVFs has been a challenge in part because there is a social component to the concept, and some organizations have noted its subjective nature (WWF International 2007). Local or traditional ecological knowledge may help identify HCVFs. HCVFs are also identified through ecological studies that identify ecologically sensitive features (e.g., calving grounds) or territories of threatened or endangered species. Forest management companies that are certified to FSC standards are required to delineate and avoid HCVFs wherever and whenever possible. The delineation of FECVs relies on NatureServe data and associated global conservation status rankings ("G ranks") to identify areas that have imperiled or critically imperiled species or communities present (SFI 2010).

A significant landscape-scale issue for forest management is the potential for habitat loss from the development of permanent and all-weather road networks, particularly in areas being harvested for the first time (it should be noted that this is more often the case in Canada than in the US where previously developed networks are usually in place). Mapping algorithms and GIS frameworks can be used to minimize the effects of road networks on areas under forest management (MacDonald 2000). If multiple industries are using a forest management area, (e.g., mining, oil and gas exploration/developing) cooperation between industries can help minimize road building, and therefore reduce cumulative impacts of permanent habitat loss.

Once harvested areas have been replanted or managed to ensure natural regeneration, monitoring is conducted to ensure that stands reach "free to grow" stages within appropriate time frames and are therefore assumed to be returning to pre-harvest conditions (e.g., OMNR 1996, 2001b). As such, harvesting may be considered as temporary habitat alteration, rather than loss, assuming measures are put in place to minimize forest type conversion (such as conifer to deciduous) which can cause habitat loss for some species and habitat gains for others as microhabitats between stand types can differ (Liechty et al. 1992).

Current forest management strategies in Canada are primarily designed around maintaining landscape-level patterns that existed prior to harvesting, an approach referred to as managing within the historic range of natural variation, or emulating natural forest disturbance (ENFD; Crow and Perera 2004). Thus, the size of both harvested and remaining forested landscape blocks, barring particular circumstances, is planned to reflect the pre-existing diversity of sizes and configurations, with consideration given to economic and social concerns. For example, extremely large cataclysmic wildfire sizes may not be represented among clear-cut sizes and water bodies are not generally harvested to their edges to maintain water quality. Emulating natural forest disturbance (e.g., OMNR 2001a) can conflict with societal and economic considerations, both of which are important elements of sustainability and both of which may have value to the sustainability of a landscape (Schmiegelow *et al.* 2006).

4.3 Future Threat Criteria

Criteria related to future threats to species or landscapes generally indicate societal concern about conservation and often reflect the priorities of the identifying entity. For example, any anthropogenic (human-caused) disturbance would be considered by some to constitute a significant threat to natural ecosystem values, whether or not such disturbances caused changes to environmental/ecological indicators. As a result, any area with significant timber value or sub-surface mineral or hydrocarbon deposits would be considered to be in an area of high future threat, as at some future time those resources may be managed or used. Natural areas that are far removed from population centers and thought to be beyond the reach of humans would be considered areas of low future threat.

Alternatively, some may have the perspective that forest management planning minimizes future threat to a landscape by providing for its long-term sustainable use while respecting biodiversity and other values already present. Conservation practices such as green-tree, coarse woody debris (CWD) and snag retention, road and skid-trail planning, riparian area avoidance, and natural disturbance emulation are all designed to maintain landscapes within a range of natural variability, and to protect functions and values such as water quality and quantity, and soil properties. Monitoring is an important part of long-term forest management and, when used in an active adaptive management framework, its results help managers respond to potential future threats. As a result of these efforts, forestry may be a net promoter of biodiversity values, particularly when contrasted with alternative land-use practices which have significant negative effects on biodiversity (Gibbs, Mackey, and Currie 2009).

4.4 Synergisms, Interactions, and Trade-Offs

It is useful to consider how the above criteria are related to one another, especially when considering conservation management choices that can be made and the efficiencies and effectiveness that can result from understanding these relationships. For example, species presence-related criteria can be inter-related, i.e., changing the amount of species richness on a landscape will tend to alter the number of rare, endemic, declining, or special species that are present. Furthermore, changes in forest management made to enhance habitat quality for one rare or declining species may diminish habitat for another. Interestingly, species richness is the only species presence-related criteria that can be estimated as a continuous variable, whereas “rare”, “endemic”, “declining”, and “special” species refer to descriptors of the state of species that may be present.

Similarly, landscape-state variables are related to one another and only two, fragmentation and representation, are measurable (although the metrics themselves may be problematic in terms of their relationship with achieving biodiversity conservation; see NCASI 2008). Alternatively, “presence of significant or outstanding ecological processes” and “habitat loss” refer to features or processes present on the landscape, and whether the feature or process is “present” on the landscape will depend on the level of representation being maintained and the effect of fragmentation on the habitat of the species or species community in question. It is also significant to note that all but two of the landscape state-related criteria (representation and habitat loss) incorporate societal values, in addition to environmental/ecological science.

Finally, future threat, as a criterion, is less aligned with biological or ecological issues than with societal norms and perceptions. Efforts to predict future threats to biodiversity have been undertaken in recent years to try and provide sound science to inform policy before potential threats become realities (Sutherland et al. 2008, 2009). However, identified future threats to biodiversity are often allied with new technologies (e.g., nanosilver and artificial life), geographic patterns of human and/or technological travel (e.g., biotechnology and invasive species), or future climate scenarios and human adaptation (e.g., increased fire risk and geo-engineering). Nonetheless, landscape-scale land acquisitions for food production and protection of forests for carbon sequestration have been identified as threats, as they may lead to large-scale conversions to agricultural use, which may have negative impacts on biodiversity (Sutherland et al. 2009).

That said, depending on the conservation aims of the organizations that use future threat levels as indicators, both high and low future threats can indicate high conservation value (Gordon, Franco, and Tyrrell 2005). Future threat criteria are perhaps the most nebulous, as perceived threat to a landscape can change in both intensity and direction with changing societal goals and desires, or disappear altogether depending on changes to local and global economics (e.g., drop in demand for oil can greatly reduce oil and gas development pressure). Because of changes in human development, technology and needs, exercises like those above can produce new threats even one year after a prior exercise. For example, Sutherland et al. (2011) identified the expansion of mining for lithium as a global conservation issue, owing to its substantial use in rechargeable batteries. Lithium mining was not mentioned in a similar exercise only one year earlier (Sutherland et al. 2010). During both exercises, in which academics and researchers were assembled to develop a list of 15 current and nascent threats to global biodiversity, there was no overlap between years.

Future threats may also be related to biological/ecological issues such as insect infestations or catastrophic disturbance events (e.g., wildfire or blowdown), which may or may not ultimately be related to human activities. For example, extreme wildfires may be the result of extreme fuel loadings (e.g., that could result from a policy of no harvest and fire suppression) combined with regionally specific and rare drought conditions. As such, future threat may be related to both long-term management and/or rare (e.g., once in 500 years) events. The inherent difficulty in incorporating the uncertainty around the timing, frequency, location, extent, severity and impact of such events limits their applicability to conservation planning.

Ideally, sound environmental policy is based on sound science, as well as relevant social and economic considerations (Figure 4.1). For example, the designation of well designed and scientifically reasoned protected areas without consulting local communities or local aboriginal peoples could result in failure of those areas to meet their conservation objectives and lack support from local people. Generally speaking, conservation decisions that are made without consideration for social consequences to people who interact with local ecosystems can create significant problems. An apt example is the cancellation of the spring black bear hunt in Ontario that created serious mistrust and animosity from the northern Ontario populace and the Ontario Ministry of Natural Resources (Lemelin 2008).

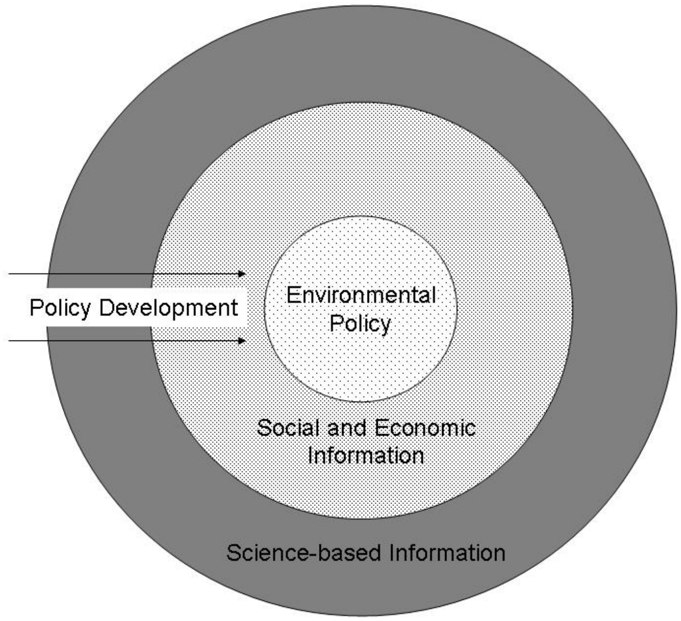

Figure 4.1 Schematic of the Progression of Environmental Policy Development

4.5 Applying Conservation Criteria to Forest Management

Forest management is both the art and science of implementing a system that allows forests to continue the sustainable provision of ecosystem supplies and services. The challenge of effective forest management is to do so based on a strong scientific platform and in a socially acceptable manner that can be economically maintained. Forest management incorporates a suite of criteria and indicators designed to help achieve this balance, many of which are included in the criteria described in this report.

To apply a conservation criterion to forest management, the first question to be asked is to what degree is a criterion measurable? Many of the criteria described here are easily measurable, and are therefore readily incorporated into forest management. While "biodiversity" as a concept is both difficult to define and probably impossible to actually measure (Bunnell and Dunsworth 2009), many of its components are readily measured and monitored, and are currently incorporated into forest management plans (e.g., species occurrence/trend criteria). In contrast, while some landscape-state criteria have had empirical measures developed for them (e.g., fragmentation), others are defined on a case-by-case basis (or study-by-study) and as a result may be less useful in an on-the-ground forest management context (e.g., intactness). When measures and indicators can be revised arbitrarily or are influenced by factors well outside the forest resource (e.g., socially derived priorities), they are less practical as tools for measuring progress towards a conservation goal, given that they can artificially increase long-term uncertainty (e.g., future threat).

Many, if not all, of the readily definable and quantifiable measures described in this report are already incorporated into sustainable forest management planning. As described and depending on the jurisdiction, forest managers on Crown land in Canada are required to consider species presence, landscape representation, special or significant ecological values, and many others in forest management plans, either by legislative imperative, sustainable forest management certification programs, or best management practices.

The incorporation of socially derived criteria, the basis for which is less empirical and can therefore change in a less predictable fashion, could introduce a significant element of uncertainty into forest management planning. While uncertainty is a common element of working within natural resource management, this uncertainty would be less related to scientific uncertainty and more related to changing societal influences and economic indicators beyond the forest boundary.

5.0 INDUSTRY RESEARCH AND INFORMATION NEEDS

Research contributes to further our understanding of the relationship between forest management and biodiversity in a number of ways. Various provincial, state or federal [e.g., National Science and Engineering Research Council (NSERC), National Science Foundation (NSF)] organizations provide funding and conduct forestry-based research, and projects are often supported by direct financial contributions from the forest products industry. Further, forest management companies may contribute in-kind (equipment) or logistical (maps and silvicultural treatments) support for ecological research on their forest tenures, and provide direct financial contributions to externally executed research by various organizations (e.g., Ducks Unlimited, World Wildlife Fund, NCASI). Finally, some companies employ research scientists who contribute research to the scientific literature.

Results of these efforts have yielded various management strategies that have contributed significantly to the conservation of biological diversity within managed landscapes (e.g., Racey et al. 1996). The success of some of these strategies has led some to speculate that forest management is compatible with some of the more challenging species for which strategies have not been developed, given sufficient research and adaptive management approaches (Racey et al. 1991), and that forest management, through guidelines and harvesting practices, has an important role to play in maintaining and in some cases promoting biodiversity (Witiw 2006; Balisky 2007). Nonetheless, there remain a number of information gaps that still need to be addressed.

At the landscape level, there is as yet no universally applied metric for measuring fragmentation. Davidson (1998) made a number of recommendations to standardize fragmentation quantification. While fragmentation remains a significant issue for forest management, particularly as it relates to the social demand to maintain large intact areas, fragmentation is less of an overall issue for management than maintaining habitat quantity and quality. Boutin and Hebert (2002) suggested that the most important landscape feature for managers to focus on should be maintaining habitat types on the landscape, and only when habitat loss is significant should managers focus on configuration (i.e., fragmentation). Further work should continue to help identify what those thresholds might be.

Much of our understanding and most management practices in Canada are related to the range of natural variability on a landscape, and with the exception of plantations, keeping the effects of management within those bounds. Nowhere is this more applied than in the boreal forest, a system heavily influenced by natural disturbance (Rowe 1972). Such management regimes are designed to emulate natural disturbance pattern, intensity and extent, with the intention of maintaining native biodiversity levels. While numerous jurisdictions have applied such an approach (e.g., Ontario; OMNR 2001a) it has been generally applied in a number of regions through the implementation of "Ecosystem-based Management," a holistic approach to forest management that encompasses natural disturbance, native biodiversity, ecological functions, socio-economic concerns, recreation, and aesthetic concerns (Gauthier et al. 2009).

However, the working hypothesis of natural disturbance emulation has only been applied in the last 10-15 years, and still has many facets that require testing (reviewed in NCASI 2006), such as the interaction of secondary processes (e.g., ericaceous shrub invasion, paludification, and compound disturbances) with harvesting. Further, owing to the limited number of years of implementation, longer-term effects of the strategy still need investigating. Many of the unknowns relate to the lack of study sites that encompass multiple scales, longer-term datasets, and alternative silvicultural practices (e.g., partial retention, scarification, and controlled burning). Finally, fine-scale systematic effects of natural disturbance emulation such as potential effects on nutrient cycling still need investigating, particularly given their role in determining long-term stand productivity.

At the stand level, research has successfully identified what structures and features should be maintained and promoted, but has largely failed to relate these features to population or individual productivity, and therefore to amounts of features needed to reach management objectives (Hannon 2005, 2006; NCASI 2009c). Development of such guidelines requires a clear understanding of the desired management outcomes (e.g., should harvested areas "life-boat" species until regeneration is advanced, or should harvested areas maintain similar species as naturally disturbed areas of similar age?), which is often lacking.

Little data exist that adequately quantify what thresholds of retained structures may be suitable for meeting management objectives, if in fact such thresholds exist and can be applied beyond a single landscape or region. However, even if thresholds are identified, different species and groups of species respond differently to retained structures. For example, while mature forest species may respond positively to green-tree retention patches if they are large enough in extent, open area species may respond negatively to such large patches, creating a trade-off between species of differing habitat type. As a result, the best strategy may be to maintain a wide range of retained structure, differing in size and extent.

6.0 CONCLUSIONS

Provincial forest management regulations, long-term forest management plans, sustainable forest management (SFM) certification programs, and other forest management guidance in Canada are in part designed to conserve at-risk biodiversity, maintain significant ecological processes, promote structural diversity, and achieve other objectives that collectively contribute to the conservation of biological diversity. As a result, many criteria used by science-based non-government organizations to measure progress towards achieving conservation objectives and select high-priority areas can be influenced positively by forest management practices implemented within the framework of regulatory and non-regulatory programs. Many conservation planning criteria have a strong scientific basis and may be grouped into three categories: 1) those related to species presence or trends (species richness, endemism, rarity, and presence of special, threatened, or declining taxa); 2) landscape status or trends (representation, significant ecological/evolutionary processes, habitat loss, fragmentation, large intact areas); and 3) future threats (high or low future threats). The criteria differ in the extent to which they incorporate environmental/ecological factors and social values and the ease and rigor with which they may be measured.

Criteria related to species presence or trends are predominantly environmental/ecological in nature. Endemic, rare, threatened, declining, and other special taxa often are globally imperiled or critically imperiled and, therefore, forest management under provincial regulation and the auspices of SFM certification is expected to conserve them on managed forest landscapes. Thus, managers have many opportunities to maintain occurrences of such species through implementation of the science-based management practices embedded in regulation and SFM certification. However, species occurrence or species richness can be promoted through application of sustainable forestry practices on managed forest landscapes, although research has shown that the response of richness to forest management differs depending on factors such as forest management practices in question, the taxonomic group being studied, and time scales being considered.

Criteria related to landscape state or trend often incorporate social values. Forest managers are expected to implement landscape-level measures that promote and maintain habitat diversity. Such directives may be mandated under government guidelines or certification programs. Therefore, managers often seek to minimize forest conversion from one stand type to another or to other land uses and sometimes use a coarse-filter approach to ensure appropriate representation (e.g., Haufler, Mehl, and Roloff 1996). Managers also may promote maintenance of significant or outstanding ecological/environmental processes (e.g., maintenance of water quantity and quality and hydrological processes), and at least partially emulate other processes such as natural disturbance (NCASI 2006).

While future threats are considered by forest managers, the link between future threats and biodiversity is not always clear, and thus the role that managed forests can play in addressing future threats may be limited. Highly forested landscapes, such as those where industrial forest management is most common, often are not immediately threatened by proximate human activities or processes that are causing the “destruction, degradation, and/or impairment of biodiversity and natural processes” (Salafsky et al. 2008, 898). Thus, they are generally afforded protection for biodiversity from immediate threats. In some jurisdictions (e.g., North America) areas under sustainable forest management are not generally recognized as permanently “protected” under IUCN classifications, where interpretation of IUCN classifications tend toward “legislated” as the primary criterion (CCEA 2008). In other jurisdictions (e.g., Europe), areas under sustainable forest management have been interpreted as protected under IUCN classification (Pröbstl, Sowa, and Haider 2010).

Because of the inter-related nature of conservation criteria, the forest industry does not and cannot consider conservation planning criteria individually. Rather, managers address them collectively and, when assessing progress towards conservation objectives, may choose to focus on quantitative measures that have a strong ecological/environmental nature. Managers also commonly face the need to make trade-offs when considering multiple criteria because management to enhance habitat for one species may diminish habitat for another. Giving appropriate consideration to social and economic factors as well as to conservation planning criteria would likely increase the odds of achieving conservation objectives on managed forest areas in a truly sustainable manner. Additional research could strengthen industry conservation practices by testing different approaches for measuring landscape-scale criteria such as fragmentation, identifying thresholds of biodiversity responses to both stand and landscape management practices, and by clarifying biodiversity response to different strategies for emulating natural disturbance.

REFERENCES

Andelman, S.J., and W.F. Fagan 2000. “Umbrellas and Flagships: Efficient Conservation Surrogates or Expensive Mistakes.” *Proceedings of the National Academy of Sciences* 97: 5954-5959.

Arcese, P., and A.R.E. Sinclair. 1997. “The Role of Protected Areas as Ecological Baselines.” *Journal of Wildlife Management* 61: 587–602.

Argus, G.W. 1976. “The Conservation of Canadian Rare and Endangered Plants.” In *Canada’s Threatened Species and Habitats. Proceedings of the Symposium on Canada’s Threatened Species and Habitats*, edited by T. Mosquin and C. Suchal, 139-143. Ottawa, ON: Canadian Nature Federation.

Armstrong, G.W., W.L. Adamowicz, J.A. Beck Jr., S.G. Cumming, and F.K.A. Schmiegelow. 2003. "Coarse Filter Ecosystem Management in a Nonequilibrating Forest." *Forest Science* 49(2): 209-223.

Balisky A. 2007. *Mistik Management Ltd. 2007: 20-Year Forest Management Plan. Volume II.* Meadow Lake, SK: Mistik Management Ltd.

Beissinger, R.R., and M.I. Westphal. 1998. "On the Use of Demographic Models of Population Viability in Endangered Species Management." *Journal of Wildlife Management* 62: 821-841.

Blanchet, S., O. Rey, R. Etienne, S. Lek, and G. Loot 2010. "Species-Specific Responses to Landscape Fragmentation: Implications for Management Strategies." *Evolutionary Applications* 3(3): 291-304.

Boman, J.S., and B.B. Casper 1995. "Differential Postdispersal Seed Predation in Disturbed Intact Temperate Forest." *American Midland Naturalist* 134: 107-116.

Boutin, S., and Hebert, D. 2002. "Landscape Ecology and Forest Management: Developing an Effective Partnership." *Ecological Applications* 12: 390–397.

Brook, B.W., M.A. Burgman, H.R. Akçakaya, J.J. O'Grady, and R. Frankham. 2002. "Critiques of PVA Ask the Wrong Questions: Throwing the Heuristic Baby out with the Numerical Bath Water." *Conservation Biology* 16(1): 262-263.

Bruner, A.G., R.E. Gullison, R.E. Rice, and G.A.B. da Fonseca. 2001. "Effectiveness of Parks in Protecting Tropical Biodiversity." *Science* 291(5501): 125-128.

Bryant, D., D. Nielsen, and L. Tangley. 1997. *The Last Frontier Forests: Ecosystems and Economies on the Edge*. Washington, DC: World Resources Institute (WRI). http://www.globalforestwatch.org/english/pdfs/Last_Frontier_Forests.pdf.

Bunnell, F.L. 1998. "Evading Paralysis by Complexity When Establishing Operational Goals for Biodiversity." *Journal of Sustainable Forestry* 7(3-4): 145-164.

———. 1999a. Foreword: "Let's Kill a Panchreston: Giving Fragmentation a Meaning." In *Forest Wildlife and Fragmentation: Management Implications*, edited by J.A. Rochell, L.A. Lehmann, and J. Wisniewski, vii-xiii. Brill, Leiden, the Netherlands.

———. 1999b. "What Habitat is an Island?" In *Forest Wildlife and Fragmentation: Management Implications*, edited by J.A. Rochell, L.A. Lehmann, and J. Wisniewski, 1-31. Brill, Leiden, the Netherlands.

Bunnell, F.L., and G.B. Dunsworth. 2009. "The Approach." In *Forestry and Biodiversity: Learning how to sustain biodiversity in managed forests*, edited by F.L. Bunnell and G.B. Dunsworth, 36-53. Vancouver, BC: University of British Columbia Press.

Bunnell, F.L., R.W. Campbell, and K.A. Squires. 2004. "Conservation Priorities for Peripheral Species: The Example of British Columbia." *Canadian Journal of Forest Research* 34: 2240-2247.

———. 2005. "Assessing the Need for Species Conservation Action in British Columbia." *BC Journal of Ecosystems and Management* 6(2): 29-37.

Cairns, M.A., and R.T. Lackey. 1992. "Biodiversity and Management of Natural Resources: The Issues." *Fisheries* 17(3): 6-10.

Canadian Council on Ecological Areas (CCEA). 2008. *Canadian Guidebook for the Application of IUCN Protected Areas Categories*. CCEA Occasional Paper No. 18. Ottawa, ON: Canadian Council for Ecological Areas (CCEA) Secretariat.

Canadian Standards Association (CSA) 2003. *Sustainable Forest Management: Requirements and Guidance.* CAN/CSA-Z809-02. Mississauga, ON: Canadian Standards Association.

Cannings, S., M. Anions, R. Rainer, and B. Stein. 2005. *Our Home and Native Land: Canadian Species of Global Conservation Concern*. Ottawa, ON: NatureServe Canada.

Caro, T.M., and G. O'Doherty 1999. "On the Use of Surrogate Species in Conservation Biology." *Conservation Biology* 13: 805-814.

Chape, S., S. Blyth, L. Fish, P. Fox, and M. Spalding. 2003. *2003 United Nations List of Protected Areas*. Gland, Switzerland and Cambridge, UK: International Union for the Conservation of Nature and Natural Resources, and UN Environment Programme World Conservation Monitoring Centre.

Cincotta, R.P., J. Wisnewski, and R. Engelman. 2000. "Human Population in the Biodiversity Hotspots." *Science* 404: 990-992.

Colwell, R.K., and J.A. Coddington. 1994. "Estimating Terrestrial Biodiversity through Extrapolation." *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences* 345: 101-118.

Cowling, R.M., and M.J. Samways 1995. "Predicting Global Patterns of Endemic Plant Species Richness." *Biodiversity Letters* 2: 127-131.

Crow, T.R., and A.H. Perera 2004. "Emulating Natural Landscape Disturbance in Forest Management—An Introduction." *Landscape Ecology* 19: 231-233.

Czech, B., and P. R. Krausman. 2001. *The Endangered Species Act: History, Conservation Biology, and Public Policy*. Baltimore, MD: Johns Hopkins University Press.

Dale, V.H., and S.C. Beyeler 2001. "Challenges in the Development and Use of Ecological Indicators." *Ecological Indicators* 1: 3-10.

Davidson, C. 1998. "Issues in Measuring Landscape Fragmentation." *Wildlife Society Bulletin* 26(1): 32-37.

Davies, K.F., C.R. Margules, and J.F. Lawrence. 2000. "Which Traits of Species Predict Population Declines in Experimental Forest Fragments?" *Ecology* 81(5): 1450-1461.

———. 2004. "A Synergistic Effect Puts Rare, Specialized Species at Greater Risk of Extinction." *Ecology* 85(1): 265-271.

de Thoisy, B., S. Brosse, and M.A. Dubois 2008. "Assessment of Large-Vertebrate Species Richness and Relative Abundance in Neotropical Forest Using Line-Transect Census: What Is the Minimal Effort Required?" *Biodiversity and Conservation* 17(11): 2627-2644.

DeLong, D.C. Jr. 1996. "Defining Biodiversity." *Wildlife Society Bulletin* 24: 738-749.

Dorazio, R.M., J.A. Royle, B. Söderström, and A. Glimskär. 2006. Estimating species richness and accumulation by modeling species occurrence and detectability. *Ecology* 87(4): 842-854.

Drapeau, P., A. Leduc, J.-F. Giroux, J.-P.L. Savard, Y. Bergeron, and W.L. Vickery 2000. "Landscape-Scale Disturbances and Changes in Bird Communities of Boreal Mixed-Wood

Forests." *Ecological Monographs* 70: 423-444.

Duinker, P.N., Y.F. Wiersma, W. Haider, G.T. Hvenegaard, and F.K.A. Schmiegelow 2010. "Protected Areas and Sustainable Forest Management: What Are We Talking About"? *The Forestry Chronicle* 86: 173-177.

Dyer, S. J., J.P. O'Neill, S.M. Wasel, and S. Boutin 2002. "Quantifying Barrier Effects of Roads and Seismic Lines on Movements of Female Woodland Caribou in Northeastern Alberta." *Canadian Journal of Zoology* 80(5): 839-845.

Ehrenfeld, D. 1978. *The Arrogance of Humanism*. New York: Oxford University Press.

Ellner, S.P., J. Fieberg, D. Ludwig, and C. Wilcox. 2002. "Precision of Population Viability Analysis." *Conservation Biology* 16: 258-261.

Environment Canada. 2004. *Threats to Water Availability in Canada*. NWRI Scientific Assessment Report Series No. 3 and ACSD Science Assessment Series No. 1. Burlington, ON: National Water Research Institute.

Fahrig, L. 2003. "Effects of Habitat Fragmentation on Biodiversity." *Annual Review of Ecology, Evolution and Systematics* 34: 487-515.

Farnsworth, N.R. 1988. "Screening Plants for New Medicines." In *Biodiversity*, edited by E.O. Wilson, 83-97. Washington, DC: National Academy Press.

Fieberg, J., and S.P. Ellner. 2000. "When Is It Meaningful to Estimate an Extinction Probability?" *Ecology* 81: 2040-2047.

Forest Stewardship Council (FSC) 1996. *FSC Principles and Criteria for Forest Stewardship*. FSC-STD-01-001 (version 4-0) EN. Bonn, Germany: Forest Stewardship Council.

Fortin, D., R. Courtois, P. Etcheverry, C. Dussault, and A. Gingras 2008. "Winter Selection of Landscapes by Woodland Caribou: Behavioural Response to Geographical Gradients in Habitat Attributes." *Journal of Applied Ecology* 45: 1392-1400.

Franklin, J.F. 1993. "Preserving Biodiversity: Species, Ecosystems, or Landscape?" *Ecological Applications* 3(2): 202-205.

Gaston, K.J., R.L. Pressey, and C.R. Margules, 2002. "Persistence and Vulnerability: Retaining Biodiversity in the Landscape and in Protected Areas." *Journal of Biosciences* 27(4): 361-384.

Gauthier, S., M.-A. Vaillancourt, A. Leduc, L. De Grandpré, D. Kneeshaw, H. Morin, P. Drapeau, and Y. Bergeron (eds). 2009. *Ecosystem Management in the Boreal Forest*. Québec: Presses de l'Université du Québec.

Gibbs, K.E., R.L. Mackey, and D.J. Currie. 2009. "Human Land Use, Agriculture, Pesticides and Losses of Imperiled Species." *Diversity and Distributions* 15: 242-253.

Gordon, E. A., O.E. Franco, and M.L. Tyrrell. 2005. *Protecting Biodiversity: A Guide to Criteria Used by Global Conservation Organizations*. New Haven, CT: Yale School of Forestry &Environmental Studies Publication Series.

Gross, A.O. 1931. "Banding the Last Heath Hen." *Bird-Banding* 2(3): 99-105.

Haber, E. 1994. *Endemic Vascular Plants of Canada Database: Documentation and Summary of Contents*. Database and report prepared for the National Atlas Information Service in cooperation with the Canadian Museum of Nature. Unpublished. Feb. 1994.

Haila, Y. 2002. “A Conceptual Genealogy of Fragmentation Research: From Island Biogeography to Landscape Ecology.” *Ecological Applications* 12: 321-334.

Hamilton, J.G., A.R. Zangerl, M.R. Berenbaum, J. Pippen, M. Aldea, and E.H. DeLucia. 2004. “Insect Herbivory in an Intact Forest Understory under Experimental CO_2 Enrichment.” *Oecologia* 138: 566-573.

Hannon, S. 2005. *Effect of Stand versus Landscape Level Forest Structure on Species Abundance and Distribution*. Edmonton, AB: Sustainable Forest Management Network.

———. 2006. “Building a Bird-Friendly Forest: Lessons from the Boreal Forest of Canada.” *Acta Zoologica Sinica* 52 (Supplement): 687-691.

Haufler, J. B., C. A. Mehl, and G. J. Roloff. 1996. “Using a Coarse-Filter Approach with Species Assessment for Ecosystem Management.” *Wildlife Society Bulletin* 24(2):200-208.

Hobson, K.A., and J. Schieck 1999. “Changes in Bird Communities in Boreal Mixedwood Forest: Harvest and Wildfire Effects over 30 Years.” *Ecological Applications* 9: 849-863.

Hoekstra, J.M., T.M. Boucher, T.H. Ricketts, and C. Roberts. 2005. “Confronting a Biome Crisis: Global Disparities of Habitat Loss and Protection.” *Ecology Letters* 8: 3–29.

Houde, I., F.L. Bunnell, and S. Leech. 2005. “Assessing Success at Achieving Biodiversity Objectives in Managed Forests.” *BC Journal of Ecosystems and Management* 6(2): 17-28.

Huffaker, C.B. 1958. “Experimental studies on predation: dispersion factors and predator-prey oscillations.” *Hilgardia* 27: 343-383.

Hughes, J.W., and T.J. Fahey 1991. “Colonization Dynamics of Herbs and Shrubs in a Disturbed Northern Hardwood Forest.” *Journal of Ecology* 79: 605-616.

Hughes, R.M., and R.F. Noss 1992. “Biological Diversity and Biological Integrity: Current Concerns for Lakes and Streams.” *Fisheries* (Bethesda) 17(3): 11-19.

Hunsaker, C. T., and D. E. Carpenter, eds. 1990. *Ecological Indicators for the Environmental Monitoring and Assessment Program.* EPA 528/3-90-060. Research Triangle Park, NC: United States Environmental Protection Agency, Office of Research and Development.

Hunter, M.L. Jr. 1991. “Coping with Ignorance: The Coarse-Filter Strategy for Maintaining Biological Diversity.” In *Balancing on the Brink of Extinction*, edited by K. Kohm, 266-281. Washington, DC: Island Press.

Ice, G.G., and Schoenholtz, S.H. 2003. “Understanding How Extremes Influence Water Quality: Experience from Forest Watersheds.” *Hydrological Science and Technology* 19(1-4):99-107.

International Union for Conservation of Nature (IUCN). 1997. *North America Centres of Plant Diversity and Endemism: Canada*. Gland, Switzerland and Cambridge, UK: International Union for the Conservation of Nature and Natural Resources, accessed April 1, 2011, www.nmnh.si.edu/botany/projects/cpd/na/na-cpde.htm#canada.

Jennings, M.D. 2000. “Gap Analysis: Concepts, Methods, and Recent Results.” *Landscape Ecology* 15: 5–20.

Jetz, W., C. Rahbek, and R.K. Colwell 2004. “The Coincidence of Rarity and Richness and the Potential Signature of History in Centres of Endemism.” *Ecology Letters* 7: 1180-1191.

Johnson, N. C. 1995. *Biodiversity in the Balance: Approaches to Setting Geographic Conservation Priorities*. Washington, DC: Biodiversity Support Program, c/o World Wildlife Fund.

Kerr, J.T. 1996. "Species Richness, Endemism, and the Choice of Areas for Conservation." *Conservation Biology* 11(5): 1094-1100.

Kirkland, G.L., and R.S. Ostfeld. 1999. "Factors Influencing Variation among States in the Number of Federally Listed Mammals in the United States." *Journal of Mammalogy* 80: 711-719.

Lackey, R.T. 2006. "Axioms of Ecological Policy." *Fisheries* 31(6): 286-290.

Lambeck, R.J. 1997. "Focal Species: A Multi-Species Umbrella for Nature Conservation." *Conservation Biology* 11: 849-856.

———. "Focal Species and Restoration Ecology: Response to Lindenmayer et al." *Conservation Biology* 16: 549-551.

Lee, P., D. Aksenov, L. Laestadius, R. Nogueron, and W. Smith. 2003. *Canada's Large Intact Forest Landscapes*. Edmonton, AB: Global Forest Watch Canada.

Lemelin, R.H. 2008. "Impacts of the Cancellation of the Spring Bear Hunt in Ontario, Canada." *Human-Wildlife Conflicts* 2(2): 148-150.

Liechty, H.O., M.J. Holmes, D.D. Reed, and G.D. Mroz. 1992. "Changes in Microclimate after Stand Conversion in Two Northern Hardwood Stands." *Forest Ecology and Management* 50(3-4): 253-264.

Lindenmayer, D.B., and J. Fischer. 2006. *Habitat Fragmentation and Landscape Change: An Ecological and Conservation Synthesis*. Washington D.C.: Island Press.

Lindenmayer, D.B., A.D. Manning, P.L. Smith, H.P. Possingham, J. Fisher, I. Oliver, and M.A. McCarthy 2002. "The Focal-Species Approach and Landscape Restoration: A Critique." *Conservation Biology* 16: 338-345.

Liu, J., M. Linderman, Z. Ouyang, L. An, J. Yang, and H. Zhang. 2001. "Ecological Degradation in Protected Areas: The Case of Wolong Nature Reserve for Giant Pandas." *Science* 292: 98-101.

Luckey, J.R., and S.S. Crawford. 2009. "Consistency of COSEWIC Species at Risk Designations: Freshwater Fishes as a Case Study." *Canadian Journal of Fisheries and Aquatic Sciences* 66: 959-971.

Ludwig, D. 1996. "Uncertainty and the Assessment of Extinction Probabilities." *Ecological Applications* 6: 1067-1076.

MacArthur, R.H., and Wilson, E.O. 1963. "An Equilibrium Theory of Insular Zoogeography." *Evolution* 17: 373-387.

———. 1967. *The Theory of Island Biogeography*. Monographs in Population Biology. Princeton, NJ: Princeton University Press.

MacDonald, L.H. 2000. "Evaluating and Managing Cumulative Effects: Process and Constraints." *Environmental Management* 26(3): 299-315.

MacNally, R., and A.F. Bennett 1997. "Species-Specific Predictions of the Impact of Habitat Fragmentation: Local Extinction of Birds in the Box-Ironbark Forests of Central Victoria, Australia." *Biological Conservation* 82: 147-155.

MacNally, R., A.F. Bennett, and G. Horrocks 2000. "Forecasting the Impacts of Habitat Fragmentation. Evaluation of Species-Specific Predictions of the Impact of Habitat Fragmentation on Birds in the Box-Ironbark Forests of Central Victoria, Australia." *Biological Conservation* 95: 7-29.

Margules, C.R., A.O. Nicholls, and R.L. Pressey 1988. "Selecting Networks of Reserves to Maximise Biological Diversity." *Biological Conservation* 43: 63-76.

Margules, C.R., and R.L. Pressey. 2000. "Systematic Conservation Planning." *Nature* 405: 243–253.

Master, L., D. Faber-Langendoen, R. Bittman, G. A. Hammerson, B. Heidel, J. Nichols, L. Ramsay, and A. Tomaino. 2009. *NatureServe Conservation Status Assessments: Factors for Assessing Extinction Risk*. Arlington, VA: NatureServe.

McBroom, M.W., Beasley, R.S., Chang, M., Gowin, B., and Ice, G. 2003. "Runoff and Sediment Losses from Annual and Unusual Storm Events from the Alto Experimental Watersheds, Texas: 23 Years after Silvicultural Treatments." In *Proceedings of the First Interagency Conference on Research in the Watersheds*, edited by K.G. Renard, S.A. McElroy, W.J. Gburek, H.E. Canfield, and R.L. Scott, 607-613. United States Department of Agriculture, Agricultural Research Service.

McCloskey, J. M., and H. Spalding. 1989. "A Reconnaissance Level Inventory of the Amount of Wilderness Remaining in the World." *Ambio* 18(4):221–227.

McKee, J.K., P.W. Sciullia, C.D. Fooce, and T.A. Waite. 2003. "Forecasting Global Biodiversity Threats Associated with Human Population Growth." *Biological Conservation* 115:161-164.

McNeely, J.A. 1988. *Economics and biological diversity*. Gland, Switzerland: International Union for Conservation of Nature and Natural Resources (IUCN).

McNeely, J., and K. Miller 1984. *National Parks Conservation and Development: The Role of Protected Areas in Sustaining Society*. Washington, DC: Smithsonian Institute Press.

Mech, L.D. 1967. "Telemetry as a Technique in the Study of Predation." *Journal of Wildlife Management* 31(3): 492-496.

Merriam-Webster OnLine, s.v. "intact," accessed March 31, 2011, http://www.merriam-webster.com/dictionary/intact.

Mooers, A.O., L.R. Prugh, M. Festa-Bianchet, and J.A. Hutchings 2007. "Biases in Legal Listing under Canadian Endangered Species Legislation." *Conservation Biology* 21: 572-575.

Myers, N., R.A. Mittermeier, C.G. Mittermeier, G.A.B. da Fonseca, and J. Kent 2000. "Biodiversity Hotspots for Conservation Priorities." *Nature* 403: 853-858.

Naidu, S.L., and E.H. Delucia 1999. "First-Year Growth Response of Trees in an Intact Forest Exposed to Elevated CO_2." *Global Change Biology* 5: 609-613

National Council for Air and Stream Improvement, Inc. (NCASI). 2006. *Similarities and Differences Between Harvesting- and Wildfire-Induced Disturbances in Fire-Mediated Canadian Landscapes*. Technical Bulletin No. 924. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc.

———. 2008. *Fragmentation in the Boreal Forest and Possible Effects on Terrestrial Wildlife*. Technical Bulletin No. 959. Research Triangle Park, NC: National Council for Air and Stream Improvement.

———. 2009a. *Effects of Forest Management on Water Resources in Canada: A Research Review.* Technical Bulletin No. 969. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc.

———. 2009b. *Water Profile of the United States Forest Products Industry*. Technical Bulletin No. 960. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc.

———. 2009c. *Wildlife Response to Stand-Level Structural Retention Practices in the Boreal Forest*. Technical Bulletin No. 964. Research Triangle Park, NC, National Council for Air and Stream Improvement.

———. 2010. *Water profile of the Canadian Forest Products Industry*. Technical Bulletin No. 975. Research Triangle Park, NC: National Council for Air and Stream Improvement, Inc.

Natural Resources Canada (NRCAN) 2009. *The State of Canada's Forests: Annual Report 2009.* Ottawa, ON: Natural Resources Canada.

Newark, W.D. 1996. "Insularization of Tanzanian parks and the Local Extinction of Large Mammals." *Conservation Biology* 10(6): 1549-1556.

Norse, E.A., Ecological Society of America, and Wilderness Society (US). 1986. *Conserving Biological Diversity in Our National Forests*. Washington, DC: Wilderness Society.

Noss, R.F. 1987. "From Plant Communities to Landscapes in Conservation Inventories: A Look at the Nature Conservancy (USA)." *Biological Conservation* 41: 11-37.

———. 1989. "Indicators for Monitoring Biodiversity: A Hierarchical Approach." *Conservation Biology* 4(4): 355-364.

Office of Technology Assessment (OTA) 1987. *Technologies to Maintain Biological Diversity*. OTA-F-330. Washington, DC: US Government Printing Office.

Olson, D.M., and E. Dinerstein 1998. The Global 200: "A Representation Approach to Conserving the Earth's Most Biologically Valuable Ecoregions." *Conservation Biology* 12: 502-515.

Ontario Ministry of Natural Resources (OMNR). 1987. *Bald Eagle Habitat Management Guidelines*. Thunder Bay, ON: Ontario Ministry of Natural Resources.

———. 1996. *Forest Management Planning Manual for Ontario's Crown Forests*. Toronto, ON: Queen's Printer for Ontario.

———. 2001a. *Forest Management Guide for Natural Disturbance Pattern Emulation*, Version 3.1. Toronto, ON: Queen's Printer for Ontario.

———. 2001b. *Silvicultural Effectiveness Monitoring Manual for Ontario*. Toronto, ON: Queen's Printer for Ontario.

Orme, C.D., R.G. Davies, M. Burgess, F. Eigenbrod, N. Pickup, V.A. Olson, A.J. Webster, T. Ding, P.C. Rasmussen, R.S. Ridgely, A.J. Stattersfield, P.M. Bennett, T.M. Blackburn, K.J. Gaston, and I.P.F. Owens. 2005. "Global Hotspots of Species Richness Are Not Congruent with Endemism or Threat." *Nature* 436: 1016-1019.

Paine, R.T. 1966. "Food Web Complexity and Species Diversity." *American Naturalist* 100: 65-75.

Pearson, D.L., and S.S. Carroll. 1998. "Global Patterns of Species Richness: Spatial Models for Conservation Planning Using Bioindicator and Precipitation Data." *Conservation Biology* 12(4): 809-821.

Perera, A.H., and L.J. Buse 2004. "Emulating Natural Disturbance in Forest Management: An Overview." In *Emulating Natural Forest Landscape Disturbance: Concepts and Applications*, edited by A.H. Perera, L.J. Buse, and M.G. Weber, 3-7. New York: Columbia University Press.

Poor, C.J., and McDonnell, J.J. 2007. "The Effects of Land Use on Stream Nitrate Dynamics." *Journal of Hydrology* 332:54-68.

Potapov P., M.C. Hansen, S.V. Stehman, T.R. Loveland, and K. Pittman. 2008a. "Combining MODIS and Landsat Imagery to Estimate and Map Boreal Forest Cover Loss." *Remote Sensing of Environment* 112(9):3708–3719.

Potapov P., A. Yaroshenko, S. Turubanova, M. Dubinin, L. Laestadius, C. Thies, D. Aksenov, A. Egorov, Y. Yesipova, I. Glushkov, M. Karpachevskiy, A. Kostikova, A. Manisha, E. Tsybikova, and I. Zhuravleva. 2008b. "Mapping the world's intact forest landscapes by remote sensing." *Ecology and Society* 13(2) 51. http://www.ecologyandsociety.org/vol13/iss2/art51/.

Potvin, F., L. Bélanger, and K. Lowell 2000. "Marten Habitat Selection in a Clearcut Boreal Landscape." *Conservation Biology* 14: 844-857.

Prendergast, J.R., R.M. Quinn, J.H. Lawton, B.C. Eversham, and D.W. Gibbons. 1993. "Rare Species, the Coincidence of Diversity Hotspots and Conservation Strategies." *Nature* 365: 335-337.

Pressey, R.L., 1994. "*Ad Hoc* Reservations: Forward or Backwards Steps in Developing Representative Reserve Systems?" *Conservation Biology* 8: 662–668.

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

Preston, F.W. 1962a. The canonical distribution of commonness and rarity: Part I. *Ecology* 43: 185-215.

———. 1962b. The canonical distribution of commonness and rarity. Part II. *Ecology* 43: 410-432.

Pröbstl, U., M. Sowa, and W. Haider 2010. *Sustainable Forest Management and Protected Areas: Perspectives from Central Europe*. Supplement to *Relationships between Protected Areas and Sustainable Forest Management: Many Shades of Green: A State of Knowledge Report*, edited by Y.F. Wiersma, P.N Duinker, W. Haider, G.T. Hvenegaard, and F.K.A. Schmiegelow. Edmonton, AB: Sustainable Forest Management Network.

Rabinowitz, D. 1981. "Seven Forms of Rarity." In *The Biological Aspects of Rare Plant Conservation*, edited by H. Synge, 205-217. Chichester, UK: John Wiley & Sons.

Rabinowitz, D., S. Cairns, and T. Dillon. 1986. "Seven forms of rarity and their frequency in the flora of the British Isles." In *Conservation Biology: The Science of Scarcity and Diversity,* edited by M. Soulé, 182-204. Sunderland, MA: Sinauer Associates.

Racey, G. D., K. Abraham, W.R. Darby, H.R. Timmerman, and Q. Day. 1991. "Can Woodland Caribou and the Forest Industry Coexist: The Ontario Scene." *Rangifer Special Issue* 7:108-115.

Racey, G., A. Harris, L. Gerrish, E. Armstrong, J. McNicol, and J. Baker. 1999. *Forest Management*

Guidelines for the Conservation of Woodland Caribou: A Landscape Approach. Thunder Bay, ON: Ontario Ministry of Natural Resources.

Reid, W.V. 1998. "Biodiversity Hotspots." *Trends in Ecology and Evolution* 13(7): 275–280.

Ridder, B. 2008. "Questioning the Ecosystem Services Argument for Biodiversity Conservation." *Biodiversity Conservation* 17: 781-790.

Riffell, S., J. P. Verschuyl, D. Miller, and T. B. Wigley. 2011. "Biofuel Harvests, Coarse Woody Debris, and Biodiversity – A Meta-Analysis." *Forest Ecology and Management* 261(4): 878-887.

Rodrigues, A.S.L., H.R. Akçakaya, S.J. Andelman, M.I. Bakarr, L. Boitani, T.M. Brooks, J.S. Chanson, L.D.C. Fishpool, G.A.B. Da Fonseca, K.J. Gaston, M. Hoffmann, P.A. Marquet, J.D. Pilgrim, R.L. Pressey, J. Schipper, W. Sechrest, S.N. Stuart, L.G. Underhill, R.W. Waller, M.E.J. Watts, and X. Yan. 2004. "Global Gap Analysis: Priority Regions for Expanding the Global Protected-Area Network." *BioScience* 54(12) 1092-1100.

Rodrigues, A.S.L., and K.J. Gaston 2002. "Rarity and conservation planning across geopolitical units." *Conservation Biology* 16: 674-682.

Rohner, C., and C.J. Krebs 1998. "Response of Great Horned Owls to Experimental 'Hot Spots' of Snowshoe Hare Density." *The Auk* 115: 694-705.

Rosenzweig, M.L. 1995. *Species Diversity in Space and Time*. Cambridge, UK: Cambridge University Press.

Rowe, J. S. 1972. *Forest Regions of Canada*. Ottawa, ON: Department of Fisheries and Environment.

Sæther, B-E., S. Engen, R. Lande, P. Arcese, and J.N.M. Smith. 2000. "Estimating the Time to Extinction in an Island Population of Song Sparrows." *Proceedings of the Royal Society of London, Series B* 267: 621-626.

Sagarin, R.D., and S.D. Gaines 2002. "The 'Abundant Centre' Distribution: To What Extent Is It a Biogeographical Rule?" *Ecology Letters* 5: 137-147.

Salafsky, N., D. Salzer, A. J. Stattersfield, C. Hilton-Taylor, R. Neugarten, S. H. M. Butchart, B. Collen, N. Cox, L. L. Master, S. O'Connor, and D. Wilkie. 2008. "A Standard Lexicon for Biodiversity Conservation: Unified Classifications of Threats and Actions." *Conservation Biology* 22:897-911.

Sanderson, E.W., M. Jaiteh, M.A. Levy, K.H. Redford, A.V. Wannebo, and G. Woolmer. 2002. "The Human Footprint and the Last of the Wild." *BioScience* 52(10):891–904.

Sarkar, S., R. L. Pressey, D. P. Faith, C. R. Margules, T. Fuller, D. M. Stoms, A. Moffett, K. A. Wilson, K. J. Williams, P. H. Williams, and S. Andelman. 2006. "Biodiversity Conservation Planning Tools: Present Status and Challenges for the Future." *Annual Review of Environment and Resources* 31(1):123-159.

Schiller, A., C. T. Hunsaker, M. A. Kane, A. K. Wolfe, V. H. Dale, G. W. Suter, C. S. Russell, G. Pion, M. H. Jensen, and V. C. Konar. 2001. "Communicating Ecological Indicators to Decision Makers and the Public." *Conservation Ecology* 5(1): 19. http://www.consecol.org/vol5/iss1/art19/.

Schmiegelow, F.K.A., D.P. Stepnisky, C.A. Stambaugh, and M. Koivula 2006. "Reconciling Salvage Logging of Boreal Forests with a Natural Disturbance Management Model." *Conservation*

Biology 20(4): 971-983.

Schnitzer, A.S., and W.P. Carson. 2001. "Treefall Gaps and the Maintenance of Species Diversity in a Tropical Forest." *Ecology* 82: 913-919.

Schorger, A.W. 1955. *The Passenger Pigeon: Its Natural History and Extinction*. Madison, WI: University of Wisconsin.

Seymour, R.S., and M.L. Hunter, Jr. 1992. *New Forestry in Eastern Spruce-Fir Forests: Principles and Applications to Maine*. Miscellaneous Publication 716. Orono, ME: University of Maine, Maine Agricultural and Forest Experimental Station.

Sierra, R., F. Campos, and J. Chamberlin 2002. "Assessing Biodiversity Conservation Priorities: Ecosystem Risk and Representativeness in Continental Ecuador." *Landscape and Urban Planning* 59: 95-110.

Simon, N.P.P., F.E. Schwab, and R.D. Otto. 2002. "Songbird Abundance in Clear-Cut and Burned Stands: A Comparison of Natural Disturbance and Forest Management." *Canadian Journal of Forest Research* 32: 1343-1350.

Sinclair, A.R.E. 1998. "Natural Regulation of Ecosystems in Protected Areas as Ecological Baselines." *Wildlife Society Bulletin* 26: 399–409.

Sinclair, A.R.E., S.A.R. Mduma, and P. Arcese 2002. "Protected Areas as Biodiversity Benchmarks for Human Impact: Agricultural and Serengeti Avifauna." *Proceedings of the Royal Society of London B* 269: 2401-2405.

Sleep, D.J.H., M.C. Drever, and K.J. Szuba. 2009. "Potential Role of Spruce Budworm in Range-Wide Decline of Canada Warbler." *Journal of Wildlife Management* 73(4): 546–555.

Sokal, R., and F.J. Rohlf. 1981. *Biometry*. 2nd ed. San Francisco: W.H. Freeman.

Spears, J. 1988. "Preserving Biological Diversity in the Tropical Forests of the Asian Region. In *Biodiversity*, edited by E.O. Wilson, 393-402. Washington, DC: National Academy Press.

Sustainable Forestry Initiative (SFI) 2010. *SFI 2010-2014 Standard*. Washington, DC: Sustainable Forestry Initiative. http://www.sfiprogram.org/sustainable_forestry_initiative_standard.php.

Sutherland, W.J., M.J. Bailey, I.P. Bainbridge, T. Brereton, J.T.A. Dick, J. Drewitt, N.K. Dulvy, N.R. Dusic, R.P. Freckleton, K.J. Gaston, P.M. Glider, R.E. Green, A.L. Heathwaite, S.M. Johnson, D.W. MacDonald, R. Mitchell, D. Osborn, R.P. Owen, J. Pretty, S.V. Prior, H. Prosser, A.S. Pullin, P. Rose, A. Stott, T. Tew, C.D. Thomas, D.B.A. Thompson, J.A. Vickery, M. Walker, C. Walmsley, S. Warrington, A.R. Watkinson, R.J. Williams, R. Woodroffe, and H.J. Woodroof. 2008. "Future Novel Threats and Opportunities Facing UK Biodiversity Identified by Horizon Scanning." *Journal of Applied Ecology* 45: 821-833.

Sutherland, W.J., S. Bardleys, L. Bennun, M. Clout, I.M. Côté, M.H. Depledge, L.V. Dicks, A.P. Dobson, L. Fellman, E. Fleishman, D.W. Gibbons, A.J. Impey, J.H. Lawton, F. Lickorish, D.B. Lindenmayer, T.E. Lovejoy, R. MacNally, J. Madgwick, L.S. Peck, J. Pretty, S.V. Prior, K.H. Redford, J.P.W. Scharlemann, M. Spalding, and A.R. Watkinson. 2011. "Horizon Scan of Global Conservation Issues for 2011." *Trends in Ecology and Evolution* 26(1): 10-16.

Sutherland, W.J., M. Clout, I.M. Côté, P. Daszak, M.H. Depledge, L. Fellman, E. Fleishman, R. Garthwaite, D.W. Gibbons, J. De Lurio, A.J. Impey, F. Lickorish, D. Lindenmayer, J. Madgwick, C. Margerison, T. Maynard, L.S. Peck, J. Pretty, S. Prior, K.H. Redford, J.P.W.

Scharlemann, M. Spalding and A.R. Watkinson. 2009. "Horizon Scan of Global Conservation Issues for 2010." *Trends in Ecology and Evolution* 25(1): 1-7.

Taylor, P.W. 1986. *Respect for Nature. A Theory of Environmental Ethics*. Princeton, NJ: Princeton University Press.

Tear, T. H., P. Kareiva, P. L. Angermeier, P. Comer, B. Czech, R. Kautz, L. Landon, D. Mehlman, K. Murphy, M. Ruckleshaus, L. M. Scott, and G. Wilhere. 2005. "How Much Is Enough? The Recurrent Problem of Setting Measurable Objectives in Conservation." *BioScience* 55(10):835-849.

Thomas, C.D., and H.C. Mallory 1985. "Rarity, Species Richness and Conservation: Butterflies of the Atlas Mountains in Morocco." *Biological Conservation* 33: 95-117.

Thompson, K., and A. Jones. 1999. "Human Population Density and Prediction of Local Plant Extinction in Britain." *Conservation Biology* 13: 185-189.

Trejo, I., and R. Dirzo 2000. "Deforestation of Seasonally Dry Tropical Forest: A National and Local Analysis in Mexico." *Biological Conservation* 94: 133-142.

Ugland, K.I., J.S. Gray, and K.E. Ellingsen. 2003. "The Species-Accumulation Curve and Estimation of Species Richness." *Journal of Animal Ecology* 72: 888-897.

Venter, O., N.N. Brodeur, L. Nemiroff, B. Belland, I.J. Dolinsek, and J.W. Grant. 2006. "Threats to Endangered Species in Canada." *BioScience* 56(11): 903-910.

Verschuyl, J., S. Riffell, D. Miller, and T. B. Wigley. 2011. "Biodiversity response to Intensive Biomass Production from Forest Thinning in North American Forests - A Meta-Analysis." *Forest Ecology and Management* 261:221-232.

Vié, J.-C., C. Hilton-Taylor, and S.N. Stuart, eds. 2009. *Wildlife in a Changing World—An Analysis of the 2008 IUCN Red List of Threatened Species*. Gland, Switzerland: International Union for Conservation of Nature and Natural Resources.

Watson, D.M. 2002. "A Conceptual Framework for Studying Species Composition in Fragments, Islands and Other Patchy Ecosystems." *Journal of Biogeography* 29: 823-834.

West, N.E. 1994. "Biodiversity and Land Use." In *Sustainable Ecological Systems: Implementing an Ecological Approach to Land Management*, W.W. Covington and L.F. DeBano, technical coordinators, 21-36. General Technical Report GTR-RM-247. Collins, CO: United States Department of Agriculture Forest Service.

Wiersma, Y.F., and Nudds, T.D. 2009. "Efficiency and Effectiveness in Representative Reserve Design in Canada: The Contribution of Existing Protected Areas." *Biological Conservation* 142: 1639-1646.

Wilcove, D.S., D. Rothstein, J. Dubow, A. Phillips, and E. Losos. 1998. "Quantifying Threats to Imperiled Species in the United States." *BioScience* 48(8): 607-615.

Wilczynski, C.J., and S.T.A. Pickett 1993. "Fine Root Biomass within Experimental Canopy Gaps: Evidence for Below-Ground Gap." *Journal of Vegetation Science* 4: 571-574.

Williams, C.B. 1943. "Area and Number of Species." *Nature* 152: 264-267.

Williams, P.H., and K.J. Gaston 1994. "Measuring More of Biodiversity: Can Higher-Taxon Richness Predict Wholesale Species Richness?" *Biological Conservation* 67: 211-217.

Williams, T.M., Hook, D.D., Lipscomb, D.J., Zeng, X., and Albiston, J.W. 2000. “Effectiveness of Best Management Practices to Protect Water Quality in South Carolina Piedmont.” In *Tenth Biennial Southern Silvicultural Research Conference Proceedings*. General Technical Report SRS-30. Asheville, NC: United States Department of Agriculture Forest Service, Southern Research Station.

Witiw, J. 2006. *DMI Strategy Framework for the Identification and Maintenance of a Local Continuous Reserve Network. A Proposed Forest Resources Strategy*. Daishowa Marubeni International Ltd. Peace River Pulp Division, Forest Resources.

World Commission on Environment and Development (WCED). 1987. *Our Common Future*. Oxford, UK Oxford University Press.

World Conservation Union 1994. Guidelines for protected area management categories. IUCN/CNPPA/WCMC, Gland, Switzerland/Cambridge, UK.

WWF International. 2007. *High Conservation Value Forests: The Concept in Theory and Practice*. Gland, Switzerland: Forests for Life Programme, WWF International.

Wye Group. 2002. *Endangered Forests: Priority High Conservation Value Forests for Protection Guidance for Corporate Commitments*, accessed March 31, 2011, http://www.environmentalpaper.org/documents/EF-Report.pdf.

Zaviezo, T., A. A. Grez, C. F. Estades, and A. Pérez. 2006. “Effects of Habitat Loss, Habitat Fragmentation, and Isolation on the Density, Species Richness, and Distribution of Ladybeetles in Manipulated Alfalfa Landscapes.” *Ecological Entomology* 31:646–656.

APPENDIX A

ENDEMIC SPECIES IN CANADA (FROM CANNINGS 2005)

Global Rank*	Scientific Name	Common Name (English)	Common Name (French)	Province/Territory
Mammals (2)				
G1	*Marmota vancouverensis*	Vancouver Island Marmot	Marmotte de I'île Vancouver	BC
G3	*Sorex maritimensis*	Maritime Shrew	Musaraigne des Maritimes	NB, NS
Freshwater Fishes (15)				
GX	*Coregonus species 1*	Dragon Lake Whitefish	Corégone du lac Dragon	BC
GX	*Gasterosteus sp. 12*	Hadley Lake Limnetic Stickleback	Épinoche limnétique du lac Hadley	BC
GX	*Gasterosteus sp. 13*	Hadley Lake Benthic Stickleback	Épinoche benthique du lac Hadley	BC
G1	*Coregonus huntsmani*	Atlantic Whitefish	Corégone de l'Atlantique	NS
G1	*Cottus sp. 2*	Cultus Pygmy Sculpin	Chabot pygmée	BC
G1	*Gasterosteus sp. 1*	Giant Stickleback	Épinoche géante	BC
G1	*Gasterosteus sp. 16*	Vananda Creek Limnetic Stickleback	Épinoche limnétique du ruisseau Vananda	BC
G1	*Gasterosteus sp. 17*	Vananda Creek Benthic Stickleback	Épinoche benthique du ruisseau Vananda	BC
G1	*Gasterosteus sp. 2*	Enos Lake Limnetic Stickleback	Épinoche limnétique du lac Enos	BC
G1	*Gasterosteus sp. 3*	Enos Lake Benthic Stickleback	Épinoche benthique du lac Enos	BC
G1	*Gasterosteus sp. 4*	Paxton Lake Limnetic Stickleback	Épinoche limnétique du lac Paxton	BC
G1	*Gasterosteus sp. 5*	Paxton Lake Benthic Stickleback	Épinoche benthique du lac Paxton	BC
G1	*Lampetra macrostoma*	Lake Lamprey	Lamproie du lac Cowichan	BC
G1	*Moxostoma hubbsi*	Copper Redhorse	Chevalier cuivré	QC
G1Q	*Spirinchus sp. 1*	Pygmy Longfin Smelt	Éperlan d'hiver nain	BC
Butterflies (5)				
G1	*Coenonympha nipisiquit*	Maritime Ringlet Satyre	fauve des Maritimes	NB, QC
G1?	*Colias johanseni*	Johansen's Sulphur	Coliade de Johansen	NU
G2	*Lycaena dospassosi*	Salt Marsh Copper	Cuivré maritime	NB, QC
G3	*Boloria natazhati*	Beringian Fritillary		BC, NT, NU, YT
G3G4	*Papilio brevicauda*	Short-tailed Swallowtail	Papillon queue-courte	NB, NL, NS, QC
Vascular Plants: Ferns and relatives (1)				
G1	*Botrychium pseudopinnatum*	False Northwestern	Moonwort Botryche	ON
Vascular Plants: Flowering Plants (45)				
GH	*Draba yukonensis*	Yukon Whitlow-grass	Drave du Yukon	YT
G1	*Atriplex nudicaulis*	Baltic Saltbush	Arroche hâtive	NL, NS
G1	*Braya longii*	Long's Braya	Braya de Long	NL
G1	*Braya pilosa*	Hairy Rockcress Braya		NT
G1	*Claytonia ogilviensis*	Spring Beautry Claytonie		YT
G1	*Draba kluanei*	Kluane Whitlow-grass	Drave de Kluane	YT
G1	*Draba pycnosperma*	Dense Whitlow-grass	Drave graines imbriquées	NL, NS, QC

(Continued on next page)

Global Rank*	Scientific Name	Common Name (English)	Common Name (French)	Province/Territory
Vascular Plants: Flowering Plants (cont'd)				
G1	*Draba scotteri*	Scotter's Whitlow-grass	Drave de Scotter	YT
G1	*Puccinellia poacea*	Goose Grass	Puccincellie	NT, NU
G1	*Salicornia borealis*	Boreal Saltwort	Salicorne boréale	MB, YT
G1	*Salix chlorolepis*	Green-scaled Willow	Saule à bractée vertes	QC
G1?	*Puccinellia macra*	Bonaventure Island Alkali Grass	Puccincellie	NB, QC
G1?Q	*Rubus adenocaulis A*	Bramble	Ronce	NS
G1?Q	*Rubus emeritus A*	Bramble	Ronce	NB
G1Q	*Potamogeton methyensis*	Methy Lake Pondweed	Potamot à lac Methy	SK
G2	*Braya fernaldii*	Fernald's Braya	Braya de Fernald	NL
G2	*Geum schofieldii*	Queen Charlotte Avens	Benoîte	BC
G2	*Salix jejuna*	Barrens Willow	Saule des landes	NL
G2	*Salix raupii*	Raup's Willow	Saule de Raup	AB, BC, NT, YT
G2	*Salix turnorii*	Turnor Willow	Saule de Turnor	SK
G2	*Saxifraga gaspensis*	Gaspé Saxifrage	Saxifrage de Gaspé	QC
G2	*Stenotus macleanii*	Maclean's Goldenweed	Sténote de Maclean	YT
G2	*Symphyotrichum laurentianum*	Gulf of St. Lawrence Aster	Aster du golfe Saint-Laurent	NB, NS, QC
G2?	*Crataegus Canadensis*	Canada's Hawthorn	Aubépine du Canada	QC
G2?	*Rubus suppar A*	Bramble	Ronce	NB, NS
G2?	*Rubus weatherbyi*	Weatherby's Dewberry	Ronce de Weatherby	NB, NS
G2?Q	*Rubus quaesitus A*	Bramble	Ronce	NB, PE
G2G3	*Deschampsia mackenzieana*	Mackenzie Hairgrass	Deschampsie du bassin du Mackenzie	SK
G2G3	*Puccinellia bruggemannii*	Prince Patrick Alkali Grass	Puccincellie	NU
G2G3	*Salix silicicola*	Felt-leaf Willow	Saule silicicole	NU, SK
G2G4	*Atriplex franktonii*	Frankton's Saltbush	Arroche de Frankton	NB, NS, PE, QC
G2G4	*Puccinellia ambigua*	Alberton Alkali Grass	Puccinellie trompeuse	NB, NL, NS, ON, PE, QC
G3	*Antennaria eucosma*	Newfoundland Pussytoes	Antennaire élégante	NL, QC
G3	*Arnica louiseana*	Lake Louise Arnica	Arnica à lac Louise	AB, BC
G3	*Bidens amplissima*	Vancouver Island Beggar-ticks	Grand bident	BC
G3	*Carex rufina*	Snowbed Sedge	Carex à écailles rousses	MB, NT, QC
G3	*Enemion savilei*	Savile's False Rue-Anemone	Isopyre de Savile	BC
G3	*Pedicularis palustris*	Marsh Lousewort	Pédiculaire des marais	NL, NS, QC
G3	*Puccinellia deschampsioides*	Polar Alkali Grass	Puccinellie	MB, NT, NU, QC, YT
G3	*Ranunculus allenii*	Allen's Buttercup	Renoncule d'Allen	NL, NU, QC
G3	*Saxifraga taylorii*	Taylor's Saxifrage	Saxifrage de Taylor	BC
G3?	*Draba subcapitata*	Ellesmere Island Whitlow-grass	Drave	NU
G3?	*Parrya arctica*	Arctic False-wallflower		NT, NU
G3?	*Puccinellia laurentiana*	Tracadigash Mountain Alkali Grass	Puccinellie	NB, PE, QC
G3G4	*Saxifraga stellaris*	Starry Saxifrage	Saxifrage étoilée	NL, NU
G3G4Q	*Festuca frederikseniae*	Viviparous Fescue	Fétuque de Frederiksen	NL, QC

*NatureServe™ Global Conservation Rank: G = global status; 1-5 = risk of extinction (1 = high; 5 = low), multiple rankings (e.g., G2G5) indicate range of likely status; X = likely extinct; ? = assigned ranking uncertain; Q = denotes questionable taxonomy.